Proceedings of the

FIFTY-SECOND ANNUAL

Gulf and Caribbean Fisheries Institute

KEY WEST, FLORIDA USA

NOVEMBER 1999

HOSTED BY:

Florida Fish and Wildlife Conservation Commission

Florida Sea Grant College Program

Florida Keys National Marine Sanctuary

FORT PIERCE, FLORIDA 2001

Gulf and Caribbean Fisheries Institute, Inc.

(A South Carolina Non-Profit Corporation)

Executive Office
Harbor Branch Oceanographic Institution, Inc.
5600 US Highway 1 North
Fort Pierce, FL 34946

Proceedings of the

FIFTY-SECOND ANNUAL

Gulf and Caribbean Fisheries Institute

KEY WEST, FLORIDA

NOVEMBER 1999

Library of Congress Catalog Card Number
52-033783

Edited by:

R. LeRoy Creswell

FORT PIERCE, FLORIDA 2001

We would like to extend our grateful appreciation to the following sponsors of the 52nd Gulf and Caribbean Fisheries Institute

American Fisheries Society - Florida Chapter

The Government of Bermuda

Center for Marine Conservation

Coastal Research and Education

Florida Fish and Wildlife Conservation Commission

Florida Sea Grant College Program

Harbor Branch Oceanographic Insitution, Inc.

Konky Joes Island and Resort Wear

National Oceanic and Atmospheric Administration
Florida Keys National Marine Sancturary
Intergovernmental and Recreational Fisheries
Office of Habitat Conservation
Office of Special Projects
Southeast Fisheries Science Center

The Nature Conservancy

US Environmental Protection Agency

University of Miami - Rosenstiel School of Marine and Atmospheric Sciences

Opening Remarks

52nd Gulf and Caribbean Fisheries Institute Meeting
November 1-5, 1999
Key West, Florida

Gregg Waugh, Chairman

Good morning and welcome to the 52nd Annual Gulf and Caribbean Fisheries Institute Meeting. It gives me great pleasure to be here in beautiful Key West, Florida meeting with old friends, making new friends, and exchanging ideas at a GCFI meeting.

Most importantly, I would like to take this opportunity to thank Bob Glazer for organizing this year's meeting. Bob and all the volunteers serving on various committees, and the volunteers on his staff, all deserve a hearty round of applause.

LeRoy Creswell is recovering from an illness and will not be here with us in Key West. LeRoy asked that I pass along his welcome and his wish for an interesting and productive meeting. LeRoy also deserves our thanks for all his work on behalf of GCFI.

The Gulf and Caribbean Fisheries Institute (GCFI) was founded in 1947. For 37 years, GCFI operated as an informal association under the sponsorship of the University of Miami. In 1985, with encouragement from the University, GCFI became an independent not-for-profit corporation formally dedicated to its original purposes. GCFI is governed by a Board of Directors elected by and from its membership. Particular effort is made to ensure balanced representation from throughout the region in its annual programming and decision-making processes. You the membership are what GCFI is all about. Please get involved by serving on committees, reviewing manuscripts, acting as a moderator, or even hosting a GCFI meeting. Make GCFI what you want it to be by getting involved.

Bob Glazer and the Program Committee have put together an excellent meeting for us to enjoy. Sessions will address demersal fisheries (particularly snappers, groupers & grunts), invertebrate fisheries (particularly conch and spiny lobster), socio-economic aspects of fisheries management, Caribbean aquaculture, recreational fisheries, pelagic fisheries, habitat assessment and human impacts, marine fishery reserves, and the importance of Caribbean recruitment to Florida's fisheries. In addition, this year will see the return of a trade show/reception and poster session/reception. These two events should provide many opportunities to meet and share information which is one of the primary functions of GCFI. Also, a special HAACP Workshop will be held to provide the training necessary to obtain a FDA food handler's certification. If you have suggestions for future meetings, please talk with Bob or another of the Board members.

The GCFI Business Meeting will take place Wednesday morning at 8:00 a.m. Please come to this important meeting and participate in designing future activities involving GCFI. Once again, Welcome to the 52nd Annual GCFI Meeting and enjoy!

TABLE OF CONTENTS

Demersal Fisheries

Invertebrate Fisheries

Socio-Economics of Fisheries Management

Advances in Caribbean Aquaculture

Recreational Fisheries

Habitat Assessment

Pelagic Fisheries

Anthropogenic Impacts of Coastal Habitats

Marine Fisheries Reserves

The Origin of Florida Fisheries

Preliminary Results on the Reproductive Cycle of the Black Grouper, *Mycteroperca bonaci*, from the Southern Gulf of Mexico

XIMENA RENÁN[1], THIERRY BRULÉ[1], TERESA COLÁS- MARRUFO[1],
YAZMÍN HAUYON[2] and CHRISTIAN DÉNIEL[3].
[1]*Centro de Investigación y Estudios Avanzados del IPN,*
Unidad Mérida, México
[2]*Institut de Biologie Animale*
Université de Lausanne, Suisse
[3]*Université de Bretagne Occidentale*
Institut Universitaire Européen de la Mer
Brest, France

ABSTRACT

The Black Grouper, *Mycteroperca bonaci*, locally known as "Negrillo" is one of the 17 grouper species found on the Campeche Bank. Its importance relies on being after the *Epinephelus morio*, the largest catching volume specie and a very important source of food for the state of Yucatan, Mexico. In spite of this fact, the *M. bonaci* has not been well studied and little is known about its biology, not only in this particular zone but also throughout all its distribution area. This paper presents results on sex determination, sexuality and sexual maturation of *M. bonaci*. After three years of sampling on the Campeche Bank a gonadic histologycal analysis was done. It included more than 800 samples of individuals between 50 to 130 cm (FL). Sex determination studies included 605 females, 205 males and five individuals in sexual inversion. Results confirm that *M. bonaci* is a protogynous hermaphrodite. Considering the sexual classes and seasonal variation in mean of GSI, gonadal activity was found to be from December to March, with a spawning period clearly identified in January to March.

KEY WORDS: Campeche Bank, hermaphroditism, *Mycteroperca bonaci*.

INTRODUCTION

The Campeche Bank is a continental platform of about 175,000 km^2, which surrounds the Yucatan Peninsula and is limited to 25 km inwards the Gulf of Mexico (García 1980). This Bank is part of the region covered by the Western Central Atlantic Fishery Commission (WECAFC), the international organization which looks after the fisheries management of the Caribbean Sea, the Gulf of Mexico and their Atlantic approaches, form Cape Hatteras to Recife, Brazil (Brulé and Déniel 1996).

This platform is the center of fisheries in the state of Yucatan. Landings of octopus and groupers comprise the most important fisheries in this area. In fact, by 1995 grouper landings comprised 91.5% of the entire national fishing production in Mexico. Based on studies by Colás-Marrufo et al. (in press) and Tuz- Sulub (1999), 17 different species of groupers were identified on the Campeche Bank. Of these species, the most important are the red grouper (*Epinephelus morio*), the black grouper (*Mycteroperca bonaci*) and the gag (*Mycteroperca microlepis*). The entire catch was established from these three species, which contribute the greatest numbers of individuals and biomass. The black grouper contributes 40% of the total weight of the entire state production (Tuz-Sulub 1999).

Black groupers occur off Bermuda and are distributed from southern Florida through the southeastern part of the Caribbean and West Indies to northern South America and throughout the Gulf of Mexico. Its habitat is irregular bottom, such as rocky relief and coral reefs (Cervigón and Velasquez 1966). Young black groupers tend to occur in shallow waters, whereas larger fish are restricted to depths greater than 20 m (Fischer 1978). This specie is reported to have a length of at least a meter and weight more than 65 kg (Böhlke and Chaplin 1968, Manooch and Manson 1987, Heemstra and Randall 1993).

According to Sadovy (1994), evidence exists that the populations of groupers from the west center Atlantic show typical signs of overfishing. There has been an abrupt decrease in the average of size and number of the fishing stock, an increase in the effort required to generate maximum sustainable yield (MSY) and the extinction or disruption of reproductive aggregations. Although none of this evidence appears in grouper populations on the Campeche Bank, leaders of the state government of Yucatan have expressed their concern to the Fisheries Secretary, in particular that no population of groupers nn the Campeche Bank has an established management plan.

Nevertheless, the selection of closure season or minimum size of capture is very difficult due to the fact that the fishery in the Campeche Bank include 17 different groupers species, and information on their biology is needed. This is one of the main objectives of the present study, to provide with essential knowledge on the reproductive biology of the black grouper such as sex determination, sexuality and sexual maturation. This data is required to better understand and manage the grouper stocks of the Campeche Bank.

METHODS

Samples of the black grouper were taken from April 1996 to June 1998. Four Yucatan commercial fishing boats collected the samples, most of them in the northeastern part of the Campeche Bank, at 29 different fishing sites, using standard hook and line gear (Figure 1). Data obtained for each sample included collection date, location, fork (FL) and standard length (SL) (cm), total (TW) and eviscerated weight (EW) (g), gonads weight (GW) and liver weight (LW) (g. A small part of the gonad (2 cm^3) was removed and fixed in Bouin's fluid for the histological study of oogenesis and spermatogenesis. The preserved gonads were embedded in paraffin, thin sectioned at 6 µm, and stained as suggested in Gabe (1968). Gonad sections were examined to determine sex and gonad development. The different stages of the oogenesis and spermatogenesis were taken from the suggested microscopic cellular characteristics established by Moe (1969) and Brulé and Déniel (1996) for *Epinephelus morio*. According to Coleman's formula (Coleman, 1991), an "effective diameter" ($d_e = [ab^2]^{1/3}$, with a: major axis and b: minor axis) was calculated for 100 oocytes of each oocyte stage.

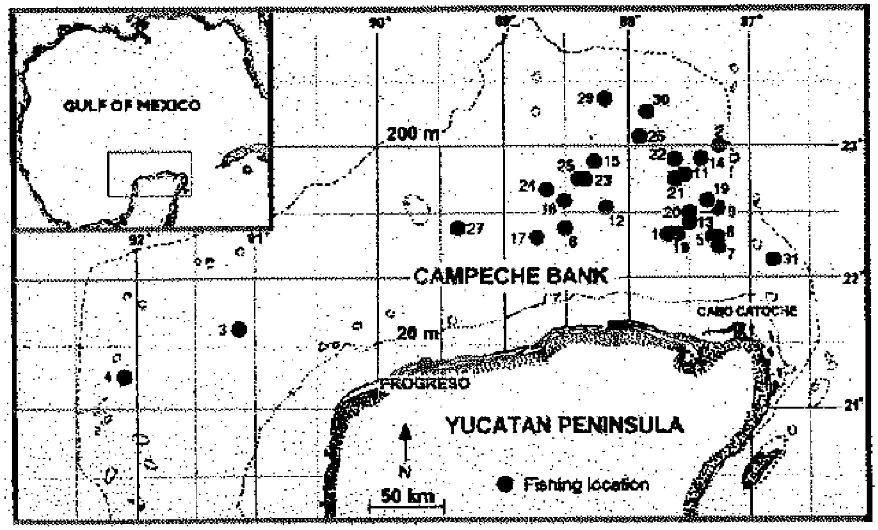


Figure 1. Map of the Campeche Bank and the north coast of the Yucatan Peninsula, showing the 29 fishing sites where black grouper individuals were collected.

The standard of judgement used to distinguish the transitional individuals was taken from five criteria established by Sadovy and Shapiro (1985), Collins et al. (1987) and Shapiro (1987). The distribution of sex in 5 cm FL size groups were compared through a Kolgomorov- Smirnov test, and the mean size per sex were established with a Z test (Scherrer 1984). To ascertain if the sex ratio obtained could be compared to an equilibrated sex ratio (1:1) a Chi- Square test was accomplished (α = 0.05) (Scherrer 1984). For the gonad development, specimens were gathered into nine different classes, which fits in one of the histological conditions of female, transitional and male individuals (Brulé and Déniel, 1996). An accumulated frequency histogram of sexual classes was made for each sex during an annual cycle. The reproductive state of the Black grouper was further examined by calculating a gonadosomatic index (GSI = GW*100/EW) for each individual. Data from the same months for the three years were combined and the results were presented as monthly mean values.

RESULTS

A total of 815 Black Groupers, ranging from 55 cm to 130 cm (FL) and 650 g to 34,500 g (EW) were examined during this study. All of the individuals were captured offshore at different depths in the Campeche Bank, from April 1996 to June 1998. By histological analysis we determined 74% of the captured individuals were females (N = 605), 25% were males (N = 205) and 1% (N = 5) were transitional individuals. Female sizes ranged from 55 cm to 125 cm, males from 85 cm to 130 cm (FL) and individual in sexual inversion ranged from 90 cm to 110 cm (Figure 2). Three criteria were observed:

i) 75.5% (N = 154) males had a lumen inside their testis
ii) 35.5% exhibited spermatic sinuses in the testicular membrane (N = 73).
iii) The presence of five transitional individuals 1% (N = 5) Sadovy and Shapiro (1985). The size frequency distribution of females was different from that of the males (Kolmogorov- Smirnov, D = 0.756). Besides females had a mean of the fork length (96.82 ± 12.23 cm) smaller than the mean fork length of males (105.13 ± 13.98 cm)(Z = 24.88). The mean fork length of transitional individuals was 107.3 ± 9.67 cm.

The sex ratio (M:F) considered only females and males (N = 810), was 1:3 (Chi- Square = 52.84, DF = 1). A Chi- Square test was accomplished for each of the different size classes established previously (5 cm each), only where males and females were found together (Table 1).

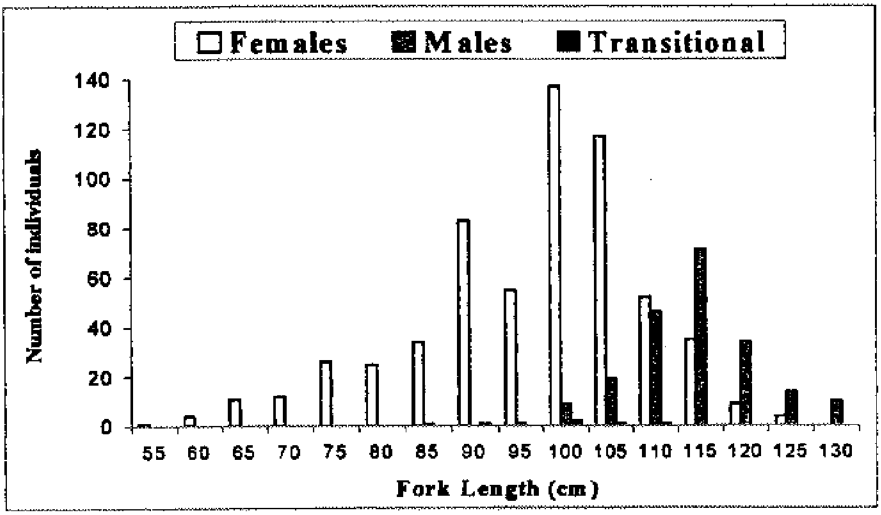


Figure 2. Size frequency distribution of *M. bonaci* caught on the Campeche Bank (females = 605, males = 205 and transitional = 5 individuals)

Histological analysis established that a number of males displayed remaining oocytes inside their testes (11.25%, N = 27), the other males did not show this remaining cellular condition (88.75%, N = 178). It was established by a Kolgomorov-Smirnov test, that the distributions of males with or without remaining oocytes, differed significantly (D = 0.756). On the other hand, the mean of fork length of males with oocytes was not significantly different from the mean of males without oocytes in their testis (Z = 0.000002) (Figure 3).

Six stages of oogenesis were established through observable cytologic changes during gametogenesis (post-ovulatory follicules and atresic oocytes were not considered as stages). The effective diameter of 100 oocytes in each of the six cellular stages, shows a quantitative characteristic. The mean diameter of the oocytes increased gradually from stage I (9.31 ± 3.82 μm) throughout stage VI (870.84 ± 114.66 μm) (Table 2). Five stages of spermatogenesis were observed.

Table 1. Sex ratio by size classes of *M. bonaci* on the Campeche Bank

Class Distinction	Size Classes	Males	Females	Sex Ratio
55	52.6 - 57.5	0	1	-
60	57.6 - 62.5	0	4	-
65	62.6 - 67.5	0	11	-
70	67.6 - 72.5	0	12	-
75	72.6 - 77.5	0	26	-
80	77.6 - 82.5	0	25	=
85	82.6 - 87.5	1	34	1:34*
90	87.6 - 92.5	0	83	-
95	92.6 - 97.5	1	55	1:55*
100	97.6 - 102.5	9	137	1:15*
105	102.6 - 107.5	19	117	1:06*
110	107.6 - 112.5	46	52	1:1.13**
115	112.6 - 117.5	71	35	1:0.49**
120	117.6 - 122.5	34	9	1:0.26*
125	122.6 - 127.5	14	4	1:0.28*
130	127.6 - 132.5	10	0	-
	TOTAL	**205**	**605**	**1:2.95***

* Statistically different **Not statistically different

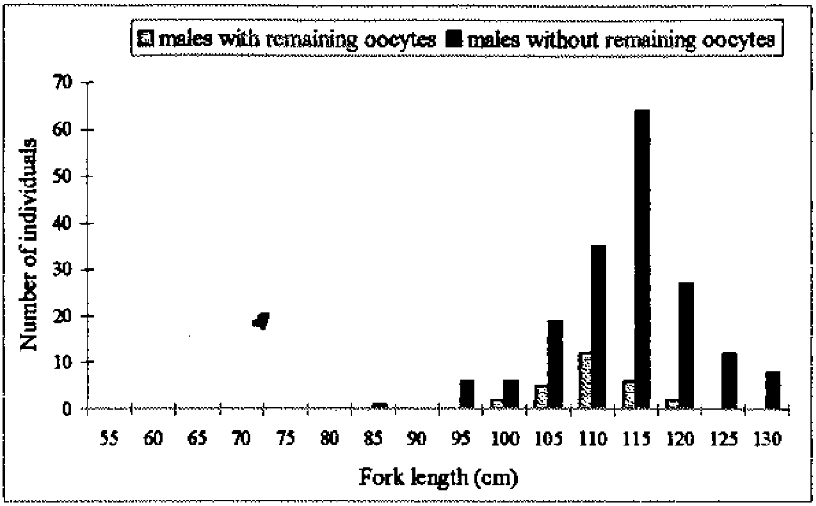


Figure 3. Frequency distribution of *Mycteroperca bonaci* males with and without remaining oocytes in the tests.

Table 2. Effective diameter calculated for the oocytes and their nucleus. (Size range for each stage in parenthesis).

Cellular Stage	Cellular diameter (mm)	Nuclear diameter (mm)
I Primary oocyte	15.062 ± 5.23 (1.745 - 19.195)	2.769 ± 2.076 (0.0769 - 10.32)
II Immature oocyte	23.186 ± 6.637 (9.096 - 56.726)	10.111 ± 3.636 (2.672 - 23.287)
III Primary vitellogenesis	59.649 ± 16.562 (19.1 - 77.591)	22.136 ± 6.73 (6.6060 - 26.747)
IV Early Secondary vitellogenesis	161.762 ± 41.463 (34.085 - 192.316)	34.44 ± 7.858 (9.125 - 29.372)
V Late secondary vitellogenesis	262.961 ± 66.958 (199.483 - 576.988)	45.862 ± 8.188 (13.152 - 43.561)
VI Hyaline oocyte*	870.84 ± 114.66 (193.60 - 883.0)	

* Value obtained by measuring, under a binocular microscope, fresh hyaline oocytes (N = 100) samples from ripe ovaries

Changes in the monthly mean gonadosomatic indexes confirmed the histological observations of spawning and resting periods. The relative weight of ovaries increased slowly from October to November and rapidly from January to February. The GSI of males did not show great variations throughout the year. The GSIs reached maximum values in January (female: 1.4%; males: 0.14%) and February (female: 2.2%; males 0.18%) indicating ripening. Relative weight began to decrease in March to reach its minimum values in May for females (0.12%) and on March for males (0.11%) indicating that gametes had been shed (Figure 4).

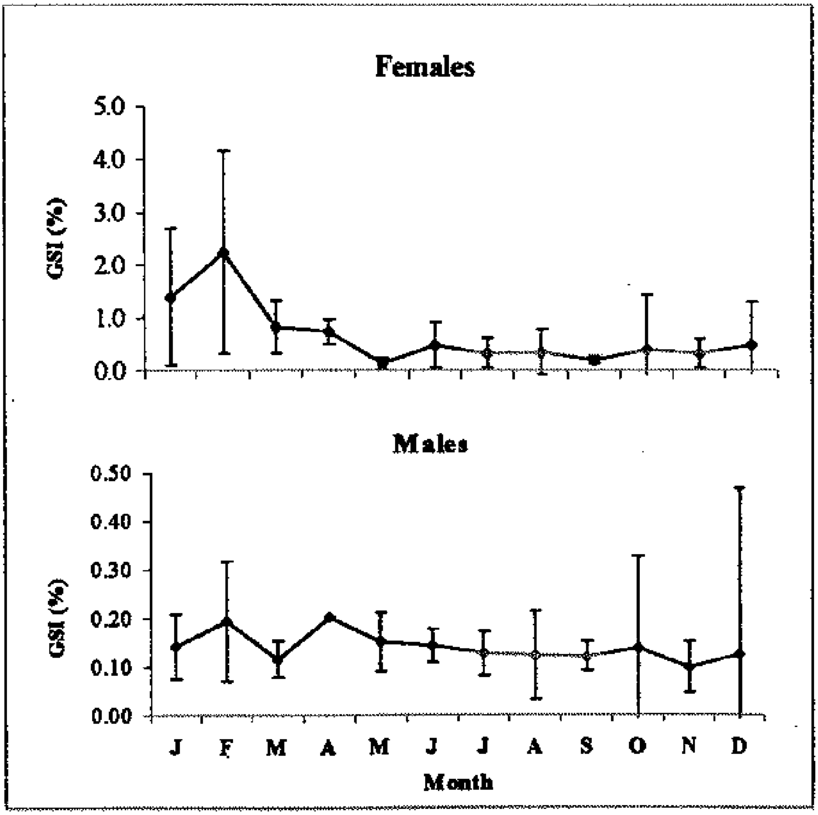


Figure 4. Seasonal variations in mean gonadosomatic index (%) for females and males of *M. bonaci* from Campeche Bank. Vertical bars indicate standard deviation.

Vitellogenesis started in September, for the most precocious females, and continued throughout April. The highest percent of ripening occured during December to April and for ripe females from January to April. In May only immature and atretic vitellogenic oocytes remained in the ovaries, indicating that females have entered the spent-resting period. However, immature females were found throughout the annual cycle (Figure 5).

Ripening males were observed throughout the annual cycle with the exception of April, during which only one ripe individual was found. Riep males were observed every month of the year, many entering a spent period from January through July (Figure 5).

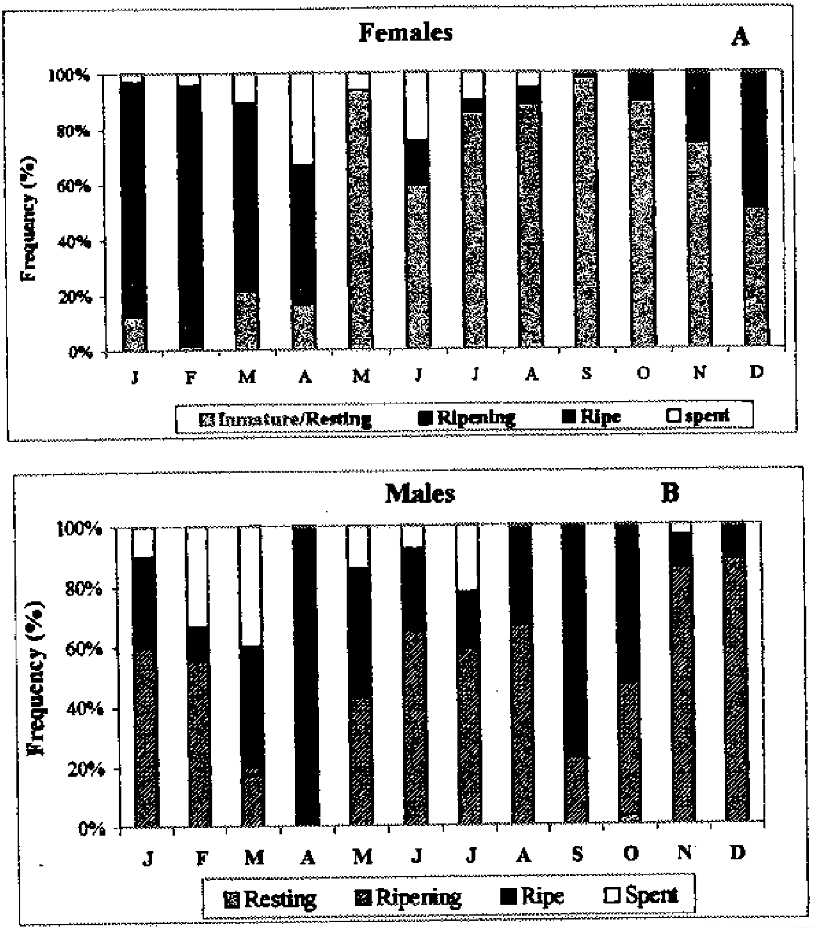


Figure 5. Percent frequency distribution of gonad development classes by month during an annual cycle of *Mycteroperca bonaci* (A: females; B: males)

DISCUSSION

The results indicate that *Mycteroperca bonaci* is a protogynous hermaphrodite; smaller individuals are females and older individuals become actively reproductive males (Smith 1959, Collins et al., 1987, Shapiro 1985, Shapiro 1987 y García-Cagide y García 1996). Three of the five criteria suggested by Sadovy and Shapiro (1985) for the classification of a protogynous hermaphrodites were identified. Males had a lumen inside their testis, present spermatic sinuses into the testicular membrane, and five individuals were established to be transitional individuals with an ovarian lumen along with spermatogonian and spermatocites groups. This is the most important criterion to established hermaphroditism in any specie. According to Shaprio (1987), one of the several difficulties of defining the time of sex change form the occurrence of transitionals in monthly samples, is the relatively small number of transitional gonads found. One reason for finding small numbers of transitional individuals is that sex change may be completed very quickly (Brulé and Déniel 1996). Transitional black groupers were found during five months of the year, including one month of the peak spawning. Nevertheless, García-Cagide and García (1996) established that the black grouper is a monadric specie, and this corroborates with the results obtained in this study, as the presence of the ovaric lumen inside the testis of 75% of the males. However, more information about juveniles of the population is needed in order to establish the sex of young individuals, and therefore conclude if all adults males would have previously been females.

Additional criteria for the classification of protogynous hermaphrodite are the bimodal population size distribution and sex ratio. For instance, in size distribution bimodal histograms, females tend to appear at smaller size classes (55 - 125 cm) and males were found in bigger size classes (85 - 130 cm).

The sex ratio was 1:3 (M:F)(N = 810) clearly showing bias towards females, as expected, since *M. bonaci* is a protogynous hermaphrodite. This sex ratio is different from the one obtained by García-Cagide and García (1996) for *M. bonaci* in Cuba (1: 30.3) (N = 209), which is the only study about the reproduction of the black grouper. The results suggest that for the smaller size classes the sex ratio was biased towards the females (85 - 105 cm) and in the upper size classes (120 - 130 cm) biased towards the males .

Another criterion of protogynous hermaphroditism, is the presence of remaining cells as oocytes inside the mature testis. 11.25% of males did not have remaining oocytes (100 - 120 cm FL) and 88.75% did present this feature (130 - 189 cm FL). This criterion is the least reliable because it could indicate other characteristics, such as a bisexual period or juvenile female phase and its found even in some gonocoric species (Sadovy and Shapiro 1987).

Mature males and females were found in all size classes from 60 – 130 cm FL. These agree with results found in other studies, such as García–Cagide and García (1996) in which mature females and males were found at 85 and 100 cm (FL), respectively. In addition, Valdés Muñoz (1980) found on the Cuban platform five mature females between 57.5 and 64.5 cm (FL) and three males with 62.3 – 67.8 cm (FL). Similarly, Smith (1959) found fourteen mature females of about 45.2 cm and seven mature males with 107.5 cm (FL).

Although some grouper species are known to spawn over 6 - 8 months, most spawn over 1- 5 months, and many spawn primarily during 1 - 2 months (Shapiro 1987, Bullock and Smith 1991). Histological observations of the changes in the gonad development and seasonal variation in mean GSI, indicated a gonadal activity from September to February, with a peak spawning activity in January to March and a resting period from May to September. García-Cagide and García (1996) reported that the spawning period of the black grouper happens during winter with a peak in November and February. Smith (1961) reported some mature black grouper individuals on Campeche Bank from July to August and in the Bahamas in January. Erdman (1956) observed mature individuals in February at Puerto Rico. These results agree with the present study, and it could be concluded that black groupers spawn in winter, primarily in January to March. Even though there is no evidence that black groupers form spawning aggregations on the Campeche Bank, Domeier and Colin (1997) established that *M. bonaci* aggregates in the same sites as *E. striatus* in Belize (January and February) and in Honduras (January), although these spawning aggregations have not been studied.

The monthly samplings of *Mycteroperca bonaci* were homogeneous since all size classes were established between the same rank (females: 55 - 125 cm; males: (85 - 130 cm). This could be achieved because the fishing method was selective and it was included in the same fishing plan. There are no fishing regulations for black grouper in the southeastern part of the Gulf of Mexico, with the exception of 3,900 tons that the Cuban fleets are allowed to catch (Brulé y Colás, 1997). One proposal made to the Yucatan Government for the regulation of groupers is to establish a minimum catch size over 35 cm (FL). Of the 815 samples of black grouper captured during this study, all individuals were beyond this limit. It has been established that adult groupers, including black grouper, live in deep reef zones, were as the juveniles live in coastal zones (Jory and Iversen 1989, Parrish 1987, Sluka and Sullivan 1996). Therefore, we could conclude that most of the individuals captured were adults based on the vertical distribution of most groupers present. One of the main purposes of this study was to contribute with reproductive data of one of the most important fishing specie on the Campeche Bank, and therefore help to establish a regulation program for this resource.

AKNOWLEDGMENTS

This research was supported by grant 2184P-B9507 from the Science and Technology Council of Mexico (CONACYT), grant c1*0432 ME (JR) from the European Community (EC) and the SEMARNAP/ S.S.S. "24 de Febrero"/ CINVESTAV agreement for using the fishing vessel "UNICAP VII". For the assistance during the course of this study, we would like to thank R. Robles de Benito, V. Alcantar-Cárdenas and M. Garduño-Andrade form SEMARNAP/ INP-CRIPY (Yucatán); J. Peraza-Menéndez and M. Castillo-Martínez from CECADESU/ CREDES - Yucalpetén; J. Rodríguez-Felix from S.S.S. "24 de Febrero" fishing cooperative; A.M. Pech from CONYUC fish house; and L. Contreras García, the port Captain of Progreso. For helping with all the aspects of fields collections, we also thank M. Sánchez-Crespo, V. Durarte- García, S. Mena- González, c. Ureña- chio, J. Hernández-Viguegas, T. Ramírez- Hernández and P. Mina-Coello.

LITERATURE CITED

Böhlke E.J. and C.C.G. Chaplin. 1993. *Fishes of the Bahamas and Adjacent Tropical Waters.* University of Texas, Press, Austin.

Bullock, L.H. and G.B., Smith. 1991. *Seabasses (Pisces:Serranidae).* Memoirs of the Hourglass cruises. Vol 8, Part 2, Florida.

Brulé T. y T., Colás. 1997. Explotación de los serránidos en el sureste del Golfo de México: evolución y situación actual. *Proc. Gulf and Carib. Fish. Inst.* **49**:173-214.

Brulé, T. and C. Déniel. 1996. Biological research on the red grouper (*Epinephelus morio*) from the southern Gulf of Mexico. Pages 28-42 in: . Arreguín-Sánchez, J. Munro, L. Balagos and D. Pauly (eds) *Biology, Fisheries and Culture of Tropical Groupers and Snappers.* ICLARM Conf. Proc.

Cervigón, E. y F. Velásquez. 1966. Las especies del género *Mycteroperca* en las costas de Venezuela. Estación de Investigaciones Marinas de Margarita. Fundación la Salle de Ciencias Naturales. Contribución No. 23. 77–143.

Colás-Marrufo, T., T. Brulé. y C. Déniel. 1998. Análisis preliminar de las capturas de meros realizadas a través de unidades de la flota mayor en el sureste del Golfo de México. *Proc. Gulf Carib. Fish. Inst.* **50**:780-803.

Coleman, R.M. 1991. Measuring parental investment in nonsherical eggs. *Copeia* **4**:1092 –1098.

Collins, M.R., C.W. Walte., W.A. Roumillat and P.L. Stubbs. 1987. Contribution to the life history and reproductive biology of gag

(*Mycteroperca microlepis*) (Serranidae), in South Altantic Bight. *Fisheries Bulletin* **85**:648-653.

Domeier. M y P. Colin. 1997. Tropical reef fish spawning aggregations defined and review. *Bull. Mar. Sci.* **60**(3):698-726.

Erdman, D. 1956. Recent Fish Records from Puerto Rico. *Bull. Mar. Sci.* **6** (4):315–339.

Fischer, W. (Ed). 1978. *Species identification Sheets for Fisheries Purposes Western Central Atlantic (Fishing area 31).* Vol 1-7. FAO, Rome.

Gabe. 1968. *Techniques Histologiques.* Masson, Paris. 113 pp.

García. 1980. Caracterización general del Banco de Campeche. Centro de Inv. Pesquera. Rev. Cub. Inv. Pes. 5 (2).

García-Cagide, A., y T., García. 1996. Reproducción de *Mycteroperca bonaci* y *Mycteroperca venosa* (Pisces: Serranidae) en la plataforma cubana. *Rev. Biol. Trop.* **44**(2): 771- 780.

Heemstra and Randall. 1993. *FAO Species Catalogue. Groupers of the world (family Serranidae, subfamily Ephinephelinae).* An annotated and ilustrated catalogue of the grouper, rockcod, hind coral, grouper and lyretail species known to date. FAO Fisheries Synopsis. FAO, Rome. (125) 16.

Jory, D., and E. Iversen. 1989. Species Profiles: Life histories and Enviromental Requirements of Coastal Fishes and Invertebrates (South Florida). *U.S. Fish Wild. Ser. Biological Report* **82**:11-110.

Manooch, C., and D. Mason. 1987. Age and growth of the warsaw gouper and black grouper from the southeast region of the United States. *Northeast Gulf Science* **9**(2):65-75.

Moe, M.A. 1969. Biology of the Red Grouper *Epinephelus morio* (Valenciennes) from the Eastern Gulf of Mexico. Prof. Pap. Ser. Fla. Dep. Nat. Resour. Mar. Res. Lab. **10** 95 pp.

Parrish, J.D. 1987. The Trophic of Snappers and Groupers. Pages 405 - 463. in: Polunin and C.M. Roberts (eds). *Reef Fisheries.* Capman and Hall, London.

Sadovy, Y. 1994. Grouper stocks of the Western Central Atlantic: the need for management and managemente needs. *Proc. Gulf and Carib. Fish. Inst.* **43**:43-64.

Sadovy, Y., and D.Y. Shapiro. 1987. Criteria of diagnosis of hermaphroditism in fishes. *Copeia* **1**:136-156.

Shapiro, D.Y. 1985. Sex reversal and sociodemographic processes in coral reef fishes. Pages 103-118 in: G.W. Potls and R. J. Wootton (eds). *Fish Reproduccion: Strategies and Tactics.* 3rd Ed. Academic Press, London.

Shapiro, D.Y. 1987. Reproduction in groupers. Pages 295-327 in: J.J. Polovina and S. Ralston (eds). *Tropical Snappers and Groupers. Biology and Fisheries Management.* Westview Press, Boulder.

Scherrer, B. 1984. *Biostatistique*. Gaëtan Morin Editeur, Bocherville, Québec.

Sluka, R., and K.M., Sullivan. 1996. The Influence of habitat on the size distribution of groupers in the upper Florida Keys. *Enviromental Biology of Fishes* **47**:177-189.

Smith, C.L. 1959. Hermaphroditism in some serranid fishes from Bermudas. *Pap. Michigan Acad. Sci.* **44**:111-119.

Smith, C.L. 1961. Synopsis of biological data on groupers (*Epinephelus morio* and allied genera) of the Western Atlantic. *FAO Fish. Biol. Synop.* **23**: 1–62.

Tuz-Sulub, A.N. 1999. Composición, distribución e importancia pesquera de los serrándios (subfamilia: Epinephelinae) en el Banco de Campeche. Thesis. Universidad Autónoma de Yucatán. Licenciatura en Biología. México.

Valdés- Muñoz, E. 1980. Toxicidad y Alimentación de algunos peces sospechosos de provocar la ciguatera. Informe Científico- Técnico No. 123 pp.

Tag and Recapture Study of Red Hind and Coney at Three Spawning Aggregation Sites Off the West Coast of Puerto Rico

AIDA ROSARIO JIMÉNEZ and MIGUEL FIGUEROLA FERNÁNDEZ
Fisheries Research Laboratory/BFWL/PR-DNER
P.O. Box 3665, Marina Station
Mayaguez, Puerto Rico 00681

ABSTRACT

A pilot study of mark-recapture for red hind (*Epinephelus guttatus*) and coney (*Cephalopholis fulva*) was started at three closed spawning aggregations off the West Coast of Puerto Rico. It is expected that the closed season and area, will provide for the population, to recover from the heavy fishing pressure to which they are submitted. Also it will provide a better understanding of the movement behavior during the spawning aggregation period of red hinds through other research methods, such as tag and release. A study of this type provides information on the ability of the species to identify their spawning site (homing) and their distribution. Luckhurst (personal communication) has shown that red hinds have the ability to home and identify particular spawning sites.

A total of 374 red hinds and 579 coneys were tagged and released between January and June 1999. At the beginning of the study (first two months) the animals were released, approximately nine kilometers from the area of capture. Thirty-five (35) recaptures of red hinds have been recorded, representing 7.49% of total tagged individuals. A few red hinds have been recaptured more than once, for instance one individual has been caught three (3) times at the site of first capture. To maximize the probability of recapturing coneys, a single site (Abrir La Sierra) was selected to capture and release the last 193 individuals. Thus far, no coney has been recaptured.

KEY WORDS: Coneys, red hinds, tagging

INTRODUCTION

Red hind has become the serranid of great commercial importance in Puerto Rico (Matos 1997). This species gained commercial value after the commercial extinction of the Nassau grouper in Puerto Rico fisheries. These two species of serranids share a common factor that could lead to the commercial extinction of the red hind also. They do aggregate to reproduce in very specific areas and time. This factors make them highly susceptible to be overfished, before they have the opportunity of reproducing. Red hinds are of great commercial importance through its range of distribution in the Caribbean. It is believed to be over fished in many areas besides Puerto Rico and the U.S.Virgin Islands (Beets and Friedlander 1992, Sadovy and Figuerola 1992, Sadovy 1994). The species composition of fisheries-independent surveys carried out by the Fisheries Research Laboratory by the end of the 1960s and beginning of the 1970s, placed red hinds as the fourth species of importance (Juhl 1969, Juhl 1972, Juhl and Suaréz-Caabro 1972, 1973). Due to the sharp decline of the major serranid species and the increase in landings of red hinds, there has been an increased interest in studies regarding the reproduction, and age and growth. Among those are Sadovy et al. (1992, 1994) and Shapiro et al. (1993).

One aspect of the red hind spawning aggregation that has yet to be defined is whether red hind travel to spawn to a determined area, every year. Sadovy et al. (1992) showed that red hind does not necessarily moves to the nearest spawning site. Shapiro et al. (1993) showed that specific sites of fish concentrations within identified areas may vary from year to year and even during an annual aggregation. Whether this situation is an artifact of over-exploitation of the aggregation or an intrinsic characteristic of it, is unknown. Once again, this situation poses a great constraint to management, since one may close a spawning aggregation site thinking that it will help the species, while it might be possible that the fish are no longer aggregating at that area to spawn. Reproductive strategies of red hinds are of particular interest to many researchers since they appear to represent an intermediate between the extremes shown by other species. Red hinds are protogynous hermaphrodites, which form short-term spawning aggregations (Burnett-Herkes 1975, Colin et al. 1987, Sadovy et al. 1992). Individuals may live for 18 years or more and, during the non-reproductive season, live in overlapping home ranges (García-Moliner 1986).

Coneys are members of the serranidae exploited with red hinds. Commercial landing data of this species are not as specific as for red hinds since they are reported under the groupers classification. Overall coneys fluctuate in being among the three most caught species in fisheries-independent surveys. They also make up the second most important grouper species in commercial landings data. In contrast to red hinds, coneys have not been reported to form spawning aggregations, although, they do often spawn at the same area of red hinds. During spawning aggregations of the red hind that occurred from December to March 1995, ripe coneys, both males and females, were collected from September thru March. The peak of ripe individuals was recorded in December (1994) and January (1995). Data suggested that coneys spawned in the same area used by red hinds, with a week of difference between them. Data also suggested that two spawning activities occurred, one in January and a second one during February. Colin et al. (1987) also reported spawning activity of coneys near a red hind spawning site off the south coast of Puerto Rico. In 1993-1994 was the first time we could prove that coneys spawn in the same area and in a similar period as red hinds (Rosario 1994). One factor that has made this possible was the monitoring of an area off the west coast platform, besides the site usually monitored at the Bajo de Cico. As proven by the data obtained (Rosario 1989), coneys are not as abundant at the Bajo de Cico (stations 95 and 96). Therefore, data collected at previous red hind spawning aggregations in that area did not show a clear pattern of coney reproductive strategies. The Abrir La Sierra area for sampling the red hind aggregation corresponded to the station that yielded the highest number of coneys in our surveys.

The Fisheries Research Laboratory (FRL) has monitored for the last ten (10) years a red hind spawning aggregation off the west coast of Puerto Rico (Rosario 1989, 1992, 1994, 1996). The area is known as the Bajo de Cico bank, located 28 km west northwest from Mayagüez. Two other aggregation sites have been monitored sporadically - the Abrir La Sierra found 28 km southwest of Mayagüez, and the Tourmaline Reef approximately 15 km west of Mayagüez. This monitoring has been used to characterize the reproduction of the red hinds in western Puerto Rico (Sadovy et al. 1994). Nevertheless, certain aspects remain unclear regarding the reproductive strategies of this species. Maintaining a close monitoring of these areas is important now that they are to be closed to all fishing during the spawning aggregation period. The closed season and protected areas will provied an opportunity for the population to recover from the heavy fishing pressure, as well as provide a better opportunity to understand the movement behavior during the spawning aggregation period of the red hinds through other research methods, such as tag and release. A study of this type will give the opportunity to detect the ability of the species to identify spawning sites (homing), and to determine the distribution of the species. Luckhurst (personal communication) has shown that red hinds have the ability of homing and identify particular spawning sites. Mark-recapture experiments provide important information on movement behavior during the reproductive period.

The objectives of the project were to characterize the movement behavior during reproductive period of red hinds and coneys at three spawning aggregations sites thru mark and release methods during the spawning periods of January, February and/or March 1999 and to collect a minimum of thirty gonads samples per species (red hinds and coneys) to assess spawning activity.

METHODS AND MATERIALS

Sampling was carried out using fish hooks (size #06) with squid as bait. Specimens were marked and released at three aggregation sites. Some details of sampling were subject to minor modifications that depended on logistics and prevailing conditions of weather and boats. For instance, extreme weather conditions forced the decision to include a sampling area known as Macamba or station 7, located southwest (15 km), of the intended areas. Logistics to improve the rate of survival required that fishing took place in waters less than 30 m.

Efforts were directed to tag approximately 500 individuals of each species. Specimens were first measured (in millimeters), and "vented" if necessary. (In this process the air in the gas bladder that had expanded when the specimen is brought to the surface is released by inserting a 18-gauge needle through the body wall under the pectoral fin until it penetrated the gas bladder. Gentle pressure is applied to the abdominal area that forced the air out of the bladder.) All fishes were doubled tagged with a numbered Floy anchor tag (FD-94) inserted on each side into the musculature at an angle of about 45°, 1 - 2 cm below the origin of the second dorsal fin. Ideally, the tag is anchored behind the second dorsal fin rays supports (pterygiophores). The specimens then were placed in standard fishers' ice chest filled with sea water and allowed to recover before released. The water in the container was pumped every 10 minutes to ensure that the specimens were in fully oxygenated water. Red hinds and coneys were released at sites individually at a distance about 5 km from the collection site. Coordinates of capture and release sites were taken with a GPS (global positioning system) and these sites were plotted on a chart of the area. The tagging methodology has been successfully used with red hinds (*Epinephelus guttatus*) in Bermuda (Luckhurst 1996, Luckhurst and Hateley MS).

A notice to the public (commercial and recreational fishers) was issued notifying of the intention on tagging and releasing red hinds and coneys at these sites. It was requested if any tagged specimen were captured outside the closed areas to return the fish and/or the tag to the Fisheries Research Laboratory.

RESULTS

A total of 579 coneys and 374 red hinds were marked and released in 52 trips between January 1999 and the 10 of June 1999. Although the intention was to tag fish in the closed areas off the West Coast of Puerto Rico, other areas were included due to logistics. Most individuals were sampled in two stations. The Abrir La Sierra spawning aggregation area was where most of the individuals were tagged and released. A total of 264 red hinds (70.6%) and 395 coneys (68.2%) were sampled in Abrir La Sierra. Macamba was the other area where the rest of the individuals were sampled. A total of 34 red hinds (9.1%) and 128 coneys (22.1%) were collected in Macamba. The approximated distance between these two areas is 18km.

Sampled coneys ranged from 120 to 296 mm TL, with an average of 210.5 mm ± 23.4. Individuals captured at Abrir La Sierra (station 59) ranged from 150 to 296 mm, with an average of 208.9 ± 30.0. Meanwhile, sampled coneys at Macamba (station 7) ranged in size from 158 to 270 mm, with an average of 212.8 mm ± 23.3 (Figure 1).

Not a single coney has been recaptured by our crew or by fishers. Efforts to mark and release coneys at a single location (Abrir La Sierra) to maximize the probability of recapture has not resulted in recaptures.

Red hinds sample ranged in size from 153 to 416 mm TL with a mean size of 278.7 mm ± 47.7. Red hinds sampled at Macamba ranged in size from 200 to 416 mm TL, with an average size of 278.7 mm ± 43.1. Those sampled at Abrir La Sierra ranged from 172 to 403 mm TL, with a mean size of 276.9 mm ± 47.0 (Figure 2).

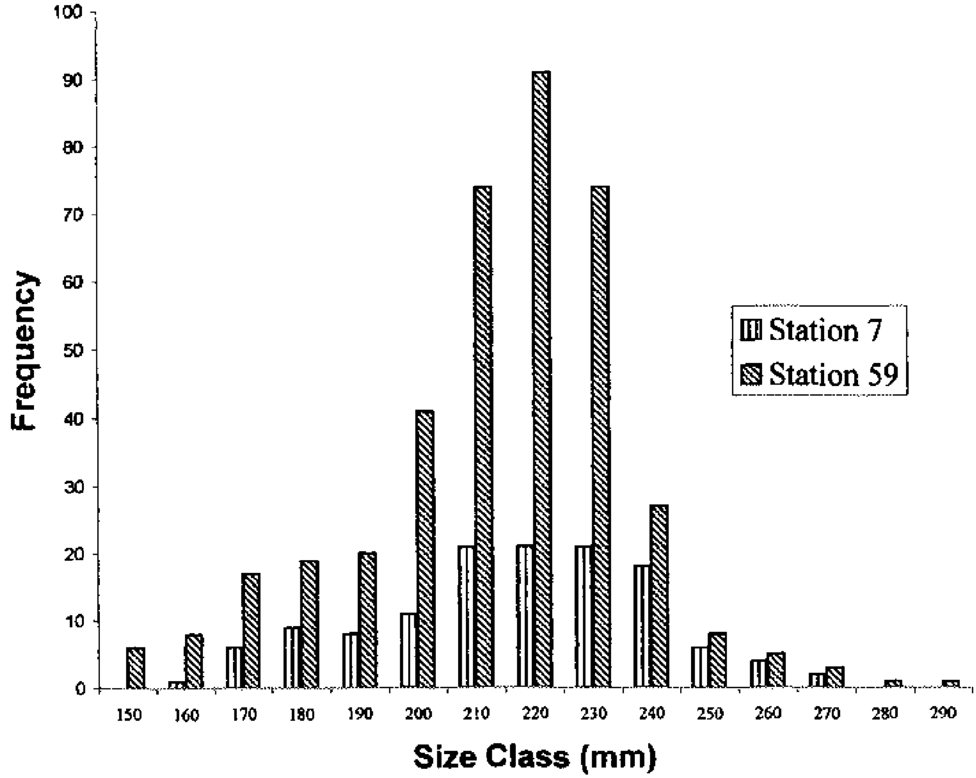


Figure 1. Size frequency distribution of sampled coneys (*Cephalopholis fulva*) at two red hinds spawning aggregation sites off the West Coast of Puerto Rico. Sampled sites are station 7 or Macamba and station 59 or Abrir La Sierra.

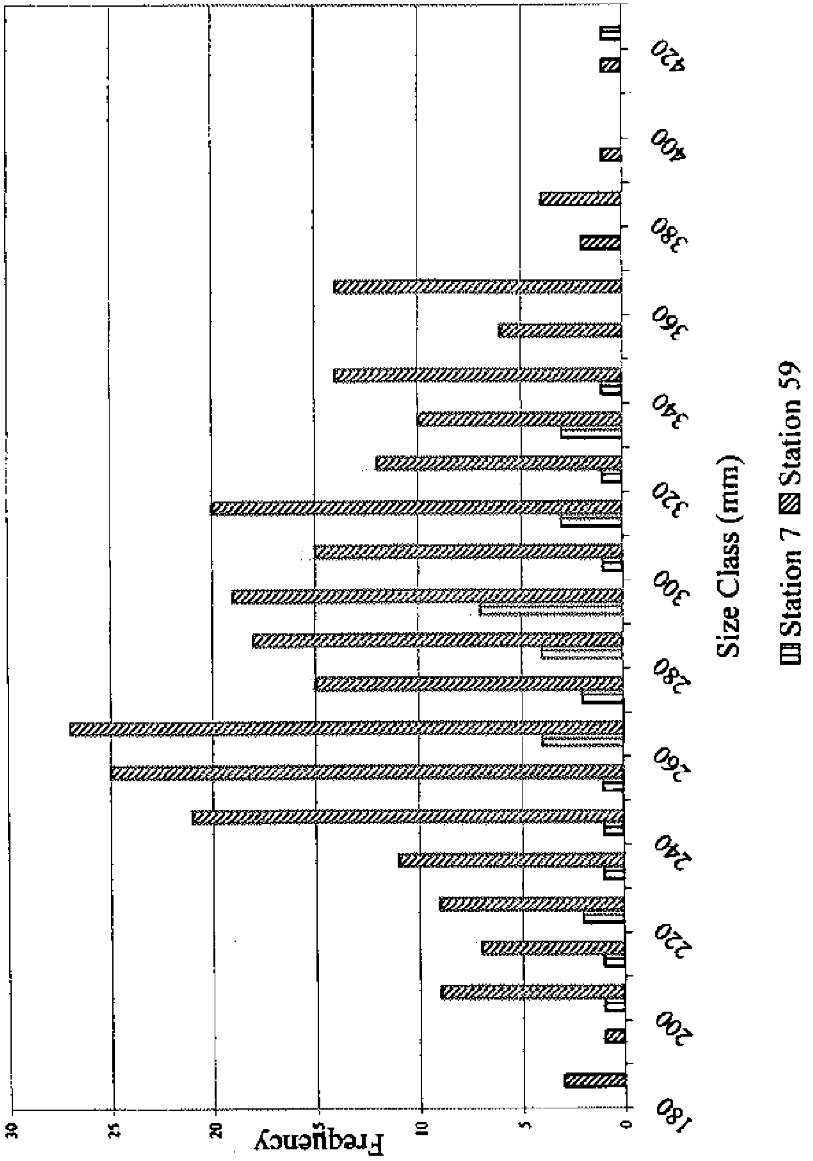


Figure 2. Size frequency distribution of sampled red hinds (*Epinephelus guttatus*) at two spawning aggregation sites off the West Coast of Puerto Rico. Sampled sites are station 7 or Macamba and station 59 or Abrir La Sierra.

A total of twenty-eight (28) red hinds accounted for thirty-five (35) recaptures between the period of February 8, 1999 and September 23, 1999. The number of days from the time of mark and release to recapture ranged from 2 to 191 days, with an average of 53.9 days. One individual was recaptured three times. The first time was 22 days between release and recapture; the second time was 28 days, and the third time 35 days between recaptures. Six individuals were recaptured two times. The number of days between recaptures of those red hinds ranged from 2 to 53 days. The size of the recaptured red hinds ranged from 204 to 337 mm TL, with a mean size of 267.0 ± 36 mm(Figure 3).

Twenty-seven of the red hinds were marked at Abrir La Sierra and released either at the same area of first capture or at approximately 5nm from the area of first capture. A single red hind marked at Macamba has been recaptured. The individual was released 5 km north of the area of first capture. All individuals returned to the area of first capture or very close to that area (Figure 4a). After first recapture the animals were set free at the same area of recapture. One red hind was recapture by a fisher, which was tagged at Abrir La Sierra, liberated 9.3 km northeast of Abrir La Sierra and recaptured 24.1 km south of the area in which was released (Figure 4b). The time span between the tag and recapture was 187 days. The exact area of recapture was reported by the fisher using a GPS.

DISCUSSION

In this study, red hinds were recaptured when they returned to the spawning area during the spawning season, suggesting a homing ability. These results are compatible with those obtained by Sadovy *et al.* (1992) in which they reported that red hinds do not necessarily move to the nearest spawning site. The red hind that was recaptured 24 km south of the tag and release area, suggested that this individual did not, move to the nearest area, Macamba. What we can not establish is if the animal after being released, returned to Abrir La Sierra, before moving south to the area where it was recaptured.

Few tagged fish have been recovered in this study, suggesting that post tagging mortality may be high. Efforts to tag and release red hinds will continue for the next year and half through a proposal to determine the demography of the species off the West Coast of Puerto Rico. We expect to expand our knowledge on the reproductive biology of this species, as well as gather additional information on the coney.

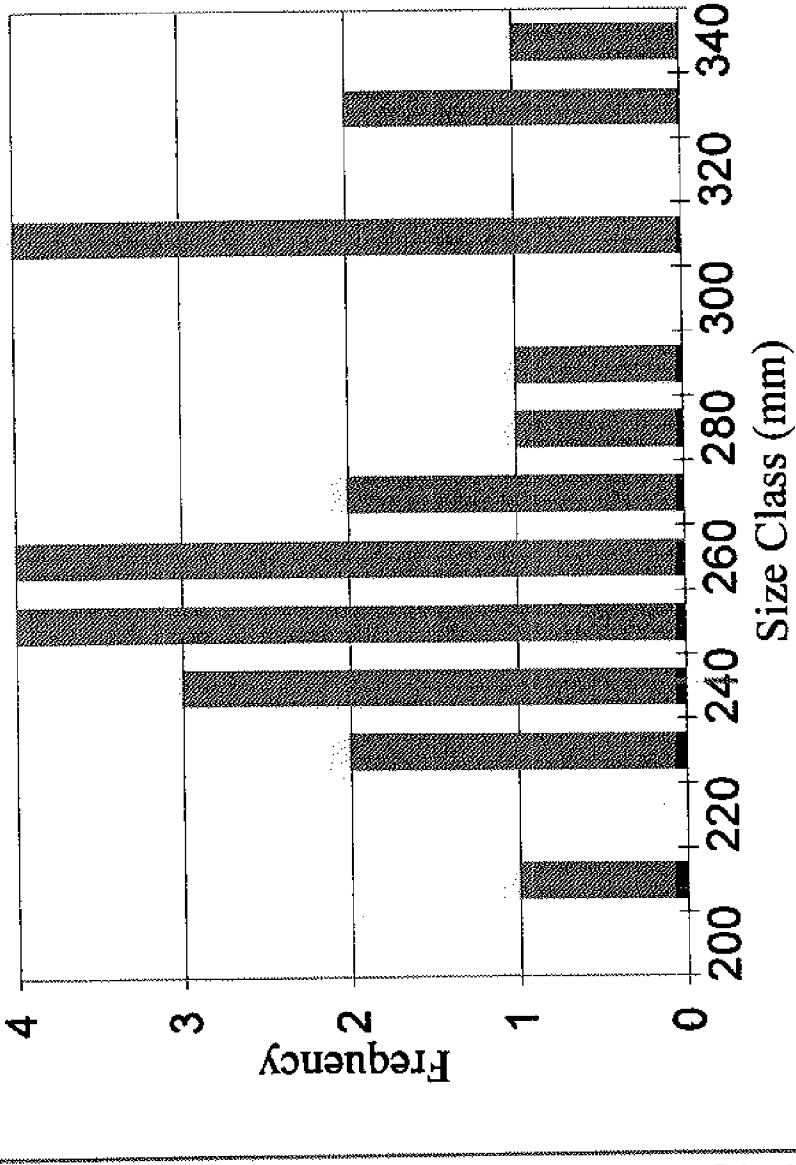


Figure 3. Size frequency distribution of recaptured red hinds (*Epinephelus guttatus*) at two spawning aggregation sites off the west coast of Puerto Rico. Sampled sites are station 7 (Macamba) and station 59 (Abrir La Sierra).

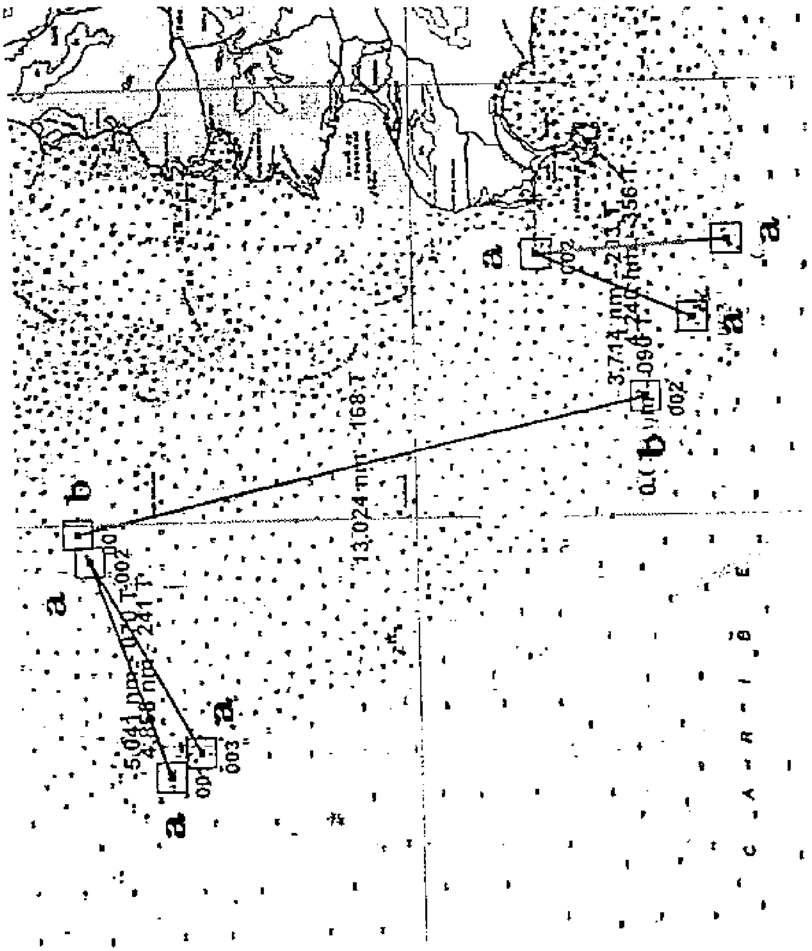


Figure 4. Map showing the two main capture-recapture areas visited during the study. A.) Areas where individuals returned to the same zone of first capture. B) Area where one individual was released after being tagged at Abrir La Sierra and released 9.3 km north, and recaptured 24 km south of the released area.

released 9.3 km north, and recaptured 24 km south of the released area.

ACKNOWLEDGMENT

We would like to express our gratitude to Ms. Graciela García-Moliner of the DoC/NOAA/Caribbean Fishery Management Council for drafting the figure on the nautical chart.

LITERATURE CITED

Beets, J. & A. Friedlander. 1992. Stock analysis and management strategies for red hind, *Epinephelus guttatus*, in the U.S. Virgin Islands. *Proc. Gulf Carib. Fish. Inst.* 42:66-80.

Burnett-Herkes, J. 1975. *Contribution to the biology of the red hind, Epinephelus guttatus, a commercially important serranid fish from the tropical western Atlantic.* Ph.D. Dissertation, University of Miami, Coral Gables. 154 pp.

Colin, P.L., D.Y. Shapiro, and D. Weiler. 1987. Preliminary investigations of reproduction of two species of groupers, *Epinephelus guttatus* and *E. sriatus* in the West Indies. *Bull. Mar. Sci.* **40**:220-230.

García-Moliner, G.E. 1986. Aspects of the social spacing, reproduction and sex reversal in the red hind, *Epinephelus guttatus*. M.Sc. Thesis, University of Puerto Rico, Mayagüez. 104 pp.

Juhl, R. 1969. Exploratory fishing surveys and gear test in Puerto Rico. *Contr. Agrop. Pesq. Fish Wildl. Div. Dept. Agric.*1 (1):1-39.

Juhl, R. 1972. A report on exploratory fishing and gear tests in Puerto Rico from 1969 to 1972. *Contr. Agrop. Pesq. Fish Wildl. Div. Dept. Agric.* **4**(3):1-63.

Juhl, R., and J.A. Suárez-Caabro. 1972. A report on exploratory fishing survey and gear test in Puerto Rico from 1967 to 1972. *Contr. Agrop. Pesq. Fish Wildl. Div. Dept. Agric.* **4**(3):1-63.

Juhl, R. and J.A. Suárez-Caabro. 1973. Fish pot fisheries in Puerto Rico. *Contr. Agrop. Pesq. Fish Wildl. Div. Dept. Agric.***5**(4):1-18.

Matos, D. 1997. Commercial fisheries statistics: Puerto Rico/State Federal Cooperative Statistics Program 1994 - 1996. Department of Natural and Environmental Resources. Final Rep. NMFS. 74 pp.

Luckhurst, B.E. 1996. Report on a program to tag and release tiger groupers *Mycteroperca tigris* (Pisces: Serranidae) captured from a spawning aggregation of this specie at Vieques, March 1996. Rep. Carib. Fish. Man. Coun. P.R. 1-11.

Luckhurst, B.E. and J. Hateley. Spawning aggregation dynamics, homing and growth of red hind, *Epinephelus guttatus* (Pisces:Serranidae) using a tag-recapture methodology at Bermuda. *Bull. Mar. Sci.* In Press.

Rosario, A. 1989. Fisheries-independent monitoring of commercially exploited reef fish and spiny lobster resources in Puerto Rico. Completion Rep. CODREMA R. Mayaguez, Puerto Rico. 1-114.

Rosario, A. 1992. Shallow-water reef fish monitoring. Completion Rep. Fisheries Research Laboratory/DNR, Mayaguez, Puerto Rico. 1-60.

Rosario, A. 1994. Caribbean/NMFS Cooperative SEAMAP Program Shallow-water reef fish monitoring. Progress Rep. NMFS. 1-73.

Rosario, A. 1996. Caribbean/NMFS Cooperative SEAMAP Program Shallow-water reef fish monitoring. Annual Rep. P.R. Dept. Nat. Env. Res. 1-133.

Sadovy, Y. 1994. Grouper stocks of the western central Atlantic: the need for management and management needs. *Proc. Gulf Carib. Fish. Inst.* **43**:43-63.

Sadovy, Y., and M. Figuerola. 1992. The status of the red hind fishery in Puerto Rico and St. Thomas, as determined by yield-per-recruit analysis. *Proc. Gulf Carib. Fish. Inst.* **42**:23-38.

Sadovy, Y., M. Figuerola and A. Román. 1992. Age and growth of red hind, *Epinephelus guttatus*, in Puerto Rico and St. Thomas. U.S. *Fish. Bull.* **90**:516-528.

Sadovy, Y., A. Rosario and A. Román. 1994. Reproduction in an aggregating grouper, the red hind, *Epinephelus guttatus*. *Env. Biol. Fish.* **41**:269-286.

Shapiro, D. Y., Y. Sadovy, and M. Angela McGehee. 1993. Size, composition, and spatial structure of the annual spawning aggregation of the red hind, *Epinephelus guttatus* (Pisces: Serranidae). *Copeia* **2**:399-406.

Species Composition, Abundance and Catch Rates of Fish Caught on the Formigas Bank, Jamaica

SANDRA GRANT
Ministry of Agriculture, Fisheries Division
P.O. Box 470, Marcus Garvey Drive,
Kingston, Jamaica

Fishers, who fish on the Formigas Bank, approached the Fisheries Division, regarding the status of the fishable resource on the Bank. Their main concern was the reduction in catch, and suggested that as a management strategy the Division should close the Bank for at least one year. The Division with assistance from CARICOM Fisheries Resource Assessment and Management program embarked on a data collection programme to gather information on species composition, abundance and catch rates by gear type. The results were compared with data from Alice Shoal, Morant Bank and Discovery Bay. There were significant difference in the catch rates and species composition and abundance by gear type. Although, the overall catch rate by gear type is lower that other areas, complete closure of the area might not be necessary.

KEY WORDS: Abundance, catch rates, species composition

INTRODUCTION

The Jamaica Fisheries Division (FD) began catch and effort data collection on Formigas Bank from January to October 1996, in response to a request from the fishermen. They felt the bank was over-exploited due to a reduction in catch rate of reef fish and suggested that as a management strategy the FD should close the bank for at least one year. The Division therefore saw it necessary to study the bank, first by looking at catch and effort data from the area.

The Formigas Bank

The bank is approximately 40 km NE from Manchioneal, Portland (landing site the data was collected) (Figure 1). The bank comprises mainly of three bottom types; rock (SE), seagrass (NW) and sand in the middle. Not much is known about the area.

METHODS

Approximately 130 fishermen fish on the bank, the main fishers are from Manchioneal (50% of the fishers). Fishermen travels, via motorized 8.4m fibreglass boats, four (4) hours to the bank and spend approximately six (6) hours fishing. They will return to the landing site by 2:00pm where they are meet twice per month by the data collectors from the Fisheries Division. Boat captains are questioned about the days activities, information includes, gear types, hours fishing, number of gears used, species weight and price (ex-vessel) prices are recorded.

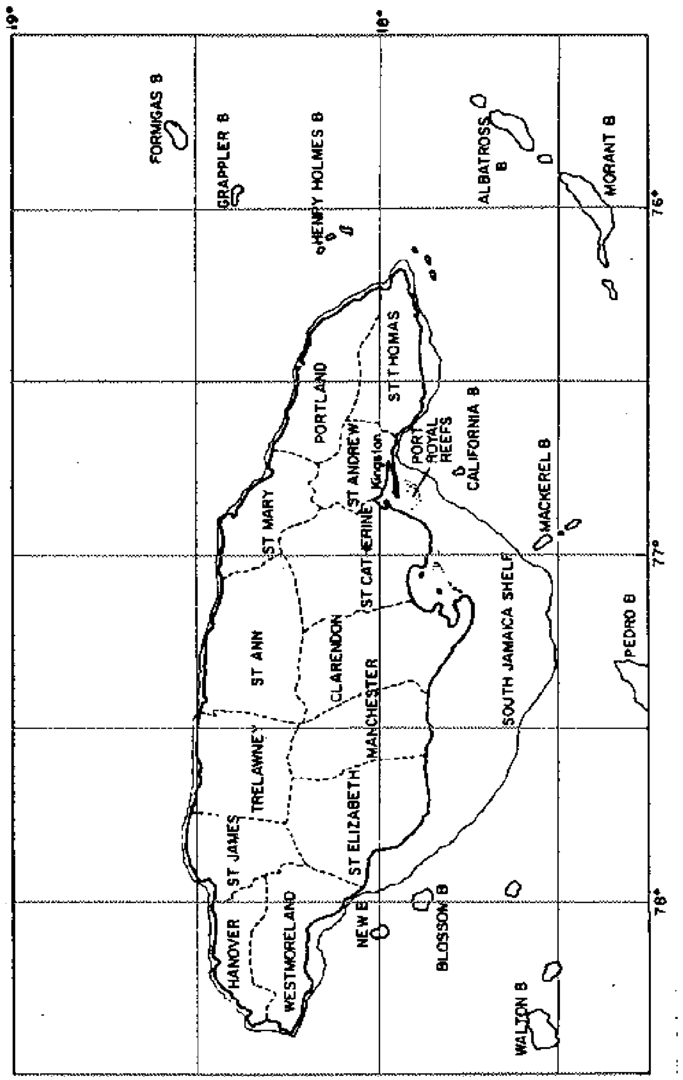


Figure 1. Map of Jamaica showing extent of shelf areas and position of proximal oceanic bank (Source Munro, 1983).

RESULTS AND DISCUSSION

Species Composition and Abundance

The main gear types used on the bank were fish pots (66%), 3.18cm (1.25") maximum diameter, speargun (28%), trolling (3%) and dropline (2%) (Appendix 1). Dropline (vertical line with 20 - 30 hooks) and trolling are not methods of choice on the bank, these methods requires fishers to use large quantities of gas. A total of 48 species (15 family groups) are landed from the bank. Pots contributed the highest number of species (41 species); speargun, 10 species; trolling, 5 species and dropline, 3 species (Appendix 1). The most dominant family group, in terms of weight, was the Parrotfishes (39%). The next most dominant family group are Squirrelfishes (19%), followed by Surgeonfish (11%), Groupers (10%) and Triggerfishes (6%) (Table 1). Lobster is the targetted species by spearfishers contibuting 86% of the total weight followed by Barracuda (6%) and Parrotfishes (3%). Barracuda (31%) were caught mainly by trolling, followed by Jacks and Dolphinfishes (25% each) and Mackerel (19%). Groupers (67%) and Snappers (33%) are the only families caught by dropline.

Species composition on the Formigas bank is relatively low compared to other areas (Table 2.)

In any reef fisheries, decreases in the abundance or biomass of piscivorous or carnivorous species have proven to be the most readily detectable effects of fishing pressure (Jennings and Polunin 1996). The abundance of herbivores (Sacridae and Acanthuridae) and carnivores (Serranidae) comprises the four most dominant family groups on the Formigas Bank, this suggests that the bank is over exploited.

Catch Rates

Fish Pot -- During this period of this activity, no pot fishing occurred in January, August and September, by Manchioneal fishers. January was due to poor weather conditions while in August fishers found a new bank approximately 15 miles from the landing site, thus August and September was spent fishing on the new bank. Of the remaining seven (7) months, the mean catch rate was 2.08 kg/trap/haul.

By species mean catch rate for Redband Parrotfish and Holocentrus ascencionis is the same 0.38 kg/trap/haul. The most dominant family group in terms of catch rates were Scaridae, followed by Holocentridae, Acanthuridae and Serranidae (Table 3).

Table 1. Ranking of family by gear

Rank	POT		SPEARGUN		TROLLING		DROPLINE	
	Family Name	%wt	Family Name	%wt	Family Name	%wt	Family Name	%wt
1	Scaridae	39	Palinuridae	86	Sphyraenidae	31	Serranidae	67
2	Holocentridae	19	Sphyraenidae	6	Carangidae	25	Lutjanidae	33
3	Acanthuridae	11	Scaridae	3	Coryphaenidae	25		
4	Serranidae	10	Lutjanidae	2	Scombridae	19		
5	Palinuridae	6	Haemulidae	1.3				
6	Balistidae	6	Balistidae	1.2				
7	Carangidae	3	Holocentridae	0.7				
8	Coryphaenidae	2						
9	Haemulidae	1.9						
10	Pomocanthidae	1.5						
11	Mullidae	0.7						

Table 3. Ranking of family group by catch rates

Rank	Pot		Spearfishing		Trolling		Dropline	
	Family Name	kg/th	Family Name	kg/dh	Family Name	kg/lh	Family Name	kg/lh
	OVERALL	2.08	OVERALL	4.34	OVERALL	1.72	OVERALL	1.57
1	Scaridae	0.70	Palinuridae	3.67	Scombridae	0.76	Serranidae	1.05
2	Holocentridae	0.32	Sphyraenidae	0.36	Sphyraenidae	0.33	Lutjanidae	0.52
3	Acanthuridae	0.17	Scaridae	0.10	Carangidae	0.09		
4	Serranidae	0.17	Lutjanidae	0.08	Coryphaenidae	0.07		
5	Palinuridae	0.13	Balistidae	0.07				
6	Balistidae	0.12	Haemulidae	0.04				
7	Carangidae	0.06	Holocentridae	0.02				
8	Coryphaenidae	0.04						
9	Haemulidae	0.04						
10	Pomocanthidae	0.03						

KEY: th – trap haul; dh – dive hour; lh – line hour.

Table 2. Number of species by gear.

Fishing Ground	# Species in Pot	# Species on Line	Source
Alice Shoal	77	23	Fisheries Division, 1986
Morant Bank	51	-	Pears, 1996
Formigas Bank	48	8	Present Study

Formigas bank is less exploited than Discovery Bay (north shelf), however, Morant Cay has a higher catch rate by family group (Table 4).

Table 4. Pot catch rate (kg/trap/haul) by family group.

Family	Formigas	Morant Cay	Discovery Bay
Scaridae	0.70	0.96	0.36
Holocentridae	0.32	0.37	0.08
Acanturidae	0.17	0.34	0.11
Serranidae	0.17	0.21	0.06

The catch rates are low when compared with other areas (Table 5), although very similar to Morant Cay.

Table 5. Pot catch rate comparison

Catch rates (kg/trap/haul)	Fishing Ground	Soak (days)	Source
21.00	Jamaica (offshore)	2	Munro, 1983
8.17	Alice Shoal	2	Fisheries Division, 1996
2.96	Morant Cay	3.2	Pears, 1996
2.60	Jamaica (inshore)	16	Munro, 1983
2.08	Formigas Bank	3	Present Study

Spearfishing

Mean catch rate is 4.34 kg/dive hours, this is higher than pots. Lobster has the highest catch rate 3.67 kg/dive hours. (Table 3).

Trolling

The main species caught are King Mackerel (0.63 kg/line hrs.), Wahoo (0.38 kg/line hrs.) and Barracuda (0.43 kg/line hrs.), the mean catch is 1.72 kg/line hrs. Catch rate on Formigas is similar to Alice Shoal (Table 6).

Table 6. Line catch rate comparison

Catch Rate (kg/line/hour)	Fishing Ground	Source
2.30	Morant Cay (SE)	Kawaguahi, 1974
2.00	Pedro Bank	Munro, 1983
1.72	Formigas Bank	Present Study
1.59	Alice Shoal	Fisheries Division, 1996

Dropline

Mean catch rate is 1.57 kg/line hours or 0.036 kg/hook hours.

CONCLUSION

Although the data suggests that Formigas Bank is as exploited as Morant Cay (offshore) and Discovery Bay (inshore), closing this bank will also justify closing the other areas as well. Although, fishers from Manchioneal area would like the Bank to be closed, experience tells us that, economic pressures will force fishers to violate this regulation. The alternative would be for the Fisheries Division to properly manage the area, by using specific management strategies in association with assistance from fishers who uses the area. Management options could include:

i) Increase mesh size of pots, from 3.18cm to 3.82cm..
ii) Investigating a new gear technology ie. Use of long-line fishing .
iii) Control of the number of fishermen on the bank.
iv) Monitor the bank for poachers.
v) Co-management

Administration of the above management options could best be achieved by co-management. With the efforts of the fishers, stakeholders, Fisheries Division and a NGO, monitoring activities on the Bank could be accomplished.

ACKNOWLEDGMENTS

To Mr. Karl Aiken for editing the manuscript. To Mr. Peter Espeut for providing insight into the benefits of co-management as a way to achieve fisheries management. And to the staff of the Data Collection and Assessment Unit of the Jamaica Fisheries Division who collected the data.

LITERATURE CITED

Hartsuijker, L. 1982. Trapfishing survey of Pedro Bank (Jamaica) 2nd Phase. Technical Report No. 4. A reassessment of the Stocks of reef fish on Pedro Bank.

Kawaguchi, K. 1974. Handline and longline fishing explorations for snapper and related species in the Caribbean and adjacent waters. *Mar. Fish. Rev.* **36**(9):8-31.

Munro, J.L. 1983. *Caribbean Coral Reef Fishery Resources.* 2nd ed.

ICLARM Studies and Reviews 7. International Center for Living Aquatic Resources Management, Manila, Phillippines. 276 pp.

Pears, R.J. 1996. Conservation and the reef fishery of the Morant Cays, Jamaica. M.Sc. Thesis. University College London 95 pp.

Preliminary Results from a Continuing Study of Spawning and Fecundity in the Red Snapper (Lutjanidae: *Lutjanus campechanus*) from the Gulf of Mexico, 1998-1999

L. ALAN COLLINS, GARY R. FITZHUGH, LAETITIA MOURAND, LINDA A. LOMBARDI, WILLIAM T. WALLING, JR., WILLIAM A. FABLE, JR., MEREDITH R. BURNETT, and ROBERT J. ALLMAN
National Marine Fisheries Service
Panama City Laboratory
3500 Delwood Beach Road
Panama City, Florida, 32408 USA

ABSTRACT

In response to increased management concerns by the National Marine Fisheries Service in early 1998, we analyzed red snapper (Lutjanidae: *Lutjanus campechanus*) gonads and otoliths from the U. S. Gulf of Mexico coasts of Texas, Louisiana, Mississippi, Alabama and west Florida (TX, LA, MS, AL and FL, respectively). Our main objective related to reproduction was to provide size and age-specific estimates of fecundity (batch fecundity and frequency of spawning) for large red snapper. We previously provided estimates of fecundity (n=66, from 349 to 820 mm total length, TL) from northeastern Gulf of Mexico red snapper. However, in order to complete size/age matrices of spawning frequency, more samples of larger females were needed. We therefore greatly expanded our sampling area and number of samplers. Most samples were provided by headboat samplers stationed from South Padre Island, TX, to St. Petersburg, FL, beginning in late June 1998 and ending in October 1999. Large red snapper (at least 500 mm TL) were non-randomly selected in an attempt to provide adequate sample numbers of large fish. The sex ratio of all 1517 fish sampled was 1:1. The spawning season off TX and LA began and ended in the same months as in the northeastern Gulf: April - May and September - October, respectively. The presence of hydrated oocytes in ovaries of some large females indicated that spawning began in April 1999 off TX and LA. A few females had hydrated oocytes in October 1998 to signal the end of the spawning season in all areas. Batch fecundity for 1998 (n = 59, from 359 to 901 mm TL) ranged from an extremely low value of 13 (in a 417 mm TL, 4 year-old TX fish) to an extremely high value of 3.4 million (in a 851 mm TL, 11 year-old LA fish). Both age and TL had an exponential relationship with batch fecundity, but age was the best predictor. Spawning frequency estimates by age (for those ages or age ranges with at least 36 females sampled in 1998) were about 50% greater for age 6-35 females than for ages 3, 4 or 5 females. Several spawning locations around the northern Gulf were also identified using headboat and fishery independent data along with histology.

KEY WORDS: Fecundity, snapper, spawning

INTRODUCTION

In early 1998, the National Marine Fisheries Service (NMFS) expressed increased concern with the management of red snapper commercial and recreational fisheries in the Gulf of Mexico. Commercial catches had peaked in 1983 and then steadily declined through 1989 (Bennett [1998]). Commercial quotas have been used since 1990 to close the fishery early (= before the end of the fishing year) every year. While red snapper commercial catches since 1983 have remained stable or increased from the northwestern and north-central Gulf (Texas, Louisiana, Mississippi and Alabama), catches from the west coast of Florida have steadily decreased since 1983 to an all-time low (a total decrease of two orders of magnitude) in 1997 (Bennett 1998). Some regional changes in landings may have been affected by the commercial quota. Recreational fishery landings in the Gulf have been somewhat more stable, but quotas also closed that fishery early in recent years and increasing minimum size limits have also affected the catch (Schirripa and Legault 1999).

Research on reproductive biology of reef fishes is important to assessing stocks, testing management tools, and evaluating habitat. Spawning potential ratios (SPRs) require age-specific fecundity estimates to determine if stocks are overfished. Reproduction studies also help to gauge the success of marine reserves as a management tool. Reef fishes are usually aggregate spawners and little is presently known about the structure and function of those aggregations. The identification of spawning sites also helps to delineate essential fish habitat.

Extensive sampling and study of both age/growth and reproduction from the north-central and northwestern Gulf was requested by NMFS to improve stock assessment of this species. We had previously studied red snapper reproduction and published our results on histology and fecundity estimates from the northeastern Gulf of Mexico (Collins, et al. 1996). Samples from the north-central and northwestern Gulf of Mexico were more difficult to obtain and required the assistance of all Gulf samplers from the NMFS Beaufort, NC, Headboat Survey Program.

Our objectives were threefold: (1) to acquire greater numbers of large female red snapper that could be used for estimating fecundity; (2) to determine if spawning times for this species were similar in the eastern and western Gulf of Mexico; (3) to identify red snapper spawning sites using histology and catch location data from fishers.

METHODS

Our methods were identical to those in Collins et al. (1996), except that field sampling was much more extensive and we specifically selected larger (>499 mm total length, TL) fish in the present study. Fish <500 mm TL were randomly sampled. Red snapper gonads and otoliths were sampled mainly from recreational headboats out of west Florida (FL), Alabama (AL), Louisiana (LA), and Texas (TX). Charterboats from Panama City, FL, and tournaments in FL and Mississippi (MS) were sampled to a lesser extent. A few fishery-independent samples came from NMFS scientific surveys off Panama City, FL, and MS.

For each fish sampled in the field, fork length (FL) and TL were first measured to the nearest mm, and total wet weight was usually taken to the nearest 0.01 kg. Gonads were then removed, placed dry in heavy-plastic bags and kept on ice until processed. A sagittal otolith was also removed from each fish. Samplers in TX, LA, and AL shipped otoliths and gonads on ice to our lab in Panama City by overnight mail.

In the laboratory, gonads were processed as soon as possible (usually within four hours of shipment-arrival). Excess tissue was removed and a small sample of each gonad was examined at 250x to determine final sex and preliminary stage of gonad maturation (1-resting; 2-early developing; 3-late developing; 4-ready to spawn or spawning ; 5-recently spawned or spawned-out; (West 1990). The diameter of the largest oocyte (MAXOD) found in the small sample was recorded for each female. All gonads were weighed to the nearest 0.1 g before most samples were placed in 10% buffered formaldehyde solution (mixed according to Hunter 1985) inside a sealed plastic bag.

A gonadosomatic index (GSI = 100 * gonad weight/ total weight), the MAXOD, and preliminary gonad stages were used to generally delineate the spawning season, as well as to compare to the final staging from histology. After at least two weeks in 10% formaldehyde solution, tissue samples were used for standard histological slides (Fitzhugh, et al. 1993). Slides were then viewed at our laboratory in order to record sex, stage and quality of preservation. Histological-stages for females were determined by the most-advanced stage of oocyte development found in each fish: 1-primary growth; 2-cortical alveolar; 3-vitellogenic; 4-early coalescing or nucleus migrating ; 5-fully hydrated; 6-spawning or recently spawned (with fresh post-ovulatory follicles, POFs); 7-at least 50% atretic. Histological-stages for males were: 1-inactive; 2-active, with many secondary spermatocytes; 3-developing ,with some spermatids in ducts; 4-ripe, with large pools of spermatozoa in ducts.

Batch fecundity and spawning frequency were estimated using the hydrated oocyte method of Hunter, et al. (1985) and Hunter and Macewicz (1985), respectively. Only histological stage 4 and 5 females were used for batch fecundity estimates. We used the length of the smallest hydrated female as a benchmark for selecting fish included in the spawning frequency estimate: histological stage 1,2, 3 and 7 females were counted as not spawning while stage 4, 5 and 6 females were counted as spawning. Batch fecundity was regressed on TL and age using linear and non-linear models.

Age was determined from sections of sagittal otoliths following the methods of Fitzhugh, et al. (1999). Nelson (1980) validated rings on red snapper otoliths as annular marks.

Spawning sites were identified as those locations where at least one female with hydrated oocytes or fresh POFs was found. Locating these sites required the catch-coordinates from fishers and a histological slide from each fish.

RESULTS

A total of 1,517 red snapper have been sampled between Port Isabel, TX, and St. Petersburg, FL, during the period February 13, 1998, through October 16, 1999. Sex ratio was 1:1 by chi-square analysis (Zar 1984) and 51.6% of all fish collected were females. Headboat samples outnumbered those from all other modes with n = 954 (62.9%) and charterboat samples were the next most-dominant mode with n = 340 (22.4%). Most samples (42.8%) came from FL, followed by TX (31.6%), LA (12.6%), AL (11.7%) and MS (1.3%).

As expected (see Figure 17 in Schirripa and Legault 1999), most large specimens (>499 mm TL; n = 517) came from the northwestern/north-central Gulf of Mexico. These TX and LA fish made up 56.1 % of this high-priority size-group for fecundity estimates, with Florida fish making up 26.9%. Texas and LA also produced 74.2 % of the largest fish (700 to 972 mm TL; n = 120). Sex ratio of these largest fish was about the same as for the total sampled (1:1 by chi-square analysis (Zar 1984), with 53.3% female.

The spawning season for red snapper from FL to TX in this study was April or May through September or October, according to GSI (Figure 1), MAXOD, preliminary gonad-staging and histological gonad-staging. Our sampling of TX, LA, MS and AL in 1998 did not start until early July, so the onset of spawning in those states is based on 1999 sampling only. The end of spawning in those same states to the west of FL also was shown by one year's data (1998) because headboat sampling ceased in late August 1999 due to the fishery being closed in all federal waters.

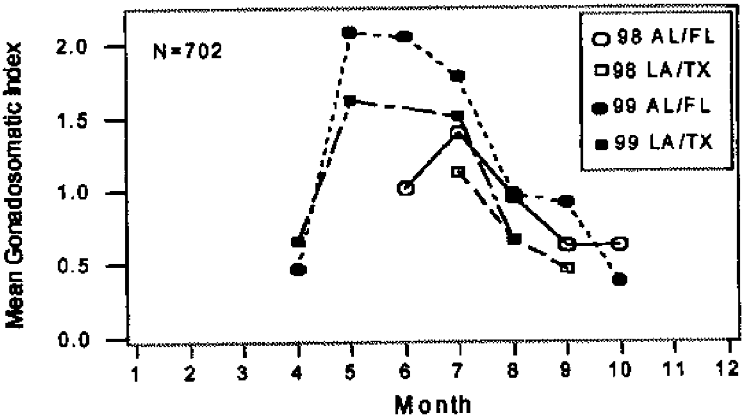


Figure 1. Mean GSI of female red snapper collected by all gear types from northeastern (FL/AL) and north-central/northwestern (LA/TX) Gulf of Mexico, 1998-1999. Only months with n > 10 are shown.

Some locations of spawning activity off TX, LA, AL and FL were determined by the use of coordinates provided to the Headboat Survey Program by headboat-captains (pers. comm., R. Dixon, NMFS Beaufort, NC July 1999) and also from NMFS fishery independent sampling (Figure 2). Headboat catch locations were 10' by 10' grids where at least one day's catch (mostly during 1998) included at least one female red snapper with hydrated oocytes. Catch coordinates from most 1998-1999 headboat trips were not available. Fishery independent sampling that revealed the location of females with hydrated oocytes off AL and FL was conducted on a "R/V Ferrell" longline cruise and on several day-trips by NMFS Panama City personnel. Scientists on board those vessels (Pers. comm., D. DeVries, A. David, NMFS Panama City, October 1999) provided data on exact catch-location, catch depth, total red snapper caught and bottom temperature (Table 1). Headboat data on 1999 red snapper spawning sites off west-central FL are shown in Table 2.

Batch fecundity was estimated for all females (n = 59) that contained intact ovaries with hydrated oocytes and no fresh POFs. Dates of catch on these fish ranged from late June to mid-September. All Gulf states except MS were represented in these subsamples. Total length and total weight ranged from 359 to 901 mm and 0.77 to 8.08 kg, respectively. Due to the small sample size by state, all fish on which batch fecundity was estimated were combined for analysis.

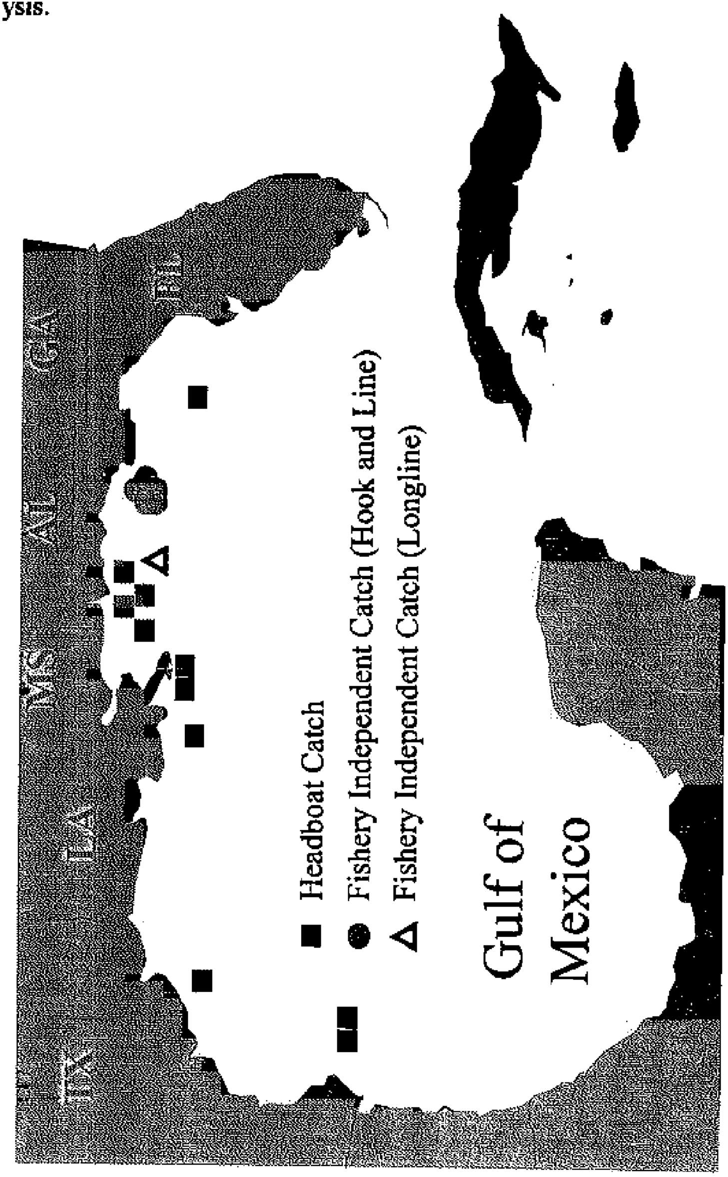


Figure 2. Red snapper spawning locations, 1998-1999.

Table 1. Fishery independent data on 1999 red snapper spawning sites off Alabama (AL) and northwest Florida (FL).

State	Date	Degrees Latitude (N)	Degrees Longitude (W)	Depth (m)	Bottom Temperature (°C)
AL	5/18	29.557	87.460	122.0	20.6
FL	7/15	29.733	86.127	47.2	27.0
FL	7/28	29.816	65.914	38.4	26.8
FL	7/28	29.882	85.832	32.9	26.5
FL	9.29	29.618	86.052	61.0	29.0
FL	10/13	29.825	85.998	40.2	29.0

Table 2. Headboat data on 1999 red snapper spawning sites shown by occurrence of females in the Florida Middle Ground. Approximate location = 28°N latitude and 84°W longitude)

Date	Approx. Depth (m)	N	TL range (mm)	Max. oocyte Dia. (mm)	Histological stage of ovary
5/21	23.8	6	420-490	0.52	Late vitellogenic
8/5	not given	6	440-601	1.94	Full hydrated

Batch fecundity estimates ranged widely from nearly zero (in a 4 year old fish) to more than 3 million (in an 11 year old fish) (Figure 3). Most fish sampled, aged 8 years or less, showed batch fecundities of less than 1 million. Some fish, aged 4 years or less, showed extremely low batch fecundities of less than 200.

Both age and TL had an exponential (third order polynomial) relationship with batch fecundity, but age was the best predictor (Figure 3). Older individuals are potentially much more highly fecund. The TL relationship was: batch fecundity = $(8.0*10^{-9}TL^3) - (8.0*10^{-6}TL^2) + (2.2*10^{-3}TL)$; ($r^2 = 0.34$, $n = 59$).

Estimates of spawning frequency are incomplete, but age 6-35 females seemed to spawn about 50% more times than age 3,4 or 5 females did in 1998. Some spawning frequency estimates were made difficult by postmortem decay in some samples. The decay caused a problem separating some stage 3 and stage 4 ovaries. However, all ovaries can still be staged.

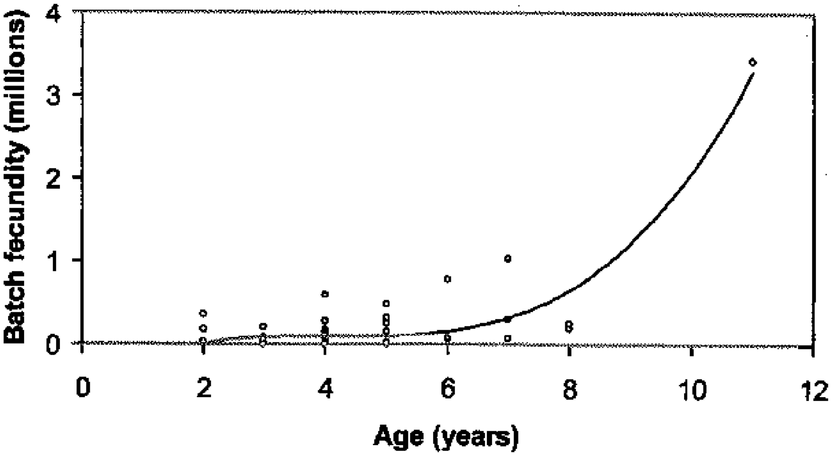


Figure 3. 1999 NW Florida red snapper spawning locations.

DISCUSSION

Our results on spawning months, locations and temperatures (assuming that spawning takes place near the bottom) and batch fecundity estimates generally agree with published studies. We recorded spawning from April - May through September - October and similar results were reported for the northeastern Gulf (Moe 1963, Futch and Bruger 1976, Collins, et al. 1996), the northwestern Gulf (Moseley 1965, and Bradley and Bryan 1976) and in the southern Gulf (Camber 1955). Collins, et al. (1980) also reported larval red snapper off Texas during May, July through September, and November. Bottom temperatures recorded at our spawning sites were 20.6 and 26.5 to 29.0°C. (Table 1); similar temperatures for spawning in captivity were reported by three studies (23 to 25°C. by Arnold, et al. 1978; 25 to 27°C. by Minton, et al. 1983, and 24 to 27°C. by Chesney and Filippo 1994). Our batch fecundity range of 13 (for a 417 mm TL, 4 year old fish) to 3.4 million (for a 851 mm TL, 11 year old fish) compares favorably to those in Collins, et al. (1996) (their range of batch fecundity was 458, for a 349 mm TL fish that couldn't be aged, to 1.7 million for a 820 mm TL fish that was 12 years old).

Although we increased the number of large red snapper on which batch fecundity has been estimated by sampling a much greater area of the Gulf of Mexico in 1998, many of the larger specimens we sampled could not be used for batch fecundity estimation because they were either males or non-hydrated females. We were, however, able to find almost as many fish (of all lengths) on which batch fecundity was estimated in our first year of this new sampling program (n = 59) as we had previously (off northwest FL) in 3 years (n = 66, in Collins, et al. 1996).

Although we requested larger fish from the port samplers, older-aged fish (i.e. > age 5) were rare in accordance with findings from a complementary aging study (Fitzhugh, et al. 1999). With the inclusion of some fish aged 6-8 and a single 11 year old into the reproductive analysis, it is apparent that batch fecundity increases exponentially with age and that some younger fish (e.g., < age 4 or 5) are showing extremely low batch fecundities (i. e., thousands or less). Batch fecundities typically range from tens of thousands to millions in some other demersal commercial-sized fishes (Fitzhugh, et al. 1993, Nieland and Wilson 1993, Wilson and Nieland 1994). The documentation of extremely low batch fecundities and good estimates of their proportions among age classes within a given spawning year would be of great interest in estimation of spawning potential and as a factor in monitoring recruitment variability.

We are not aware that extremely low batch fecundities are often reported in the fisheries literature. Loss of ova upon sampling and improper preservation of gonads have been suspected causes for biased estimation of fecundity; these are not uncommon problems among fishery-dependent dockside sampling programs.

Hydration of ova occurs within a diel cycle and spawning likely occurs after dusk (Chesney and Filippo 1994). Sampling partially hydrated females during daylight hours may also result in estimates of batch fecundity that are biased (low).

Extremely variable (i.e. low) batch fecundity data are also likely to be closely scrutinized, and eliminated as biologically unreasonable outliers when a goal is to provide "good" equation fits to fecundity data (see Schirripa and Legault 1999). However, we also noticed that these extremely low batch fecundities were detected from ovaries that were small in size, light in weight, and low in corresponding GSI value (hydrated ovaries < 20 g and with GSI < 0.7). This could not be explained by artifacts of sampling. Improper preservation and loss of running-ripe ova are likely to explain decreases of ovary weight (and batch fecundity) of only a few percent. Red snapper are known to mature very early (age 2) given their estimated longevity (about age 50) (Futch and Bruger 1976, Wilson et al. 1994). Relatively low ovarian weight occurring naturally among some females classified as mature seems to coincide with extremely low batch fecundity.

An examination of red snapper induced to spawn by hormone injection also revealed extremely low batch fecundities (hundreds to a few thousand) among the smallest fish (presumably age-2, based on expected size at age) in contrast to larger and presumably older females (Chesney and San Filippo 1994). Lacking more detailed reproductive information for a stock assessment, Schirripa and Legault (1999) looked at the Chesney and Filippo report and postulated a case where the fecundity-length relationship could be a two-tiered function with a steep initial slope for the first maturing females. Our initial findings based on field samples highlights the relevancy of Chesney and Filippo's observations of extremely low batch fecundity and provides support for the two-tier case presented in Schirripa and Legault (1999).

Although red snapper is one of the most studied fishes from the Gulf of Mexico, there are very few positive identifications of spawning sites and depths. Moe (1963) published habitat descriptions and fishermen's observations of red snapper spawning sites (determined from running ripe fish) at 13 - 16 fathoms (24 - 29 m) due south of Panama City, FL. Subsequently, research publications have reviewed anecdotal information, distribution of larvae and juveniles, and provided suppositions about larval transport pathways to advance what was known about possible spawning locations and depths (Moseley 1965, Beaumariage and Bullock 1976, Futch and Bruger 1976, Collins, et al. 1980). Bradley and Bryan (1975) report the only other fishermens' observation which increased the known depth of spawning (off TX) to about 20 fathoms (37 m). Futch and Bruger (1976) and Bradley and Bryan (1976) conducted surveys to specifically identify spawning locations and failed to find actively spawning red snapper. Collins et al. (1980) gave catch locations of 225 larvae and early juveniles, suggesting a general spawning area and depth range off TX. Our limited results to date indicate that spawning may be occurring at depths greater than previously reported. We have found actively spawning fish (occurrence of at least one hydrated female within a catch) from 24 to 60 m using hook and line gear. One hydrated female (out of 7 total red snapper) was collected during a fishery-independent long-line survey at 122 m depth. A hook-and-line fishery independent survey of the Texas Flower Gardens (Nelson, 1988) is also notable for the deeper depth distribution of "ripe" red snapper females. By interpolation of results in Nelson (1988) it can be estimated that about 10 ripe females captured from a total of 53 females during summer cruises were likely to have come from depths deeper than 50 m (99% of the red snapper captured). These findings, that red snapper spawning is not commonly detected and can occur across a broad depth zone (mid-shelf to slope), clearly reflect that we don't know enough about conditions and habitats important for red snapper reproduction.

Of special interest are some spawning sites off west-central FL shown by samples from headboats out of St. Petersburg (Figure 3, Table 2 and West 1999). Although red snapper in this area were once worthy of "a good catch" (Camber 1955) and "frequent" and "numerically dominant" (Smith, et al. 1975), catches in the 1980s and 1990s have been infrequent until the last two years (Schirripa and Legault 1999, West 1999). Red snapper may be moving back into that area from further west. Although Fable (1980) found that red snapper tagged off TX did not move far, Patterson (1999) found that this species does travel substantial distances off AL.

ACKNOWLEDGEMENTS

We especially thank the NMFS Beaufort, NC, Headboat Survey Team led by Charles Manooch and Robert Dixon. The following members of the Headboat Survey Team also deserve special mention: David Bartee, Scott Deitche (also of Florida Marine Research Institute), Amy Mach, Kit Doncaster, Jill Dzuryachko, Karen Kuebel, John Dameier, Jeff Marx and Damian Schreiber. Rene e Roman and Greg Fairclough of NMFS St. Petersburg, F L, also provided samples for this study. We also thank Ms. Cheryl Crowder and her Histology Laboratory Team at the Louisiana State University School of Veterinary Medicine for providing us with thousands of excellent histological slides for the last eight years. In addition, we appreciate the recent assistance of the following biologists/histologists in examining our images of red snapper ovaries and promptly relating their opinions back to us: Harry Grier, Lewis Bullock and Lisa Foushee (Florida Marine Research Institute); David Wyanski and Oleg Pashuk (South Carolina Marine Research Institute, Charleston); and Beverly Macewicz (NMFS, La Jolla Laboratory, CA).

LITERATURE CITED

Arnold, C.R., J.M. Wakeman, T.D. Williams and G.D. Treece. 1978. Spawning of red snapper (*Lutjanus campechanus*) in captivity. *Aquaculture* **15**:301-302.

Beaumariage, D.S., and L.H. Bullock. 1976. Biological research on snappers and groupers as related to fishery management requirements. Pages 86-94 in: H. R. Bullis, Jr., and A.C. Jones (eds.). *Proceedings: colloquium on snapper-grouper fishery resources of the western central Atlantic Ocean.* Florida Sea Grant Colloquium Report 17, 333 pp.

Bennett, J.E. [1998] Automated Landings Assessment for Responsive Management Report. National Marine Fisheries Service, Miami, FL, Sustainable Fisheries Division Contribution: SFD-98/99-39. 93 pp. Unpubl. Report.

Bradley, E., and C.E. Bryan. 1975. Life history and fishery of the red snapper (*Lutjanus campechanus*) in the northwestern Gulf of Mexico: 1970-1974. *Proc. Gulf Carib. Fish. Inst.* **27**:77-106.

Camber, C.I. 1955. A survey of the red snapper fishery of the Gulf of Mexico, with special reference to the Campeche Banks. Fl. Bd. Conserv., Tech. Ser. 12. 63 pp.

Chesney, E.J. and R. San Filippo. 1994. Size-dependent spawning and egg-quality of red snapper. Final report to U. S. Dept. Comm.,NMFS. Marine Fisheries Initiative (MARFIN) Coop. Agreement NA37FF0048-01. 17 pp.

Collins, L.A., J.H. Finucane and L.E. Barger.1980. Description of larval and juvenile red snapper, *Lutjanus campechanus*. *Fish. Bull.* **77**(**4**):965-974.

Collins, L.A., A.G. Johnson and C.P. Keim. 1996. Spawning and annual fecundity of the red snapper (*Lutjanus campechanus)* from the Northeastern Gulf of Mexico. Pages 174–188 in: F. Arreguin -Sanchez, J. L. Munro, M. C. Balgos and D. Pauly, (eds.) *Biology, fisheries and culture of tropical groupers and snappers.* Proc. ICLARM/EPOMEX International Workshop, Campeche, Mexico. October 1993.

Fable, W. A., Jr. 1980. Tagging studies of red snapper (*Lutjanus campechanus)* and vermilion snapper (*Rhomboplites aurorubens)* off the south Texas coast. *Contributions to Marine Science* **23**:115-121.

Fitzhugh, G.R., B.A. Thompson, and T.G. Snider III. 1993. Ovarian development, fecundity and spawning frequency of black drum *Pogonias cromis* in Louisiana. *Fish. Bull.* **91**:244-253.

Fitzhugh, G.R., L. A. Collins, L.A. Lombardi, M. R. Burnett, W.A. Fable and W.T. Walling [1999]. Report of red snapper aging and spawning/fecundity research. National Marine Fisheries Service, Panama City, Florida. 5 pp. Unpubl. Report.

Futch, R. B., and G.E. Bruger. 1976. Age, growth, and reproduction of red snapper in Florida waters. Pages 165-184 in: H. R. Bullis, Jr., and A. C. Jones (editors), *Proceedings: Colloquium on Snapper-Grouper Fishery Resources of the Western Central Atlantic Ocean.* Fla. Sea Grant Report No. 17.

Hunter, J. R. 1985. Preservation of Northern Anchovy in Formaldehyde Solution. Pages 63-65 in: R. Lasker (ed.) *An egg production method for estimating spawning biomass of pelagic fish: an application to the northern anchovy, Engraulis mordax.* NOAA/NMFS Tech. Rep. 36.

Hunter, J.R., N.C.H. Lo, and R.J.H. Leong.1985. Batch fecundity in multiple spawning fishes. Pages 67-77 in: R. Lasker (ed.) *An egg production method for estimating spawning biomass of pelagic fish: application to the northern anchovy, Engraulis mordax.* NOAA/NMFS Tech. Rep. 36.

Hunter, J. R. and B. J. Macewicz. 1985. Measurement of spawning frequency in multiple spawning fishes. Pages 79-94 in: R. Lasker (ed.) *An egg production method for estimating spawning biomass of pelagic fish: an application to the northern anchovy, Engraulis mordax.* NOAA/NMFS Tech. Rep. 36.

Minton, R. V., J.P. Hawke and W.M. Tatum. 1983. Hormone induced spawning of red snapper, *Lutjanus campechanus. Aquaculture,* **30**: 363-368.

Moe, M. A., Jr. 1963. A survey of offshore fishing in Florida. Fla. St. Bd. Conserv. Mar. Lab., Prof. Pap. Ser. No. 4, 117 pp.

Moseley, F.N. 1965. Biology of the red snapper, *Lutjanus aya* Bloch, of the northwestern Gulf of Mexico. *Publ. Inst. Mar. Sci. Univ. Tex.* **11**:90-101.

Nelson , R. S. 1980. Growth and mortality aspects of natural populations of the red snapper, Lutjanus campechanus, in the west central Atlantic and Gulf of Mexico. M.S. Thesis, Louisiana State University, Baton Rouge, La. 73 pp.

Nelson, R. S. 1988. *A study of the life history, ecology, and population dynamics of four sympatric reef predators (Rhomboplites aurorubens, Lutjanus campechanus,Lutjanidae; Haemulon melanurum, Haemulidae; and Pagrus pagrus, Sparidae) on the East and West Flower Garden Banks.* Ph. D. Dissertation. North Carolina State University. 197 pp.

Nieland, D.L., and C.A. Wilson. 1993. Reproductive biology and annual variation of reproductive variables of black drum in the northern Gulf of Mexico. *Trans. Am. Fish. Soc.* **122**:318-327.

Patterson, W.F. 1999. *Aspects of the population ecology of red snapper, Lutjanus campechanus, in an artificial reef area off Alabama.* Ph. D.

dissertation, University of South Alabama, Mobile, Al.

Schirripa, M.J. and C.M. Legault [1999] Status of the red snapper in U. S. waters of the Gulf of Mexico: updated through 1998. National Marine Fisheries Service, Miami, FL, Sustainable Fisheries Division Contribution: SFD-99/00-75.106 p. Unpubl. Report.

Smith, G.B., H.M. Austin, S.A. Bortone, R.W. Hastings, and L H. Ogren. 1975. Fishes of the Florida Middle Ground with Comments on Ecology and Zoogeography. Florida Marine Research Publication No. **9**, Florida Dept. of Nat. Resources, Marine Research Laboratory, St. Petersburg, Florida.

West, G. 1990. Methods of assessing ovarian development in fishes: a review. *Aust. J. Mar. Freshwater Res.* **41**:199-222.

West, J. 1999. Red snapper revival. ***Florida Sportsman***, August 1999:16-25.

Wilson, C.A., and D.L. Nieland. 1994. Reproductive biology of red drum, *Sciaenops ocellatus*, from the neritic waters of the northern Gulf of Mexico. *Fish. Bull.* **92**:841-850.

Wilson, C.A., J.H. Render, and D.L. Nieland. 1994. Life history gaps in red snapper (*Lutjanus campechanus*), swordfish (*Xiphias gladius*), and red drum (*Sciaenops ocellatus*) in the northern Gulf of Mexico; age distribution, growth, and some reproduction biology. Final report to U. S. Dept. Comm., NMFS, Marine Fisheries Initiative (MARFIN) Coop. Agreement NA17FF0383-02. 79 pp.

Zar, J. H. 1984. Testing for Goodness of Fit. Pages 40–60 in: B Kurtz (ed.) *Biostatistical Analysis*, Prentice Hall, Englewood Cliffs, NJ.

Age Estimates from Annuli in Otoliths of Red Snapper, *Lutjanus campechanus*, from the Northern Gulf of Mexico

CHARLES A. WILSON[1,2], A. LOUISE STANLEY[1],
and DAVID L. NIELAND[1]
[1]*Coastal Fisheries Institute*
[2]*Department of Oceanography and Coastal Sciences*
Center for Coastal, Energy, and Environmental Resources
Louisiana State University
Baton Rouge, LA 70803-7503 USA

ABSTRACT

The red snapper *Lutjanus campechanus* is currently under stringent federal management in the Gulf of Mexico off the southeastern United States due to apparent overfishing of many populations. Management strategies employed to promote recovery of the species are dependent upon knowledge of various demographic variables such as the ages of individuals, the distribution of these ages (cohort strength) within the population, and maximum longevity. Thus a reliable and accurate ageing methodology for red snapper is of paramount importance. Annuli on otoliths have been used to age many species of fishes, including red snapper. However, the utility of this methodology for ageing red snapper has been questioned by those who dispute both the apparent longevity (over 50 yr) of red snapper and the position of the first annulus within the otolith.

We counted annuli and assessed edge condition in sagittal otoliths of 3,791 red snapper collected from the northern Gulf off Louisiana during the periods 1989 to 1992 and 1995 to 1998. Opaque annuli were validated by marginal increment analysis to form once per year from December through June. Among the otoliths examined, estimated ages ranged from 0.5 to 52.6 yr for individuals from 104 mm to 1,039 mm total length and from 0.017 kg to 22.793 kg total weight. The great heterogeneity in red snapper age at a given total length or total weight limits the use of morphometric variables as predictors of age.

KEY WORDS: Age, otoliths, red snapper

INTRODUCTION

The red snapper *Lutjanus campechanus* (Poey) (Family Lutjanidae) has been a significant element of the past and current commercial and recreational fisheries in the Gulf of Mexico (GOM) (Cato and Prochaska 1976, Moran 1988, Goodyear 1994, Schirripa and Legault 1999). However, documented commercial landings from United States territorial waters declined precipitously from historic highs of about 3,738 metric tons (mt) in 1974 to 1,015 mt in 1991 (Goodyear

1996, Schirripa and Legault 1999). Estimated recreational landings similarly waned from 4,734 mt in 1979 to 581 mt in 1990 (Schirripa and Legault 1997). Since 1991 both fisheries have been constrained by size limits, creel or trip limits, and quotas as established by the Gulf of Mexico Fishery Management Council GMFMC). Since 1991 the commercial fishery has achieved its allotted annual catch quota and has been subject to closure; however, the recreational fishery has often exceeded its allocation (Goodyear 1996, Schirripa and Legault 1999) and yet has experienced closure only in 1997, 1998, and 1999. The best efforts of the GMFMC and the commercial and recreational sectors notwithstanding, red snapper in the GOM may continue to be overfished (Goodyear 1995, Schirripa and Legault 1999).

Accurate information on the age structure of the red snapper population in the GOM is necessary to monitor year class strength, to conduct stock assessments, and to document population recovery. Previous efforts at estimating red snapper age have employed a variety of ageing methodologies. Bradley and Bryan (1975) cited the long red snapper spawning season and constant recruitment into the population as reasons for the difficulty in assigning red snapper ages from length frequency data. Moseley (1966) used scale annuli to age red snapper to age 4 years and advanced spawning as the causal factor in check formation. Wade (1978) also used scales to age red snapper to 9 years. Among 240 red snapper taken off the west coast of Florida, Futch and Bruger (1976) estimated red snapper ages of 1 to 5 years from 200 readable otoliths; however, they postulated ages up to 20 years for larger individuals whose otoliths were unreadable. Nelson and Manooch (1982) found red snapper age 1 to 16 years based on both scales and otoliths and demonstrated once yearly scale annulus formation in June and July from monthly mean marginal growth. A recent study has significantly extended the hypothesized longevity of red snapper in the GOM to 42 years (Szedlmayer and Shipp 1994). Render (1995) provided a preliminary validation of yearly annulus formation in sagittal otoliths and reported ages from 0 to 53 years for red snapper in Louisiana waters. Examinations of otoliths from 537 red snapper captured in the northwestern Atlantic Ocean from Beaufort, North Carolina south to the Florida Keys manifested a maximum longevity of 25 years (Manooch and Potts 1997). Among 907 red snapper from the GOM off Alabama, Patterson (1999) reported opaque annulus formation from January through June and maximum ages of 30 years for females and 31 for males.

Despite these efforts the longevity of red snapper remains controversial. Small sample sizes, a paucity of older specimens, and the failure to present legitimate validations of ages from hard parts (Beamish and McFarlane 1983) have variously hampered the above studies. It has further been speculated that larger and presumably older red snapper form numerous false annuli within otoliths (Rothschild et al. 1997). And both the timing of deposition and the position of the putative first annulus remain in question.

Otolith analyses have proven consistent in estimating ages of many fish species, including several from the temperate waters of the northern GOM (Johnson et al. 1983, Barger 1985, Beckman et al. 1988, Beckman et al. 1990, 1991). Herein we present our interpretations of the use of sagittal otoliths to estimate ages of red snapper from the GOM off Louisiana. Specifically addressed are the timing of formation and position of the first annulus, validation of the once yearly accretion of opaque annuli, and reader reproducibility.

METHODS AND MATERIALS

Red snapper from the northern GOM were sampled from recreational and commercial catches from 1989 to 1992 and from 1995 to 1998 by personnel of the Louisiana State University (LSU) Coastal Fisheries Institute and the Louisiana Department of Wildlife and Fisheries (LDWF). Although the vast majority of our sampling efforts were targeted at both wholesale facilities and charter boat docks located in Grand Isle and Port Fourchon, LA, the area of coverage in the northern GOM extended from off the Mississippi River Delta in the east to off Galveston, TX in the west. Morphometric measurements (fork length (FL) in mm, total (TW) or eviscerated body weight (BW) in g), both sagittal otoliths, and a sex determination when possible were taken from each specimen. Body weight was converted to TW, when necessary, with the equation TW = 1.101(BW) - 26.32 and total length (TL) was estimated with the equation TL = 1.073 (FL) + 3.56.

All otoliths were weighed, embedded in an epoxy resin, and then thin sectioned with a low speed saw equipped with a wafering blade as described in Beckman et al. (1988). Examinations of otolith sections were made with a compound microscope and transmitted light at 40X to 100X magnification. Counts of annuli (opaque zones) were accomplished by reading along the medial surface of the transverse section ventral to the sulcus; annuli were often inconsistent in other regions of the otolith section. Annulus counts were performed by two readers (A. L. Stanley and A. J. Fischer) without knowledge of date of capture or morphometric data. The appearance of the otolith margin was also coded as either opaque or transparent (Beckman et al. 1988, Beckman et al. 1990, 1991). Sections were recounted a second time by both readers when initial counts disagreed. Rather than excluding the small number of individuals for which a consensus could not be reached after a second reading, the assigned annulus count for these was that of Stanley. Fischer's annulus count and edge condition were used in those circumstances where Stanley's were missing. Annulus counting error between the two readers was evaluated after both the initial and second readings of the otolith sections. Reproducibility of the counts was determined with the coefficient of variation, the index of precision (Chang 1982), and average percent error (Beamish and Fournier 1981).

The frequency of opaque annulus formation was determined by marginal increment analysis and by plotting percent occurrence of otoliths with opaque margins by month (Beckman et al. 1988, Beckman et al. 1990, 1991). If one opaque and one translucent zone are shown to be formed each year, validation of annuli as being accreted once yearly is accomplished. Age estimates of red snapper were based on otolith opaque annulus counts and adjusted by edge condition. Based on previous studies of red snapper reproduction (Render 1995, Collins et al. 1996), a uniform hatching date of 1 July was assigned for all specimens.

RESULTS

During eight years of variable collection effort, 3,791 red snapper from recreational (N=274) and commercial (N = 3,517) catches were sampled for morphometric data and sagittal otoliths. Among the 1,438 male and 1,542 female specimens for which sex could be determined, females ranged from 242 to 1,039 mm TL and from 0.160 to 22.793 kg TW; males were 245 - 946 mm TL and 0.190 - 13.695 kg TW. Composite ranges for all specimens of either known or unknown sex were 104 - 1,039 mm TL and 0.017 - 22.793 kg TW. Distributions of 3,787 available TL and 3,718 available TW are shown in Figure 1 A and B, respectively.

Sagittae of red snapper are ovate, laterally compressed and have an indented sulcus on the medial surface. Although one can count purported growth rings in relatively small whole otoliths of red snapper less than age 5, it is difficult to discern annuli in the larger otoliths of older individuals (Futch and Bruger 1976). Thin transverse sections of the otoliths exposed semi-distinct translucent and opaque annuli which alternated from the core to the growing edge (Figure 2). The assumptive first annulus posed the most consistent problem for the readers. This annulus appeared as a diffuse "smudge" of opaque material variously located from contiguous to and continuous with the otolith core to totally isolated and somewhat distant form the core. Nonetheless, annulus counts ranging from 0 to 53 and edge conditions were determined by at least one reader for all 3,791 individuals sampled.

Reader One (Fischer) considered all of the otolith sections to be of sufficient quality to produce annulus counts; reader two (Stanley) provided annulus counts from all but two sections. After the initial counts, consensus between readers was achieved for 2804 individuals (Table 1). A second reading of the 987 sections for which annulus counts differed produced consensus for 3,762 individuals. The degree of agreement in red snapper opaque annulus counts between the two readers in each of the two readings was assessed. Average percent error (APE), coefficient of variation (CV), index of precision (D), and percentages of absolute differences in counts are given in Table 1.

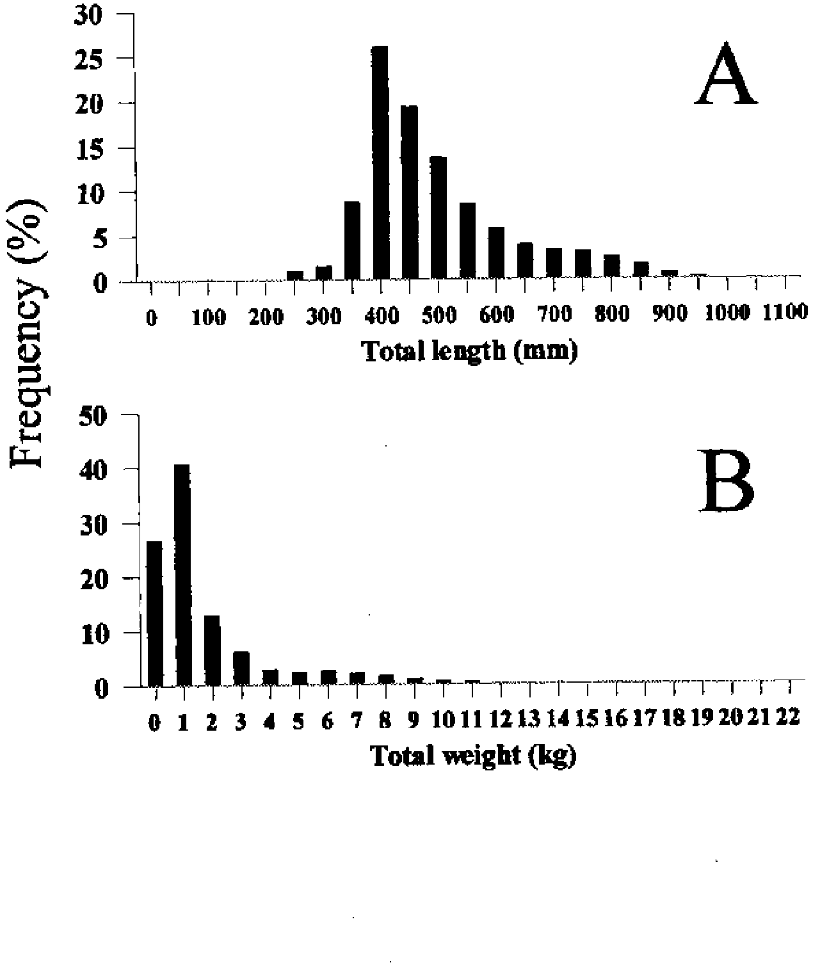


Figure 1. Frequency histograms for (A) total length (N=3,787) and (B) total weight (N = 3,718) of red snapper *Lutjanus campechanus* from the northern Gulf of Mexico.

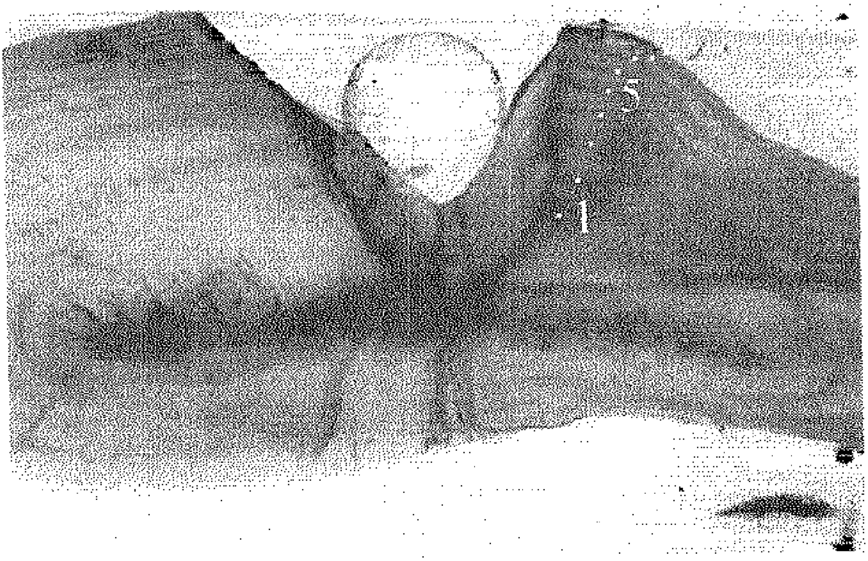


Figure 2. Transverse section of a red snapper, *Lutjanus campechanus*, sagittal otolith showing alternating opaque and translucent annuli. White dots indicate the opaque annuli. The first and fifth annuli are indicated with numerals.

Table 1. Average percent error (APE), coefficient of variation (CV), index of preecision(D), and absolute differences in red snapper otolith annulus counts for two readers on first and second readings.

	1st Reading	2nd Reading
APE	0.037	0.009
CV	0.045	0.0011
D	0.032	0.0008
0	73.96%	99.29%
± 1	23.27%	0.61%
± 2	2.24%	0.08%
> ± 3	0.54%	0.03%

The assumption of once yearly opaque annulus formation was tested with marginal increment analysis. Proportions of otoliths with opaque margins were plotted by month of capture for all individuals (N = 3,791), for those from individuals presumed to be sexually immature (ages less than or equal to 5, N = 2,143), and for those from individuals of presumptive sexual maturity (ages greater than 5, N = 948). Each of the three plots (Figure 3) features a single broad peak and a single broad valley and conclusively demonstrates opaque annulus formation from December through June and translucent annulus formation from July through November. Thus, the assumption of one to one correspondence between opaque annulus counts and estimated red snapper age in years is validated. Furthermore, this correspondence is validated for immature and mature individuals of all ages.

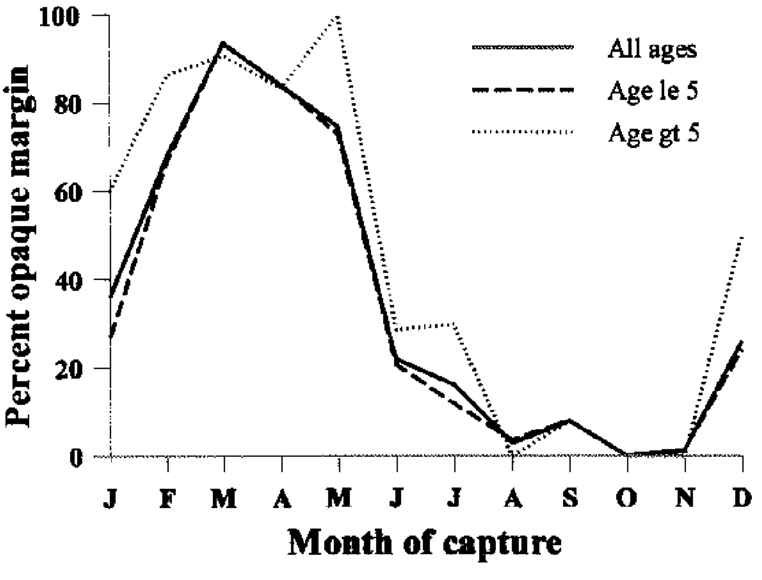


Figure 3. Percent of red snapper, *Lutjanus campechanus,* otoliths with opaque margin by month of capture. Sample sizes are 3,791 for all ages, 3,497 for ages than or equal to 5, and 294 for ages greater than 5.

Assuming once yearly accretion of opaque annuli, ages from 0.5 to 52.6 years were estimated from the annulus counts of the red snapper considered herein. The vast majority of specimens examined were ages 2 - 5 years and only 1.2% of the total number were greater than age 15 years (Figure 4). The otolith section from the oldest specimen examined is shown in Figure 5. Due to the large variability in age at a given TL (Figure 6), this variable is not a good estimator of red snapper age even at the smaller sizes where the age-size relation shows the greatest degree of linearity. Our data indicate that red snappers of 300 mm, 600 mm, and 800 mm TL could be ages 1 - 4 years, 3-9 years, and 3-50+ years, respectively.

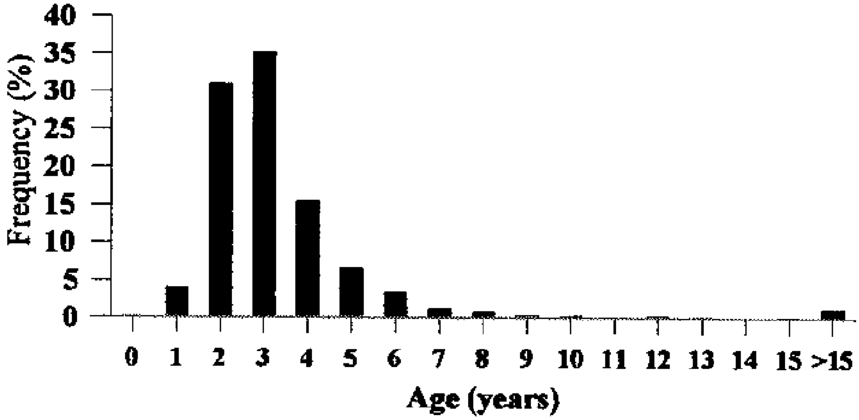


Figure 4. Frequency histogram of age in years for red snapper, *Lutjanus campechanus*, from the northern Gulf of Mexico. Sample size = 3,791.

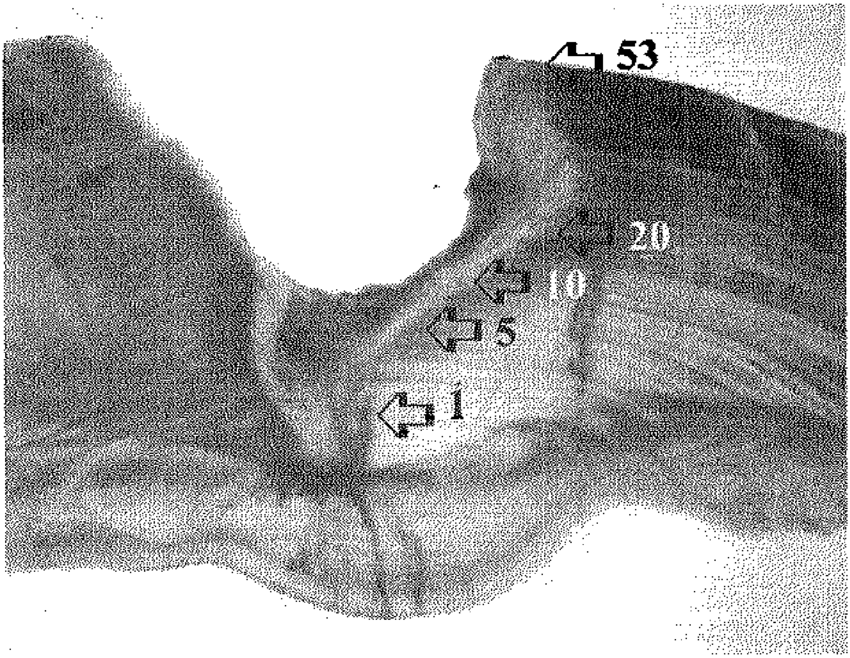


Figure 5. Transverse section of red snapper, *Lutjanus campechanus*, otolith indicating the first, fifth, tenth, twentieth, and fifty-third opaque annuli.

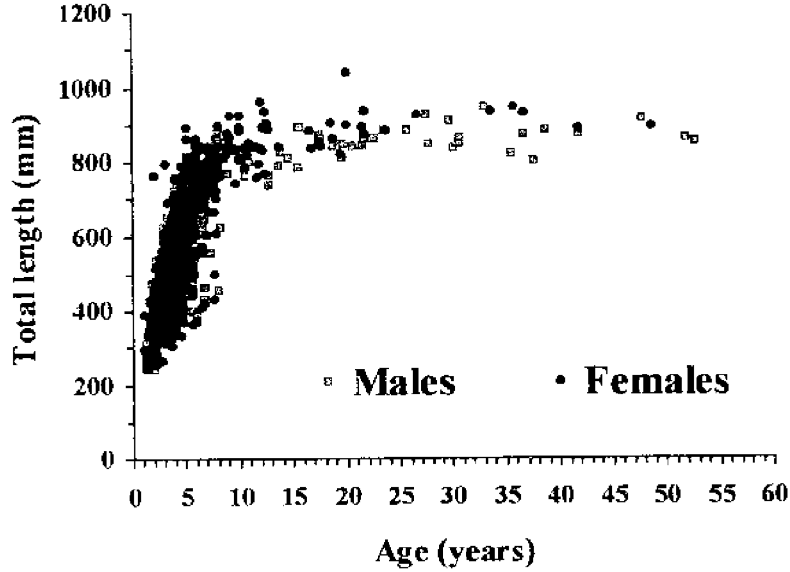


Figure 6. Scatterplot of red snapper, *Lutjanus campechanus*, age versus total length for 3,791 specimens from the northern Gulf of Mexico.

DISCUSSION

The use of otolith annuli as indicators of age has been validated for many freshwater and marine fish species, including the lutjanids *L. adetii* and *L. quinquelineatus* (Newman et al. 1996). Previous studies have utilized scales (Moseley 1966, Wade, 1978), otoliths (Futch and Bruger 1976, Render, 1995, Manooch and Potts 1997, Patterson 1999), scales and otoliths (Nelson and Manooch 1982) and scales, otoliths, and pectoral fin rays (Bortone and Hollingsworth 1980) to estimate red snapper ages. Among these, early attempts to validate age estimation from circuli of scales and annuli of otoliths have suffered from two shortcomings: 1) a small sample size and 2) a paucity of individuals over age 10 years. Nevertheless, they have produced a general consensus that transparent annuli (Nelson and Manooch, 1982) are formed during the spawning season (May to September in the GOM (Collins et al. 1996)). Our validation of opaque annulus formation in otoliths of red snapper during the winter and spring seasons is in substantial agreement with previous efforts. Given that yearly formation of opaque annuli has been validated for substantial numbers of red snapper from the Atlantic waters off North Carolina south to Florida (Manooch and Potts 1997) and the GOM waters off Alabama (Patterson

1999) and Louisiana (Render 1995, this study), the one to one correspondence between annuli and age in years should be indisputable.

Certainly, the reproducibility statistics indicate that the annuli of red snapper otoliths are more difficult to count than those of other fishes. Comparisons of between reader age estimates in several species of the family Sciaenidae have yielded near 100% agreement (Beckman et al. 1988, Beckman et al. 1990, Barbieri et al. 1993, Lowerre-Barbieri et al. 1995). Sciaenid otoliths are comparatively massive and annuli are especially well defined. Conversely, red snapper otoliths are relatively thin and fragile and the annuli become increasingly less well defined with advanced age. But, even given the above, a first reading followed by a second reading produced consensus in age estimates for 99.29% of those red snapper considered herein. Patterson (1999) reported 93.8% between reader consensus of red snapper annulus counts after two readings.

The variable position and the diffuse appearance of the first annulus formed during the first winter following hatching are functions of both the protracted red snapper spawning season and the rapid growth rate of juvenile red snapper. Those individuals which are spawned early in the season will experience proportionally more growth (and more translucent zone accretion) than will a late spawned individual before opaque annulus accretion begins during the following winter; thus the first opaque annulus will be more distant from the otolith core in the former instance than under the later circumstances. Also with the first opaque annulus accreting at a rate theoretically corresponding to the rapid growth rate experienced during the juvenile stage, the resulting first annulus is broader and more diffuse in appearance than annuli produce during times of reduced growth rates in later life.

It is difficult to compare the maximum observed red snapper longevity reported here to those reported in earlier studies due to the assortment of ageing techniques (scales, otoliths, length frequencies) and the variety of sources (commercial, recreational, or both) utilized. All show a predominance of young individuals (<10 years). However, recent advances and refinements in otolith preparation technology have allowed red snapper to be reliably aged up to 42 years (Szedlmayer and Shipp 1995), 53 years (Render 1995), 31 years (Patterson 1999), and 52 years (this study). Furthermore, Baker's (1999) radiometric validation of red snapper longevity to at least 40 years provides additional substantiation to those ages derived from otolith annuli.

As is much the case in humans, red snapper size is little indication of red snapper age. For example, consider the International Game Fishing Association world rod and reel record red snapper whose otoliths were given to us for age analysis. This individual was caught off the coast of Louisiana by Doc Kennedy of Grand Isle, LA on 23 June 1996; it was 22.793 kg (50 lb, 4 oz) TW, 1039 mm (40.9 in) TL, and 965 mm (38 in) FL. Given the immense size of this red snapper, one would reasonably expect it to be ancient by red snapper standards. However, our analysis revealed it to be only 19.98 years old. Conversely, the two oldest red snapper we encountered, 52.63 and 51.73 years old, were a comparatively small 851 mm TL and 862 mm Tl, respectively, and 7.886 kg TW and 9.188 kg TW, respectively. A similar pattern was noted by Patterson (1999) among the red snapper which he sampled from the GOM off Alabama.

Personnel at the LSU Coastal Fisheries Institute continue to investigate the nuances of deriving red snapper ages from sagittal otoliths. We have expanded our sampling efforts to include age 0 and age 1 red snappers collected during the National Marine Fishery Service's Summer SEAMAP and Fall Groundfish cruises in the GOM. Core to first annulus measurements made on otolith sections from these young individuals will give us a better understanding of when and how the first annulus is accreted. We will also soon be analyzing the otoliths of the several oldest red snapper we have encountered for the presence and quantity of Carbon-14 which was released into the atmosphere during atomic testing during the 1940s, 1950s, and 1960s. Elevated levels of this isotope in the otoliths, when compared to contemporaneous levels of the isotope in the water of the GOM, would support the longevity we have observed and report herein.

LITERATURE CITED

Baker, M. S., Jr. 1999. Radiometric age validation of red snapper, *Lutjanus campechanus*, and red drum, *Sciaenops ocellatus*, from the northern Gulf of Mexico. M.S. Thesis. Louisiana State University, Baton Rouge. 59 pp.

Barbieri, L. R., M. E. Chittenden, Jr and C. M. Jones. 1993. Age, growth, and mortality of Atlantic croaker, *Micropogonias undulatus*, in the Chesapeake Bay region, with a discussion of apparent geographic changes in population dynamics. *Fish. Bull.* **92**:1-12.

Barger, L. E. 1985. Age and growth of Atlantic croakers in the northern Gulf of Mexico, based on otolith sections. *Trans. Am. Fish. Soc.* **111**:847-850.

Bradley, E. and C. E. Bryan. 1975. Life history and fishery of the red snapper (*Lutjanus campechanus*) in the northwestern Gulf of Mexico. *Proc. Gulf Carib. Fish. Inst.* **27**:77-106

Beamish, R. J. 1979. New information on the longevity of the Pacific ocean perch (*Sebastes alutus*). *J. Fish. Res. Board Can.* **36**:1395-1400.

Beamish, R. J. and D. A. Fournier. 1981. A method for comparing the precision of a set of age determines. *Can. J. Fish. Aquat. Sci.* **38**:982-983.

Beamish, R. J. and G. A. McFarlane. 1983. The forgotten requirement for age validation in fisheries biology. *Trans. Am. Fish. Soc.* **112**:735-743.

Beckman, D. W., C. A. Wilson and A. L. Stanley. 1988. Age and growth of red drum, *Sciaenops ocellatus*, from offshore waters of the northern Gulf of Mexico. *Fish. Bull.* **87**:17-28.

Beckman, D. W., A. L. Stanley, J. H. Render and C. A. Wilson. 1990. Age and growth of black drum in Louisiana waters of the Gulf of Mexico. *Trans. Am. Fish. Soc.* **119**:537-544.

Beckman, D. W., A. L. Stanley, J. H. Render and C. A. Wilson. 1991. Age and growth-rate estimation of sheepshead *Archosargus probatocephalus* in Louisiana waters using otoliths. *Fish. Bull.* **89**:1-8.

Bortone, S. A. and C. C. Hollingsworth. 1980. Aging red snapper, *Lutjanus campechanus*, with otoliths, scales, and vertebrae. *Northeast Gulf Sci.* **4**:60-63.

Cato, J. C. and F. J. Prochaska. 1976. The Gulf of Mexico commercial and recreational red snapper-grouper fishery: an economic analysis of production, marketing, and prices. Pages 95-128 In H. R. Bullis, Jr. and A. C. Jones, eds. Proceedings: colloquium on snapper-grouper fishery resources of the western central Atlantic Ocean. *Fla. Sea Grant Coll. Rep.* **17**, 333 pp.

Chang, W.B. 1982. A statistical method for evaluating the reproducibility of age determination. *Can. J. Fish. Aquat. Sci.* **39**:1208-1210.

Collins, L. A., A. G. Johnson and C. P. Keim. 1996. Spawning and annual fecundity of the red snapper (*Lutjanus campechanus*) from the northeastern Gulf of Mexico. Pages 174-188 in: F. Arreguin-Sanchez, J. L. Munro, M. C. Balgos and D. Pauly, eds. *Biology, fisheries and culture of tropical groupers and snappers. ICLARM Conf. Proc.* **48**, 449 pp.

Futch, R. B. and G. E. Bruger. 1976. Age, growth, and reproduction of red snapper in Florida waters, Pages 165-183 in: Bullis, H. R., Jr. and A. C. Jones (eds.) *Proceedings: Colloquium on snapper-grouper fishery resources of the western central Atlantic Ocean.* Fla. Sea Grant Coll. Rep. **17**, 333 pp.

Goodyear, C. P. 1994. Red snapper in U. S. waters of the Gulf of Mexico. National Marine Fisheries Service, Southeast Fisheries Science Center, Miami Laboratory, Cont. MIA-93/94-63:1-146. Unpubl. MS.

Goodyear, C. P. 1995. Red snapper in U.S. waters of the Gulf of Mexico. National Marine Fisheries Service, Southeast Fisheries Center, Miami Laboratory, Contr. MIA-95/96-05:1-171. Unpubl. MS.

Goodyear, C. P. 1996. An update of red snapper harvest in U.S. waters of the Gulf of Mexico. National Marine Fisheries Service, Southeast Fisheries Center, Miami Laboratory, Contr. MIA-95/96-60:1-171.

Unpubl. MS.

Johnson, A. G., W. A. Fable Jr., M. L. Williams and L. E. Barger. 1983. Age, growth, and mortality of king mackerel, *Scomberomorus cavalla*, from the southeastern United States. *Fish. Bull.* **81**:97-106.

Lowerre-Barbieri, S. K., M. E. Chittenden, Jr. and L. R. Barbieri. 1995. Age and growth of weakfish, *Cynoscion regalis*, in the Chesapeake Bay region with a discussion of historical changes in maximum size. *Fish. Bull.* **93**:643-656.

Manooch, C. S., III and J. C. Potts. 1997. Age and growth of red snapper, *Lutjanus campechanus*, Lutjanidae, collected along the southeastern United States from North Carolina through the east coast of Florida. *J. Elisha Mitchell Sci. Soc.* **113**:111-122.

Moran, D. [1988] Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Gulf of Mexico) -- red snapper. U. S. Fish Wildl. Serv. Biol. Rep. 82(11,83). U. S. Army Corps of Engineers, TR EL-82-4. 19 pp. Unpubl. MS.

Moseley, F. N. 1966. Biology of red snapper, *Lutjanus aya* Bloch, of the northwestern Gulf of Mexico. *Publ. Inst. Mar. Sci., Univ. Texas* **11**:90-101.

Nelson, R. S. and C. S. Manooch III. 1982. Growth and mortality of red snappers in the west-central Atlantic Ocean and northern Gulf of Mexico. *Trans. Amer. Fish. Soc.* **111**:465-475.

Newman, S. J., D. McB. Williams and G. R. Russ. 1996. Age validation, growth and mortality rates of the tropical snappers (Pisces : Lutjanidae) *Lutjanus adetii* (Castelnau, 1873) and *L. quinquelineatus* (Bloch, 1790) from the central Great Barrier Reef, Australia. *Mar. Freshwater Res.* **47**:575-584.

Patterson, W. F., III. 1999. *Aspects of the population ecology of red snapper* Lutjanus campechanus *in an artificial reef area off Alabama.* Ph.D. Dissertation. University of South Alabama, Mobile. 164 p.

Render, J. H. 1995. *The life history (age, growth, and reproduction) of red snapper (*Lutjanus campechanus*) and its affinity for oil and gas platforms.* Ph. D. Dissertation, Louisiana State University, Baton Rouge. x + 76 p.

Rothschild, B. J., A. F. Sharov and A. Y. Bobyrev. [1997] Red snapper stock assessment and management for the Gulf of Mexico. Report submitted to the Office of Science and Technology, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U. S. Department of Commerce. 52 pp., 50 figs., 27 tables, 1 appendix. Unpubl. MS.

Schirripa, M. J. and C. M. Legault. [1997] Status of the red snapper in the U.

S. waters of the Gulf of Mexico: Updated through 1996. National Marine Fisheries Service, Southeast Fisheries Science Center, Miami Laboratory, Cont. MIA-97/98-05:1-40. Unpubl. MS.

Schirripa, M. J. and C. M. Legault. [1999] Status of the red snapper in the U. S. waters of the Gulf of Mexico: Updated through 1998. National Marine Fisheries Service, Southeast Fisheries Science Center, Sustainable Fisheries Division, Cont. SFD-99/00-75:1-29. Unpubl. MS.

Szedlmayer, S. T. and R. L. Shipp. 1994. Movement and growth of red snapper, *Lutjanus campechanus*, from an artificial reef area in the northeastern Gulf of Mexico. *Bull. Mar. Sci.* **55**:887-896.

Wade, C. W. 1981. Age and growth of spotted seatrout and red snapper in Alabama. Pages 345-354 in: J. Sweeney (ed.) *Proc. Ann. Conf. S.E. Assoc. Fish & Wildl. Agencies.* **35**.

Age Validation of Red Snapper, *Lutjanus campechanus*, and Red Drum, *Sciaenops ocellatus*, from the Northern Gulf of Mexico Using ^{210}Po/^{226}Ra Disequilibria in Otoliths

M. SCOTT BAKER, JR.[1], CHARLES A. WILSON[1],
and DANIEL L. VAN GENT[2]
[1]Coastal Fisheries Institute
Louisiana State University
Baton Rouge, LA 70803 USA
[2]Nuclear Science Center
Louisiana State University
Baton Rouge, LA 70803 USA

ABSTRACT

Radiometric analysis of naturally occurring ^{226}Ra and its daughters within otoliths was used to validate age estimates derived from sectioned otoliths of red snapper and red drum from the northern Gulf of Mexico. Ages of most fish species are often estimated by counting presumed annual growth zones in otoliths. Radiometric age validation provides an independent alternative to conventional methodology by removing reader interpretation variability from the age estimation process. Red snapper and red drum are two moderately long-lived species inhabiting the Gulf of Mexico which are currently under intense state and federal management. Despite the poor counting statistics and the high backgrounds incurred with the technique used in this study, radiometric age estimates closely approximated otolith section ages for both red snapper and red drum, thus validating the continued use of otolith sections as the most accurate way to estimate ages for these species.

KEY WORDS: Otoliths, radiometric, validation

INTRODUCTION

Most fishes are aged by counting growth increments on hard-part structures such as otoliths. For most fishes, each alternating opaque and translucent zone together in the otolith represents one year; however, it is possible that some fishes deposit variable growth zones each year. Therefore, in order for fisheries management to be successful, each species should be carefully examined to determine if the zones present in an otolith are being deposited annually. Two species that have warranted this consideration in the past are the reef-dwelling red snapper, *Lutjanus campechanus*, (family Lutjanidae) and the estuarine dependent red drum, *Sciaenops ocellatus*, (family Sciaenidae). Validation of the age determination methods used to estimate age for these species is critical to management strategies.

Methods previously used to estimate ages for red snapper have been largely inconsistent and have included presumed annulus counts from scales, whole otoliths, and otolith sections (Moseley 1966, Wade 1981, Nelson and Manooch 1982, Nelson et al. 1985, Szedlmayer and Shipp 1994, Wilson et al. 1994, Wilson et al. 1998). Over the past 30 years, these life history studies have resulted in increasing estimates of longevity (Table 1). As a result, age estimation of red snapper has been prohibitive. Unlike red snapper, age estimates of red drum from the Gulf of Mexico have easily been validated throughout its entire life history using traditional methodologies (Beckman et al. 1988, Beckman et al. 1988, Murphy and Taylor 1991). Annuli on sagittal otolith sections are now considered the best estimator of age for red snapper, as well as most fish species, from the northern Gulf of Mexico (Szedlmayer and Shipp 1994, Shirripa and Burns 1997, Wilson et al. 1994, Wilson et al. 1998). Independent validation of the otolith section technique used to age red snapper and red drum is critical for the proper management of these economically important fish species (Beamish and McFarlane 1983, Wilson et al. 1998).

Age estimation through radiometric analysis of the disequilibria of radium-226 (^{226}Ra) and its daughter elements in otoliths provides that virtually any marine fish species can be analyzed effectively over any time period up to 100 years (Bennett et al. 1982). Radiometric analysis of the naturally occurring ^{226}Ra and its daughters within otoliths has been used to validate age estimates derived from otoliths in a number of species (Bennett et al. 1982, Campana et al. 1990, Fenton et al. 1991, Smith et al. 1991, Fenton et al. 1992, Campana et al. 1993, Kastelle et al. 1994, Fenton and Short 1995, Milton et al. 1995, Stewart et al. 1995). Most of these studies however have focused on slow growing, long-lived species inhabiting deep, cold water environments with little emphasis on temperate and tropical species.

Our objective was to determine with radiometric analysis whether ages derived from sectioned otoliths indicate true (radiometric) age for red snapper, *Lutjanus campechanus*, and red drum, *Sciaenops ocellatus*, from the northern Gulf of Mexico.

Table 1. Von Bertalanffy growth parameters of red snapper, *Lutjanus campechanus*, from the northern Gulf of Mexico (O = otoliths, S = scales, F = female, M = male, MaxAge = maximum age, L_∞ = maximum length reported as fork length (FL) or total length (TL), k = growth coefficient, t_0 = time at age-0).

Method	Max. Age	L_∞	K	t_0	Source
S	4+	-	-	-	Moseley, 1966
S	9	-	-	-	Wade, 1981
S,O	13	941 TL	0.17	-0.10	Nelson and Manooch, 1982
O	10	925 TL	0.14	-0.00	Nelson et al., 1985
O	42	1025 TL	0.13	-	Szedlmayer and Shipp, 1994
O	53	772 FL$^{(F)}$	0.18	0.00	Wilson et al., 1994
		859 FL$^{(M)}$	0.09	2.21	
O	49	913 FL$^{(F)}$	0.16	0.72	Wilson, et al., 1998
		842 FL$^{(M)}$	0.17	0.58	

MATERIALS AND METHODS

The red snapper and red drum otoliths utilized in this study were sampled from the northern Gulf of Mexico commercial and recreational harvests (1986 -1998). Fork length in mm and eviscerated body weights in g were recorded for each fish. Both sagittal otoliths were removed and weighed to the nearest milligram. Sex of each specimen and location of capture was recorded when possible. Otolith preparations and ageing protocols for both red snapper and red drum followed standard procedures previously described elsewhere (Beckman et al. 1988, Wilson et al. 1994, Render 1995).

Otoliths used in the radiometric analyses were cored to the age class size of 1 to 4 years, depending on the species and the number of samples available. Since radiometric procedures used in this study require a minimum of 100 mg otolith material per sample, otolith cores were pooled for most age categories. As many as 20 red snapper sagittae (age 0.4 years) were pooled together to establish the minimum sample weight. Red drum otolith cores were larger than those of red snapper and were analyzed individually. Cores were obtained by embedding whole otoliths in araldite epoxy resin and progressively removing visible layers with a Buehler low speed saw and variable speed sanding wheel. Care was taken to clean the sanding wheel with a bristle brush and distilled water after each otolith was processed. Otolith cores were rinsed with distilled water, dried, weighed to the nearest milligram.

Radiometric analysis of ^{226}Ra and its daughters in otolith cores was conducted with a Photon - Electron Rejecting Alpha Liquid Scintillation (PERALS®) spectrometer. Alpha measurements were quantified with a Dell 450/L personal computer fitted with Maestro software, Model A65-B1, Version 3.06. Polonium-210 (half-life, 138 days) was assumed to be in secular equilibrium with ^{210}Pb (half-life, 22.3 years) in all samples since all otolith samples were collected at least 1 year prior to analysis. Otolith sample weights ranged from 0.153 to 0.733 g (mean = 0.388 g, n = 8). The mean reagent blank for 210Po was 0.056 ± 0.011 disintegration per minute (dpm). Recovery of the ^{209}Po standard was highly variable and ranged from 10% to 47%. Polonium samples and standards were analyzed for a minimum of 0.7 days and a maximum of 4.6 days (mean = 3 days, n = 8). The mean reagent blank for ^{226}Ra was 0.049 ± 0.019 dpm. Recovery of the ^{224}Ra standard was also highly variable and ranged from 14% to 60%. Radium samples and standards were analyzed for a minimum of 1 day and a maximum of 4.9 days (mean = 3.3 days, n = 7).

Otolith cores within an age group were placed in a 20 ml borosilicate glass scintillation vial and spiked with a known aliquot of ^{224}Ra and ^{209}Po. The otoliths were dissolved in 12 N HCl, reduced to 3 ml over heat, and cooled to room temperature in an ice bath. The solution was brought to neutrality by addition of concentrated NH_4OH. Before simultaneous extraction of ambient

^{226}Ra and the standard ^{224}Ra from the dissolved otolith solution, an extractive scintillator for radium (Radaex®) was converted from the acid phase to a salt in order to activate the solution. This procedure was performed by equilibrating the desired quantity of acid-form Radaex with an approximately equal volume of 0.35 M $NaNO_3$ and 0.2 M NaOH. Gentle swirling of the two-phase solution by hand for 5 minutes activated the solution. Approximately 3 g of the extractive scinillator was extracted from the equilibration vial with a 5 ml disposable syringe and transferred to the vial containing the otolith solution. The two-phase solution was capped and swirled vigorously for 15 minutes to allow for transfer of radium into the activated Radaex®. One gram of Radaex was removed from the vial by a 5 ml disposable syringe and transferred to a 10 x 75 mm borosilicate glass (or silicon) culture tube.

The solution was sparged (with a disposable Pasteur pipette as a sparging lance) with dry, toluene-saturated ultra-pure argon gas for 5 minutes and capped with a silicone stopper. To ensure no loss of solution, the top of the culture tube including the cap was sealed with parafilm. The otolith sample solution was separated from the unused radium extractive scintillator solution with a 30 mL separatory funnel. Polonium-210 and the standard ^{209}Po were extracted from the sample solution in exactly the same manner but with an extractive scintilallor for polonium (Polex®). Blanks were prepared in the exact manner as the sample/standard combinations.

Activity ratios (^{210}Po/^{226}Ra) were calculated for each red snapper (n = 6) and red drum (n = 2) sample. Lead-210 and its daughter product ^{210}Po were assumed to be in secular equilibrium in all samples. The ratio of ^{210}Po/^{226}Ra in otolith cores was used to predict a radiometric age using the equation:

$$t = \frac{-1}{\lambda_p}\left[\frac{\ln(1 - S)}{1 - Ro}\right]$$

where t = age (yrs), λ_p = decay constant for ^{210}Pb (-0.031/yr), S = $(^{210}Po/^{226}Ra)_t$ = activity at time t assuming insignificant ^{222}Rn loss, and $R_0 = (^{210}Po/^{226}Ra)_t$ = initial activity ratio at time t = 0.0 (Baker 1999). To compare the radiometric ages with the section ages of the same otoliths, the time between collection and analysis was added to the otolith section ages. A paired two-sample t-test comparison was used to test for differences between radiometric and otolith section age estimates.

RESULTS

In all red snapper otolith cores, the ^{210}Po/^{226}Ra activity ratio increased as presumed annuli count increased (Table 2). Radium-226 activity was highest in the 0.4 year core sample (0.975 (0.433 dpm/) and lowest in the three year core sample (0.208 (0.052 dpm/g). For each core sample, ^{210}Po activity (dpm/g (1σ) was proportional to ^{226}Ra activity and otolith section age. The initial activity ratio (R_0), calculated by radiometric analysis of the 0.4 year age group, was used to propagate a radiometric age estimate of -1.5 years. The negative value was assumed be a result of measurement error associated with the radiometric technique; therefore, $R_0 = 0.0$. Mean radiometric age estimates including storage time were 1.6, 6.8, 6.3, 14.1, 21.1, and 36.8 years (Figure 1a). A paired 2-sample t-test comparison indicated that radiometric ages were not significantly different from otolith section ages ($t = 0.763$, $p = 0.479$).

As with red snapper, the mean activity ratio in both red drum core samples corresponded to otolith section age (Table 2). The initial activity ratio R_0 was assumed to be 0.0 and ^{222}Rn loss from within otolith cores was assumed to be insignificant. Mean radiometric age estimates including storage time were 19.0 and 32.1 years (Figure 1b).

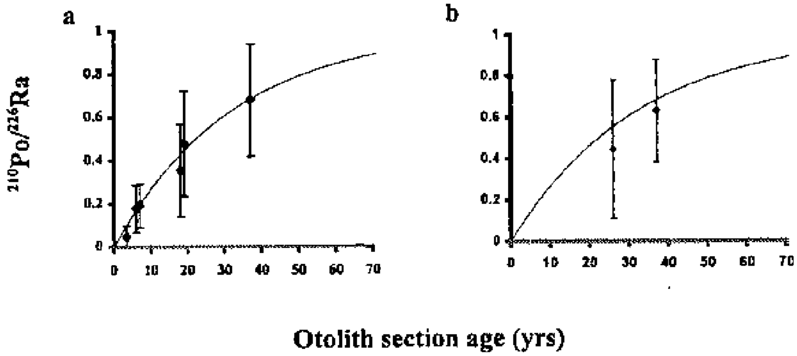


Figure 1. Observed $^{210}Po/^{226}Ra$ ratios of red snapper (a) and red drum (b) otolith cores plotted against theoretical $^{210}Po/^{226}Ra$ ratio with respect to time (—). All age estimates assume that the initial activity ratio (R_0) = 0.0 and $^{222}R_n$ loss from otoliths is negligible. Errors are expressed as ± 1σ.

Table 2. Results of the ^{210}Po/^{226}Ra age validation of red snapper and red drum otolith section age estimates from the northern Gulf of Mexico. The radiometric ages calculated include the time between collection and analysis (storage time). Otolith sample weights ranged from 0.153 to 0.733 g (mean = 0.388 g, n = 8).

Species	Otolith age (incl. storage time in years)	Cure Size (years)	^{236}Ra (dpm g^{-1} ± 1 s)	^{210}Po (dpm g^{-1} ± 1 s)	Radiometric age in years
Red snapper	3.4 (3)	0.4	0.975 ± 0.433	0.046 ± 0.046	1.6 ± 1.7
	7 (2)	2	0.447 ± 0.196	0.085 ± 0.026	6.8 ± 3.5
	6 (1)	1	0.058 ± 0.249	0.103 ± 0.047	6.3 ± 3.8
	18 (8)	2	0.375 ± 0.217	0.133 ± 0.023	14.1 ± 7.7
	19 (9)	3	0.208 ± 0.052	0.009 ± 0.044	21.1 ± 7.7
	37 (7)	2	0.708 ± 0.228	0.481 ± 0.099	36.8 ± 9.7
Red drum	26 (6)	1	0.651 ±0.040[1]	0.221 ± 0.186	19.0 ± 13.1
	39 (7)	1	0.757 ± 0.267	0.478 ± 0.054	32.1 ± 9.1

[1] Mean ^{226}Ra activity of a 70g, ~1 yr old red drum pooled otolith sample determined with gamma spectrometry (Baker 1999).

DISCUSSION

The results of this study indicated that mean radiometric age estimates closely approximated age estimates derived from otolith sections of both red snapper and red drum from the northern Gulf of Mexico. The findings confirm that red snapper deposit one opaque and one translucent zone each year at least up to 30 years of age. This study was the second instance where relatively young lutjanids (10 years of age) were successfully aged using analyses of the ^{210}Po/^{226}Ra radioisotopic pair (Milton et al. 1995). Although red drum age estimation was successful using radiometric dating, more sample analyses over a broader age range are needed. These results however, support the continued use of counting annuli on sagittal otolith sections as the most effective age determination method for red snapper and red drum from the Gulf of Mexico. Radiometric dating of otoliths requires that three assumptions be satisfied:

i) uptake of ^{226}Ra into the otolith occurs at rate constant to the mass increase of the otolith

ii) the initial activity ratio (R_0) in otoliths is close to zero or at least known

iii) the otolith remains a chemically-closed system in terms of no loss or gains of radionuclides with respect to time.

The first assumption concerning constant ^{226}Ra uptake was largely circumvented in this study due to our use of cores instead of whole otoliths. In addition, individual ^{226}Ra activities for each sample were used instead of an average ^{226}Ra activity. The second assumption was confirmed in this study through successful radiometric analysis of the young-of-the-year (YOY) age group of red snapper. Otolith microchemical analyses of Pb and Ba, stable elements which serve as proxies for ^{210}Pb and Ra, have also indicated that $R_0 = 0.0$ is applicable for both red snapper and red drum YOY (Baker, 1999). The possibility of ^{222}Rn loss is important and must be considered. Significant loss of ^{222}Rn, a noble gas, would lead to less ^{210}Po than expected and thus a reduction in the radiometric age estimate. We believe that ^{222}Rn loss is possible, given the characteristics of the gas and the internal composition of the otolith; however, the radiometric technique used in this study produced age estimates with high variability which probably masked any detectable loss of ^{222}Rn if in fact ^{222}Rn loss did occur. Nevertheless, mean radiometric and otolith section age estimates of red snapper were determined to be in close agreement.

The observed activities within red snapper otolith cores indicated that 226Ra activity decreased as otolith mass (age) increased. There are two possibilities as to why ^{226}Ra was higher in young cores rather than older ones. Radium is deposited onto the otolith at a nearly constant ratio to that of Ca (Moore et al. 1973). Red snapper exhibit a strong linear relationship between otolith weight and age until approximately 10 years of age when the relationship ceases to be significant (Wilson et al. 1998). The otolith weight-to-age relationship could explain the initial high values of ^{226}Ra in very young fish and why ^{226}Ra activities decrease with increasing age of the otolith core. Secondly, ^{226}Ra activity in general is significantly higher in the riverine and estuarine waters of the coastal shelf than areas offshore (Moore and Scott, 1986; Kraemer and Curwick 1991). Red snapper exhibit a major shift in habitat from the time when they are young (≤ 2 years) to the time of maturity, thus encountering variable concentrations of ^{226}Ra. Juvenile red snapper are commonly found on the nearshore shrimping grounds and adults are generally associated with offshore environments and areas containing hard substrate such as natural reefs and oil and gas platforms (Render 1995). Habitat preference by life history stage may play a key role in ^{226}Ra accumulation.

Although the results of this study were satisfactory to accomplish the objective of age validation, analyses of radionuclides in fish otoliths was difficult with the PERALS system. Reagent blanks used in this study were excessively high compared to those used in traditional, ultra-low level alpha spectrometry techniques; however, the significance of the high values was reduced by the high 226Ra activities observed in otolith cores. Highly variable recoveries from dissolved otolith solutions was also a significant problem and is most likely attributable to Ca interference during the radionuclide extraction procedures. The radiometric dating method should prove successful to other fishes of the Gulf of Mexico provided that these species have ^{226}Ra activities similar to the high values observed in this study. Radiometric age validation of short to long-lived marine fishes holds great promise, provided that further research is undertaken to improve existing methodology.

ACKNOWLEDGEMENTS

Funding for this research was provided by the Louisiana Department of Wildlife and Fisheries and the Louisiana State University Coastal Fisheries Institute. We thank Louise Stanley and David L. Nieland for assistance in the age and growth laboratory.

LITERATURE CITED

Baker, M.S., Jr. 1999. Radiometric age validation of red snapper, *Lutjanus campechanus*, and red drum, *Sciaenops ocellatus*, from the northern Gulf of Mexico. M. S. Thesis. Louisiana State University, Baton Rouge, LA. 65 pp.

Beamish, R.J., and G.A. McFarlane. 1983. The forgotten requirement for age validation in fisheries biology. *Trans. Am. Fish. Soc.* **112**:735-743.

Beckman, D.W., G.R. Fitzhugh, and C.A. Wilson. 1988. Growth rates and validation of age estimates of red drum, *Sciaenops ocellatus*, in a Louisiana salt marsh impoundment. *Contr. Mar. Sci.* **30**:95-98.

Beckman, D.W., C.A. Wilson, and A.L. Stanley. 1988. Age and growth of red drum, *Sciaenops ocellatus*, from offshore waters of the northern Gulf of Mexico. *Fish. Bull.* **87**:17-28.

Bennett, J.T., G.W. Boehlert, and K.K. Turkian. 1982. Confirmation of longevity in *Sebastes diploplora* (Pisces: Scorpaenidae) from 210Pb/226Ra measurements in otoliths. *Mar. Biol.* **71**:201-215.

Campana, S.E., K.C., T. Zwanenburg, and J.N. Smith. 1990. ^{210}Pb/^{226}Ra determination of longevity in redfish. *Can. J. Fish. Aquat. Sci.* **47**:163-165.

Campana, S.E., H.A. Oxenford, and J.N. Smith. 1993. Radiochemical determination of longevity in flyingfish, *Hirundicthys affinis*, using ^{228}Th/^{228}Ra. *Mar. Ecol. Prog. Ser.* **100**:211-219.

Fenton, G.E., S.A. Short, and D.A. Ritz. 1991. Age determination of orange roughy, *Hoplostethus atlanticus* (Pisces: Trachichthyidae), using ^{210}Pb/^{226}Ra disequilibria. *Mar. Biol.* **109**:197-202.

Fenton, G.E., and S.A. Short. 1992. Fish age validation by radiometric analysis of otoliths. *Aust. J. Mar. Freshwater Res.* **43**: 913-922.

Fenton, G.E., and S.A. Short. 1995. Radiometric analysis of blue grenadier, Macruronus novaezelandiae, otolith cores. *Fish. Bull.* **93**:392-396.

Kastelle, C.R., D.K. Kimura, A.E. Nevissi, and D.R. Gunderson. 1994. Using ^{210}Pb/^{226}Ra disequilibria for sablefish, *Anoplopoma fimbria*, age validation. *Fish. Bull.* **92**:292-301.

Kraemer, T.F., and P.B. Curwick. 1991. Radium isotopes in the lower Mississippi River. *J. Geophys. Res.* **96(C2)**:2797-2806.

Milton, D. A., S. A. Short, M. F. O'Neill, and S. J. M. Blaber. 1995. Ageing of three species of tropical snapper (Lutjanidae) from the Gulf of Carpentaria, Australia, using radiometry and otolith counts. *Fish. Bull.* **93**:103-115.

Moore, W.S., Krishnaswami, S., and S.G. Bhat. 1973. Radiometric determination of coral growth rates. *Bull. Mar. Sci.* **23**:1157-1176.

Moore, D.G., and M.R. Scott. 1986. Behavior of ^{226}Ra in the Mississippi River mixing zone. *J. Geophys. Res.* **91(C12)**:14,317-14,329.

Moseley, F. N. 1966. Biology of red snapper, *Lutjanus aya* Bloch, of the northwestern Gulf of Mexico. *Publ. Inst. Mar. Sci., Univ. Texas* **11**:90-101.

Murphy, M.D. and R.G. Taylor. 1991. Direct validation of ages determined for adult red drums from otolith sections. *Trans. Am. Fish. Soc.* **120**:267-269.

Nelson, R.S. and C S. Manooch III. 1982. Growth and mortality of red snappers in the west-central Atlantic Ocean and northern Gulf of Mexico. *Trans. Am. Fish. Soc.* **111**:465-475.

Nelson, R.S., C.S. Manooch, III, and D.L. Mason. 1985. Ecological effects of energy development on reef fish of the Flower Garden Banks: Reef fish bioprofiles. Final report. National Marine Fisheries Service, Beaufort Laboratory, Unpubl. MS.

Render, J.H. 1995. *The life history of red snapper, Lutjanus campechanus, and its affinity for oil and gas platforms.* Ph. D Dissertation. Louisiana State University, Baton Rouge. 76 pp.

Shirripa, M.J. and K.M. Burns. 1997. Growth estimates for three species of reef fish in the eastern Gulf of Mexico. *Bull. Mar. Sci.* **61**(**3**):581-591.

Smith. J.N., R. Nelson, and S.E. Campana. 1991. The use of Pb-210/Ra-226 and Th-228/Ra-228 disequilibria in the ageing of otoliths of marine fish. Pages 350-359 in: P. J. Kershaw, and D. S. Woodhead (eds.), *Radionuclides in the Study of Marine Processes.* Elsevier Applied Science, London.

Stewart, B.D., G.E. Fenton, D.C. Smith, and S.A. Short. 1995. Validation of otolith-increment age estimates for a deepwater fish species, the warty oreo, *Allocyttus verrucosus*, by radiometric analysis. *Mar. Biol.* **123**:29-38.

Szedlmayer, S.T. and R.L. Shipp. 1994. Movement and growth of red snapper, *Lutjanus campechanus*, from an artificial reef area in the northeastern Gulf of Mexico. *Bull. Mar. Sci.* **55**(**2-3**):887-896.

Wade, C.W. 1981. Age and growth of spotted seatrout and red snapper in Alabama. Pages 345-354 in: J. Sweeney, (ed.) Proc. Ann. Conf. S. E. Assoc. Fish and Wildl. Agencies 35.

Wilson, C.A., J.H. Render, and D.L. Nieland. [1994] Life history gaps in red snapper (*Lutjanus campechanus*), swordfish (*Xiphias gladius*), and red drum (*Sciaenops ocellatus*) in the northern Gulf of Mexico: Age distribution, growth, and some reproductive biology. Final report to U. S. Department of Commerce, Marine Fisheries Initiative (MARFIN) Program, NMFS, St. Petersburg, FL, NA17FF0383-02, 79 pp. Unpubl. MS.

Wilson, C.A., D.L. Nieland, A.L. Stanley, and A.J. Fischer. 1998. Age and size distribution of commercially harvested red snapper, *Lutjanus campechanus*, in the northern Gulf of Mexico. Final report to U. S. Department of Commerce, Marine Fisheries Initiative (MARFIN) Program, NMFS, St. Petersburg, FL, NA57FF0287, 65 pp. Unpubl. MS.

Discriminating Between Age-0 Red Snapper, *Lutjanus campechanus*, Nursery Areas in the Northern Gulf of Mexico Using Otolith Microchemistry

WILLIAM F. PATTERSON, III[1], JAMES H. COWAN, JR.[2], CHARLES A. WILSON[1], and NEAL JULIEN[3]
[1]*Coastal Fisheries Institute*
Louisiana State University
Baton Rouge, LA 70808 USA
[2]*Department of Marine Sciences*
University of South Alabama
P.O. Box 369
Dauphin Island, Alabama 36528 USA
[3]*Midwest Research Institute, Florida Division*
1470 Treeland Boulevard S.E.
Palm Bay, FL 32909 USA

ABSTRACT

Natural biogeochemical tags of age-0 red snapper, *Lutjanus campechanus*, nursery areas in the northern Gulf of Mexico were determined based on sagittal otolith microchemistry. Age-0 red snapper were collected in 1996 and 1997 from historically important nursery areas in north central, northwest and southwest areas of the United States' Gulf of Mexico. Otolith microchemistry of these fish was assayed using solution-based inductively coupled plasma-mass spectrometry (ICP-MS). In addition to Ca, four elements (Ba, Cd, Mg, Sr) were consistently detected in age-0 red snapper otolith solutions above ICP-MS limits of detection. For statistical analyses, otolith concentrations of these elements were expressed as ratios to Ca. In 1996 and 1997 there were significant differences in Ba:Ca, Cd:Ca, Mg:Ca, and Sr:Ca ratios between nursery areas. Multivariate analyses of variance, with all four element:Ca ratios as dependent variables, indicated that differences in nursery-specific elemental signatures were statistically significant in 1996 and 1997. Linear discriminant functions were computed from elemental data in each year as a tool to classify individual fish to their nursery areas of collection. In 1996, overall classification accuracy of age-0 fish to nursery area was 93%, while 1997 age-0 fish were correctly classified with an overall accuracy of 87%. Future research will focus on determining the otolith core microchemistry of adult red snapper from offshore reefs in the northern Gulf of Mexico, and on determining the nursery areas from which adults recruited based on the microchemical tags developed from age-0 red snapper otoliths.

KEY WORDS: ICP-MS, Otolith microchemistry, nursery areas

INTRODUCTION

Red Snapper are long-lived reef fish that occur in United States' waters as far north as Massachusetts, but generally are distributed from North Carolina to Florida in the Atlantic Ocean and from Florida to the Yucatan Peninsula in the Gulf of Mexico (Hoese and Moore 1977). In the northern Gulf of Mexico (hereafter Gulf), red snapper are distributed along the continental shelf out to the shelf's edge and demonstrate high affinity for vertical structure. Adults aggregate on or near coral reefs, gravel bottoms, or rock outcrops, as well as on artificial reefs, oil rigs, and ship wrecks (Moran 1988). Young red snapper spend their first year of life over the continental shelf on the shrimping grounds where they are concentrated in areas with vertical complexity, such as relic shell habitats (Moseley 1966, Szedlmayer and Howe 1998). Adult red snapper may display agonistic behavior toward young snapper (Bailey 1995), but as they grow, young fish recruit to the adult population on offshore reef structures (Moseley 1966, Szedlmayer and Howe 1998).

Red snapper are managed as a single stock in United States' waters of the Gulf. The conclusion that Gulf red snapper constitute a single stock is supported by population genetics studies that generally have reported no differences between red snapper from different geographic areas in the northern Gulf (Gold et al. 1997). Contrary to the genetic evidence, tagging studies of adult red snapper generally have shown that adult fish demonstrated high site fidelity and moved little (Beaumariage 1969, Fable, 1980, Szedlmayer and Shipp 1994). More recently, Watterson et al. (1998) and Patterson (1999) reported that red snapper demonstrated low rates of site fidelity to artificial reefs in the northern Gulf (20 - 40%/yr), and reported 17 tagged fish moved over 100 km away from their release sites. However, high rates of tag shedding reported by Patterson (1999) impede observation of long-term movement by tagged fish.

For observation of red snapper movement on longer temporal scales, a permanent tag of fish is needed. Recent studies of otolith microchemistry have shown that otolith microchemistry serves as an ideal natural biogeochemical tag of fish (Campana and Gagne 1995, Edmunds et al. 1995, Thorrold et al. 1997, 1998 a,b). Otoliths are calcium carbonate and protein structures that serve in the accustico-lateralis system of fishes. Their growth is directly related to fish growth and, traditionally, have been used as the hardpart of choice in aging studies of fishes. More recently, it has been shown that otoliths are metabolically inert once formed and incorporate minor and trace metals from surrounding water into their matrices as they accrete (Campana and Neilson 1985, Casselman 1987, Kalish 1989, Mugiya et al. 1991). Therefore, otolith microchemical analysis reveals the environmental history of fish and can be used as a natural tag (Kalish 1989, Patterson et al. 1998, Thorrold et al. 1998 a,b).

Patterson et al. (1998) reported significant differences in otolith microchemical fingerprints of age-0 red snapper collected from northern Gulf nursery areas in 1995. The purpose of the present study was to expand on this initial work by examining otolith microchemistry of age-0 red snapper collected in the northern Gulf in 1996 and 1997. Our objective was to develop natural tags of historically important red snapper nursery areas in the northern Gulf based on otolith microchemistry. Eventually, these natural tags will be used in long-term movement analysis of adult fish. If successful, this approach will allow us to address problems concerning stock mixing and stock structure in adult red snapper, as well as to determine the source of recruits to offshore reefs throughout the northern Gulf.

METHODS

Age-0 red snapper were collected from three different regions in the northern Gulf in October and November of 1996 and 1997 (Figures 1. A,B). Fish were collected using otter trawls aboard the United States' National Oceanographic and Atmospheric Administration's R/V Oregon II and the Dauphin Island Sea Lab's R/V Verril during the National Marine Fisheries Service's fall groundfish survey in each year. Fish were collected over the continental shelf in the north central Gulf off Alabama/Mississippi (NC Gulf), in the northwest Gulf off Louisiana and east Texas (NW Gulf), and in the southwest Gulf off southeast Texas (SW Gulf). Immediately following collection, fish were placed in plastic bags and frozen. Sample sizes and geographic range of sampling were increased in 1997 to better estimate region-specific otolith microchemical tags (Figures 1.A,B).

In the laboratory, fish were thawed, weighed to the nearest mg, and measured to total length (TL). Sagittae were extracted using acid-washed glass probes and acid-washed polyethylene tweezers; all materials that came in contact with extracted otoliths were acid-washed and triple-rinsed in ultrapure water (18 MΩ polished water). Extracted otoliths were scrubbed with a synthetic bristle brush, rinsed with ultrapure water, and placed in acid-leached polyethylene vials to air-dry. Further otolith cleaning and sample preparation took place in a class-100 clean room. Otoliths were cleaned with 2% ultrapure nitric acid for 10 sec, rinsed repeatedly with ultrapure water, and allowed to air-dry in cell wells. Dry otoliths were weighed to the nearest 1 x 10^{-5} g and placed in acid-leached polyethylene vials for dissolution. Otoliths were dissolved in 10% ultra-pure nitric acid at 1 ml acid per 0.2 mg otolith.

Otolith solutions were diluted 2.5 fold and analyzed for elemental composition using a Perkin/Elmer Elan 5000 inductively couple plasma mass spectrometer (ICP-MS) with an AS-90 autosampler and FIAS-400MS accessory. All analyses were performed using internal standards which were added online using the second pump on the FIAS-400MS unit. Ca and Sr were analyzed using a standard crossflow nebulizer and Scott double pass spraychamber. The FIAS accessory was used to dilute the samples 100 fold for analysis of Ca and Sr. All other analytes were determined using a Meinhard high efficiency nebulizer and cyclonic spray chamber with a 1.25 fold dilution. Due to high concentrations of Ca in otolith solutions, nickel cones of the ICP-MS were cleaned every 15-20 samples, followed by recalibration with the internal standard. In addition to otolith solutions, blank solutions were analyzed to estimate detection limits (mean + 2σ in ppm) of elements of interest.

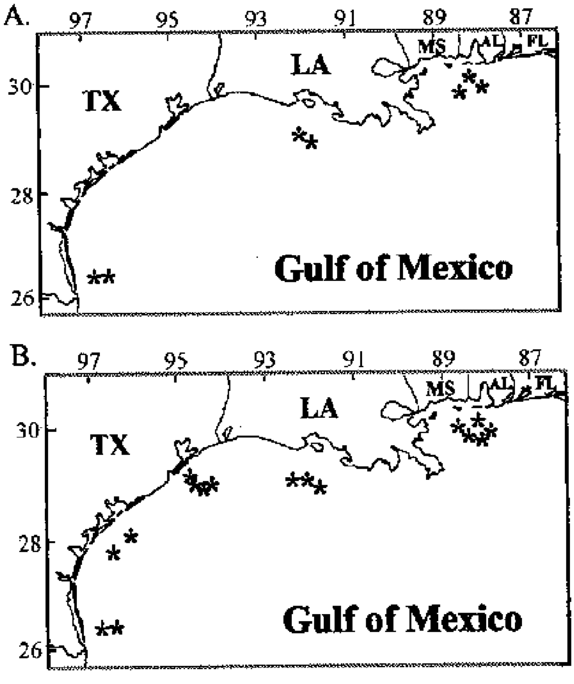


Figure 1. Maps of sampling sites for age-0 red snapper A.) in 1996 and B.) in 1997.

We were primarily interested in analyzing elements that substitute directly for Ca in otolith aragonite. Among these, Sr, Mg, Ba, Cd, Mn, Cr, Zn, Pb, and Ni were detected in otolith samples, but only Sr, Mg, Ba, Mn, and Cd concentrations were consistently above detection limits. Sr, Mg, Ba, and Mn concentrations always were well above detection limits; however, Cd concentrations were below detection limits in 26% of samples. In statistical analyses, Mn was not considered due to potential polyatomic interference from potassium oxide on mass 55.

Univariate and multivariate statistical techniques were employed to determine otolith microchemical tags unique to each nursery area. First, we tested for potential ontogenetic effects on elemental concentrations. Analyses of variance (ANOVAs) tested for differences in otolith weight and fish size between regions in 1996 and 1997. Additionally, analyses of covariance (ANCOVAs) tested for differences between regions in the relationship of otolith weight to total length in each year. Correlation analyses were performed to test if significant relationships existed between otolith weight and element:Ca ratios both within and among nursery areas. Differences in element:Ca ratios between nursery areas in each year were tested with ANOVAs. Unique otolith microchemical tags of nursery area in each year were determined using multiple analysis of variance (MANOVA) and linear discriminant function analysis (LDFA).

RESULTS

In all univariate statistical analyses, variables met the assumptions of normality and equal variances. Therefore, it was assumed that the assumptions of multivariate normality and equal variance/covariance matrices were met for multivariate statistics, although no tests of these assumptions were performed. All statistical tests were computed using SAS (SAS Institute Inc., 1990).

There was no significant difference in TL (ANOVA, $F_{2;82}$ = 1.51, p = 0.2216) or otolith weight (ANOVA, $F_{2;82}$ = 1.88, p = 0.1595) between nursery areas in 1996 (Figure 2A). There was a significant difference in TL (ANOVA, $F_{2;153}$ = 5.48, p = 0.005) and otolith weight (ANOVA, $F_{2;153}$ = 8.77, $p < 0.001$) in 1997 (Figure 2B). Because fish growth rate may affect incorporation of trace and minor elements into otoliths (Fowler 1995, Thorrold et al. 1997), and because fish sampled from the southwest Gulf were smaller than the other two regions in 1997, we tested for differences in the relationship of otolith weight and TL between areas in each year. In 1996 there was no difference in the relationship of otolith weight and TL between nursery areas (ANCOVA test for homogeneity of slopes, $F_{2;79}$ = 2.79, p = 0.0671; ANCOVA test for homogeneity of intercepts, $F_{2;79}$ = 1.69, p = 0.1991) (Figure 2A). There was also no difference in the relationship of otolith weight and TL between areas in 1997 (ANCOVA test for homogeneity of slopes, $F_{2;145} = 0.51$, $p = 0.5986$; ANCOVA test for homogeneity of intercepts, $F_{2;145} = 1.49$, $p = 0.2282$) (Figure 2B).

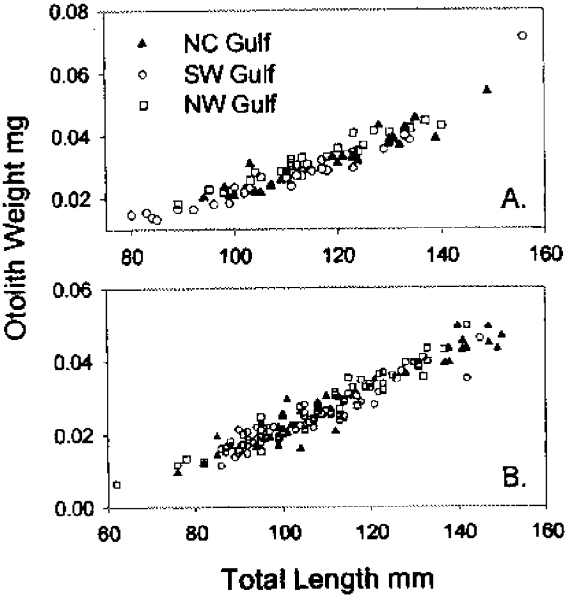


Figure 2. Relationship between otolith weight and TL for age-0 red snapper from the northern Gulf of Mexico nursery areas sampled in A.) 1996 and B.) 1997. The legend is the same for both years.

Few correlations between otolith weight and element:Ca ratios were statistically significant (Pearson's r, $p < 0.05$) within nursery areas in each year (Figure 3 A,B). For element:Ca ratios that were significant, the slopes of the relationships from different nursery areas often were opposite in direction. Therefore, although some correlations between otolith weight and element:Ca ratios pooled across nursery areas were significant, no correction for the effect of otolith weight on element:Ca ratios could be implemented via analysis of covariance (Thorrold et al. 1998b). Furthermore, because correlations between otolith weight and element:Ca ratios areas were weak and mostly non-significant when data were pooled across nursery areas, and because correlations for each element:Ca ratio within nursery areas differed in direction between areas, no systematic effect of otolith weight on element:Ca ratios was perceived (Figure 3).

In 1996, Ba:Ca ratios were statistically significant between nursery areas (ANOVA, $F_{2;82} = 3.91$, $p = 0.0239$), and Cd:Ca, Mg:Ca, and Sr:Ca, ratios were all highly significant ($p < 0.001$) between nursery areas (Figure 4). All four element:Ca ratios were highly significant between nursery areas in 1997 (Fig. 4). MANOVAs with Ba:Ca, Cd:Ca, Mg:Ca, and Sr:Ca ratios as dependent variables were significant in 1996 (Pillai's Trace, $F_{8;160} = 25.079$, $p < 0.001$) and in 1997 (Pillai's Trace, $F_{8;302} = 43.936$, $p < 0.001$). Canonical discriminant function analysis was employed as a data reduction devise to aid visualization of multivariate differences among nursery areas in each year (Figure 5 A,B). In each year, the first canonical variate accounted for approximately 75% of the discrimination between nursery areas.

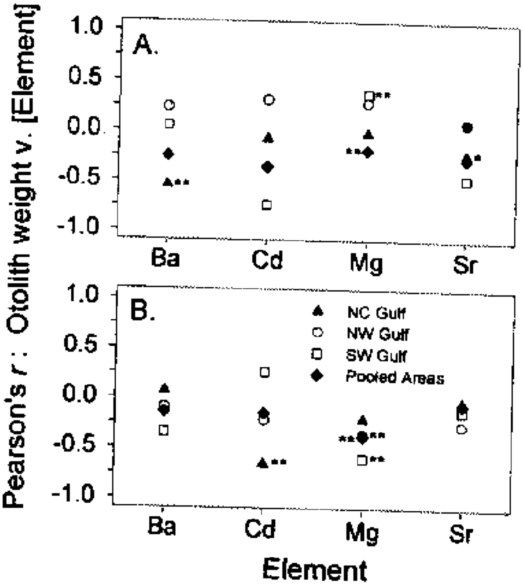


Figure 3. Correlation coefficients between element concentrations (ppm) in otoliths and otolith weight for fish sampled A.) in 1996 and B.) in 1997. One asterisk denotes $p < 0.05$ and two asterisks denote $p < 0.01$. The legend is the same for both years.

Linear discriminant function analyses computed with Ba:Ca, Cd:Ca, Mg:Ca, and Sr:Ca ratios as response variables and nursery area as the classification variable yielded clear discrimination between nursery areas in 1996 and 1997. Overall, the percentage of individual fish correctly classified to nursery area using the cross-validation algorithm in SAS was 93% in 1996 and 87% in 1997 (Tables 1 and 2) (SAS Institute Inc., 1985).

Table 1. Results of linear discriminant function analysis for classifying age-0 red snapper to nursery area in 1996 based on elemental signatures in otoliths. Bold numbers represent correct classification of individual fish to nursery area.

Area of Assignment	Area of Capture % (n)		
	NC Gulf	**NW Gulf**	**SW Gulf**
NC Gulf	**96.6 (28)**	10.0 (3)	0.0 (0)
NW Gulf	3.4 (1)	**86.7 (26)**	3.8 (1)
SW Gulf	0.0 (0)	3.3 (1)	**96.1 (25)**
Total	29	30	26
Error Rate	0.034	0.133	0.038

Table 2. Results of linear discriminant function analysis for classifying age-0 red snapper to nursery area in 1997 based on elemental signatures in otoliths. Bold numbers represent correct classification of individual fish to nursery area.

Area of Assignment	Area of Capture % (n)		
	NC Gulf	**NW Gulf**	**SW Gulf**
NC Gulf	**93.9(46)**	8.9 (5)	0.0 (0)
NW Gulf	6.1(3)	**85.7 (48)**	19.6 (9)
SW Gulf	0.0 (0)	5.4 (3)	**80.4 (37)**
Total	49	56	46
Error Rate	0.061	0.143	0.196

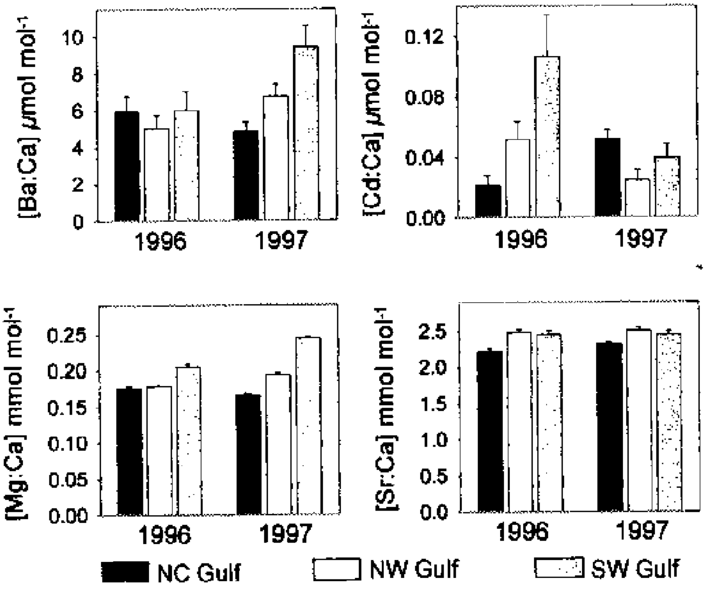


Figure 4. Plots of mean Element:Ca ratios in age-0 red snapper otoliths sampled in 1996 and 1987. Error bars are S. E. of the mean.

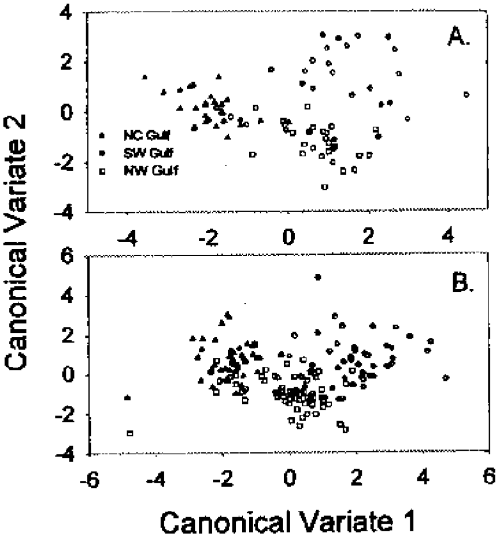


Figure 5. Scatterplots of first and second canonical variates from canonical discriminant function analysis of elemental signatures of age-0 red snapper otoliths for A.) 1996 and B.) 1997. The legend is the same for both plots.

DISCUSSION

Variability in elemental signatures was significant between nursery areas in 1996 and 1997, and differences in elemental signatures among nursery areas in both years were sufficient to allow for accurate classification of age-0 fish to northern Gulf nursery areas. There was also significant variability of individual element:Ca ratios within nursery areas between years. Statistical tests of elemental signatures were not performed between years, however, because of differences in sampling in 1996 and 1997. In 1998 and 1999, fish have been sampled from northern Gulf nursery areas similarly to those sampled in 1997. When elemental signatures of these fish are available, statistical tests of the temporal stability of nursery-specific elemental signatures will be made.

Patterson et al. (1998) reported that the relationship between otolith weight (sagittae) and TL of age-0 red snapper sampled from northern Gulf of Mexico nursery areas in 1995 did not differ among nursery areas. We have shown here that this relationship also did not differ among nursery areas for age-0 snapper sampled in 1996 and 1997. Fish sampled from the SW Gulf were smaller than the other two areas in 1997, but consistency in the relationship between otolith weight and TL among nursery areas indicates that fish from different nursery areas were growing at similar rates (Szedlmayer, 1998; Szedlmayer and Conti, 1999). Therefore, the smaller size of fish sampled from the SW Gulf in 1997 suggests that on average these fish were slightly younger than fish from the other two areas, which may partially explain some of the differences in elemental signatures between areas in 1997. However, differences in elemental signatures between nursery areas in 1997 were similar to differences in 1996 when there was no difference in mean size of fish.

One area in which we seek to improve is in our ability to quantify elements present in otoliths at trace levels. Cd was an important element in both 1996 and 1997 for discriminating between nursery areas, but concentrations of Cd in otolith solutions were often close to or below limits of detection. Among element:Ca ratios, Cd:Ca ratios showed the highest variability both within and among nursery areas, especially in 1996 when the concentrations of Cd in otolith solutions of many fish sampled from the NC Gulf were low. We feel it is not appropriate to rely on estimates of elemental concentrations that are below estimated detection limits for discrimination between nursery areas; however, Cd was included in statistical analyses here because Cd levels in most samples were above its detection limit. When LDFAs were performed without Cd:Ca ratio included as a dependent variable, overall classification accuracies were 87% in 1996 and 80% in 1997.

While the concentrations of Cd in otolith solutions were near or just above its detection limit, concentrations of Pb, Zn, Ni, and Cr were below detection limits in most samples. More reliable estimates of Cd concentrations will be attained by decreasing the dilution factor of otolith solutions, and by doing so it is likely that we also will be able to quantify other elements present at trace levels. These analyses are currently being performed, but results were not available at the time of this presentation.

The initial results of this study suggest that elemental signatures in age-0 red snapper otoliths may provide ideal natural tags of red snapper nursery areas. Future directions of our work involve developing analytical protocols which will allow us to better estimate elements that may be present in otoliths at trace levels, and to begin to examine the microchemistry of adult snapper otolith cores. We hope to be able to discriminate between age-0 red snapper nursery areas routinely with greater than 90% accuracy. Eventually, our aim is to use natural tags of red snapper nursery areas based on otolith chemistry in studies of movement of adult snapper. If successful, our approach will allow us to address many aspects of red snapper population ecology that are presently allusive. For example, in the future we hope to be able to determine the source of adult fish on offshore reefs in different regions of the northern Gulf and to estimate trans-basin movement and stock mixing rates.

LITERATURE CITED

Bailey, H.K., IV. 1995. Potential Interactive Effects of Habitat Complexity and Sub-adults on Young of the Year Red Snapper (*Lutjanus campechanus*) Behavior. M.S. Thesis. University of South Alabama. 55 pp.

Beaumariage, D. S. 1969. Returns from the 1965 Schlitz tagging program including a cumulative analysis of previous results. FL Department of Natural Resources Technical Series No. 59:1-38.

Campana, S.E. and J.D. Neilson. 1985. Microstructure of fish otoliths. Canadian Journal of Fisheries and Aquatic Sciences **42**:1014-1032.

Campana, S.E. and J.A. Gagne. 1995. Cod stock discrimination using ICPMS elemental assays of otoliths: Pages 671-692 in: D.H. Secor, J.M. Dean and S.E. Campana (eds.) *Recent Developments in Fish Otolith Research.* University of South Carolina Press, Columbia, SC.

Casselman, J.M. 1987. Determination of age and growth. p. 209-242 in: A.H. Weatherley and H.S. Gill (eds.) The Biology of Fish Growth. Academic Press, New York.

Fable, W.A., Jr. 1980. Tagging studies of red snapper (*Lutjanus campechanus*) and vermilion snapper (*Rhomboplites aurorubens*) off the south Texas coast. Contributions in Marine Science **23**:115-121.

Edmonds, J.S., N. Caputi, M.J. Moran, W.J. Fletcher, and M. Morita. 1995. Population discrimination by variation in concentrations of minor and trace elements in sagittae of two western Australian teleosts. Pages 655-670 in: D.H. Secor, J.M. Dean and S.E. Campana (eds.) *Recent Developments in*

Fish Otolith Research. University of South Carolina Press, Columbia, SC.

Gold, J.R., F. Sun, and L.R. Richardson. 1997. Population structure of red snapper from the Gulf of Mexico as inferred from analysis of mitochondrial DNA. Transactions of the American Fisheries Society **123**:386-396.

Hoese, H.D. and R.H. Moore. 1977. *Fishes of the Gulf of Mexico.* Texas A&M Press. College Station, TX. 327 p.

Kalish, J.M. 1989. Otolith microchemistry: validation of the effects of physiology, age and environment of otolith composition. Journal of Experimental Marine Biology and Ecology **132**:151-178.

Moran, D. [1988]. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Gulf of Mexico)--red snapper. U.S. Fish and Wildlife Service Biological Report 82(11.83). U.S. Army Corps of Engineers, TR EL-84-4. 19 pp.

Moseley, F.N. 1966. Biology of the red snapper, *Lutjanus aya* Bloch, of the northwestern Gulf of Mexico. Publications of the Institute of Marine Science, TX **11**:90-101.

Mugiya, Y., T. Hakomori, and K. Hatsutori. 1991. Trace element incorporation into otoliths and scales in the goldfish, *Carassius auratus*. Comparative Biochemistry and Physiology **99C**:327-331.

Patterson. W.F., III. 1999. Aspects of the population ecology of red snapper, *Lutjanus campechanus*, from an artificial reef area off Alabama. PhD. Dissertation. University of South Alabama. 164 pp.

Patterson, W.F., III, J.H. Cowan, Jr., Y.B. Graham, and W.B. Lyons. 1998. Otolith microchemical fingerprints of age-0 red snapper, *Lutjanus campechanus*, from the northern Gulf of Mexico. Gulf of Mexico Science **16**:83-91.

SAS Institute, Inc. 1985. *Procedures Guide Version 6.03.* SAS Institute Inc., Box 8000, Cary, North Carolina 27511-8000.

Szdelmayer, S.T. 1998. Comparison of growth rate and formation of otolith increments in age-0 red snapper. Journal of Fish Biology **53**:58-65.

Szdelmayer, S.T. and J. Conti. 1999. Nursery habitats, growth rates, and seasonality of age-0 red snapper, *Lutjanus campechanus*, in the northeast Gulf of Mexico. *U.S. Fishery Bulletin* **97**:626-635.

Szedlmayer, S.T. and J.C. Howe. 1997. Substrate preference in age-0 red snapper, *Lutjanus campechanus*. Environmental Biology of Fishes **50**:203-207.

Szedlmayer, S.T. and R.L. Shipp. 1994. Movement and growth of red snapper, *Lutjanus campechanus*, from an artificial reef area in the northeastern Gulf of Mexico. *Bulletin of Marine Science* **55**:887- 896.

Thorrold, S.R., C.M. Jones, and S.E. Campana. 1997. Response of otolith microchemistry to environmental variations experienced by larval and juvenile Atlantic croaker (*Micropognias undulatus*). *Limnology and Oceanography* **42**:102-111.

Thorrold, S.R., C.M. Jones, S.E. Campana, J.W. McLaren, and J.W.H. Lam. 1998. Trace element signatures in otoliths record natal river of juvenile American shad (*Alosa sapidissima*). Limnology and Oceanography **43**:1829-1835.

Thorrold, S.R., C.M. Jones, P.K. Swart, and T.E. Targett. 1998. Accurate classification of juvenile weakfish *Cynoscion regalis* to estuarine nursery areas based on chemical signatures in otoliths. *Marine Ecology Progress Series* **173**:253-265.

Watterson, J.C., W.F. Patterson, III, R.L. Shipp, and J.H. Cowan, Jr. 1998. Movement of red snapper, *Lutjanus campechanus*, in the north central Gulf of Mexico: potential influence of hurricanes. *Gulf of Mexico Science* **16**:92-104.

Elemental Signatures of Red Drum (*Sciaenops ocellatus*) Otoliths from the Gulf of Mexico and Western Atlantic

HEATHER M. PATTERSON,[1] RICHARD S. MCBRIDE[1], ROY E. CRABTREE[2], and NEAL JULIEN[3]
[1]*Florida Marine Research Institute*
Florida Fish and Wildlife Conservation Commission
100 Eighth Avenue SE
St. Petersburg, FL 33701-5095 USA
[2]*National Marine Fisheries Service, Southeast Regional Office*
9721 Executive Center Drive N
St. Petersburg, FL 33702-2434 USA
[3]*Midwest Research Institute-Florida Division*
1470 Treeland Boulevard SE
Palm Bay, FL 32909,USA

ABSTRACT

The red drum (*Sciaenops ocellatus*) is an estuarine-dependent species that supports recreational fisheries in the Gulf of Mexico and western Atlantic Ocean. In the mid-1980s, concerns about overfishing of red drum resulted in recreational catch limits and a ban on commercial fishing in Florida. Recent studies offshore of Tampa Bay, Florida, suggest that at least in this area the species is recovering. However, it is not known if this recovery is occurring throughout the entire Gulf of Mexico. Genetic studies have shown that there are weakly differentiated sub-populations in the Gulf. This could mean that a local recovery would not affect the entire region. We examined otolith chemistry as an independent measure of stock discreteness or connectivity. Otolith chemistry is a useful natural tag for examining the stock structure of fishes because it can reflect the elemental composition of the water in which the fish resides. Juvenile red drum otoliths from Texas, South Carolina, and Georgia and from three sites in Florida--Tampa Bay, Cedar Key, and Indian River-- were analyzed using solution-based inductively coupled plasma-mass spectrometry (ICP-MS). Five elements (Mg, Mn, Zn, Sr, and Ba) were routinely detected above background levels. A MANOVA (multiple analysis of variance) indicated significant differences in the otolith chemistry of the red drum collected at different sites (Wilk's Lambda = 15.80, p = 0.0001). Using a cross-validation procedure, we were able to correctly classify otoliths from the Florida sites with an accuracy rate of up to 95%. This research demonstrates that otolith chemistry may be useful in delineating the stock structure of red drum throughout their range. We plan to use this method to examine sources of recruitment to the Tampa Bay area and thereby examine the connectivity of Gulf stocks in general.

KEY WORDS: ICP-MS, otolith chemistry, *Sciaenops ocellatus*

INTRODUCTION

The red drum, *Sciaenops ocellatus*, is an estuarine-dependent sciaenid found in temperate and subtropical waters, primarily in the Gulf of Mexico (Gulf) and western Atlantic Ocean. Although the commercial fishery for red drum in the Gulf was closed during the mid-1980s because of concerns about overfishing, the species still supports important recreational fisheries throughout the Gulf and southeastern U.S. Characteristics of the species such as age, growth, reproduction, and mortality are well studied (Peters and McMichael 1987, Beckman et al. 1988, Murphy and Taylor 1990, Wilson and Nieland 1994). Recently, Murphy and Crabtree (1999) examined the age structure of adult red drum from the waters offshore of Tampa Bay, Florida. They found that reduced rates of fishing mortality resulted in a measurably abrupt increase in the survival rates of red drum beginning with the 1985 or 1986 year-class. This increase in survival rates was the result of strict fishing regulations imposed on the fishery throughout the southeastern U.S. and contributed to increased abundance of adult offshore stocks in west Florida.

Whether this recovery affected other areas of the Gulf or Atlantic is central to the purpose of this study. There is significant genetic variation (mitochondrial [mt] DNA) to suggest that two separate populations are present in Gulf and Atlantic waters (Bohlmeyer and Gold 1991, Gold and Richardson 1991, Gold et al. 1993, 1994). The geographic and oceanographic features of Florida that may preclude gene exchange between Gulf and Atlantic populations of red drum would also presumably limit increases in Gulf population size from affecting the size of Atlantic stocks of red drum. In a recent genetic study, Gold et al. (1999) were able to distinguish different groups of red drum within the Gulf. They found significant variations in the mtDNA that were consistent with an isolation-by-distance effect, wherein fish from geographically closer areas are more genetically similar than fish from more distant locations. Gold et al. (1999) noted that this genetic pattern may be due to behavioral characteristics of female red drum such as philopatry to their natal estuaries and limited coastwise movement. These processes could also restrict the effects of local increases in fish abundance in Tampa Bay to adjacent areas within the Gulf.

A method of examining stock structure that does not directly examine gene flow is otolith chemistry. Otolith chemistry of individuals will differ between areas within the Gulf where dispersal and movements of fish are limited and between areas of different water chemistry (e.g., Edmunds et al. 1989, 1991, 1992, Campana and Gagne 1995, Thorrold et al. 1997, 1998a, 1998b, Patterson et al. 1999). The elements that are deposited in the aragonite matrix of the otolith appear to be derived mainly from water (Farrell and Campana 1996) and will remain unaltered because the otolith is metabolically inert (Campana and

Neilson 1985). Thus, the chemistry of the otolith will reflect, to some degree, the chemistry of the water in which the fish resides.

The objective of this study was to examine the elemental signatures of red drum otoliths from the Gulf and western Atlantic to determine if differences in otolith chemistry could be detected and whether such differences were consistent with recent genetic data. The data presented here are part of a larger study to be published separately; our ultimate objective is to identify the source of recruitment of red drum to Tampa Bay.

MATERIALS AND METHODS

Young-of-the-year and juvenile red drum otoliths were collected in mid-to-late 1998 through early 1999 from sites in Texas, Georgia, and South Carolina and from three sites in Florida: Tampa Bay, Cedar Key, and Indian River (Figure 1; Table 1). The red drum otoliths collected at the Florida sites were collected by staff of the Florida Marine Research Institute (FMRI) field labs specifically for this study, whereas the otoliths from sites in the three other states had been previously collected by other researchers who donated them to the study. The red drum from Florida were collected for specific size, spatial, and temporal parameters, and the handling of these otoliths was controlled from the time the otoliths were removed from the fish (e.g. teflon forceps used to remove otoliths, and otoliths rinsed in Milli-Q water). Otoliths from other states were not collected for these specific parameters or handled in a controlled manner prior to their donation.

Table 1. Summary information of the red drum, *Sciaenops ocellatus,* used in this study including collection site, collection date (month/year), the number of otoliths analyzed (n), and minimum and maximum total length (TL [mm], mean).

Location	Date	n	Min-Max TL (mean)
Cedar Key	1/99-2/99	20	69 - 166 (120.4)
Indian River	2/99	20	46 - 52 (49.0)
Tampa Bay	2/99-3/99	20	66 - 147 (112.7)
Georgia	8/98-12/98	20	310 - 364 (350.6)
South Carolina	10/98-12/98	20	338 - 447 (405.2)
Texas	8/98-11/98	20	205 - 307 (280.5)

The weight of whole otoliths varied between sites (3.6 - 57.3 mg), so heavier otoliths were sectioned with a Buehler Isomet low-speed saw to fit within a weight range of 9.0 - 9.9 mg. Otoliths from Indian River were not sectioned because they already fell below this range (3.6 - 4.5 mg). Sections of whole otoliths were cut approximately 1 - 1.5 mm thick around the core and then polished evenly on all sides using grit size 220 lapping paper until the sections were in the desired weight range. Before they were acid-washed, the polished sections and Indian River otoliths were cleaned ultrasonically for 15 minutes and triple rinsed in Milli-Q water. We acid-washed the otoliths by dipping them in 1% trace-metal-grade nitric acid for 15 seconds and triple rinsing them with Milli-Q water. The otoliths were allowed to dry under a Class 100 laminar flow hood for 24 hours and then were weighed to the nearest 10 µg. The otolith sections were then dissolved in 0.5 or 0.25 ml (depending on otolith weight) of 70% trace-metal-grade nitric acid, and Milli-Q water was added in a ratio equal to 9x the acid volume. Blanks were similarly prepared so that we could calculate detection limits and for blank corrections.

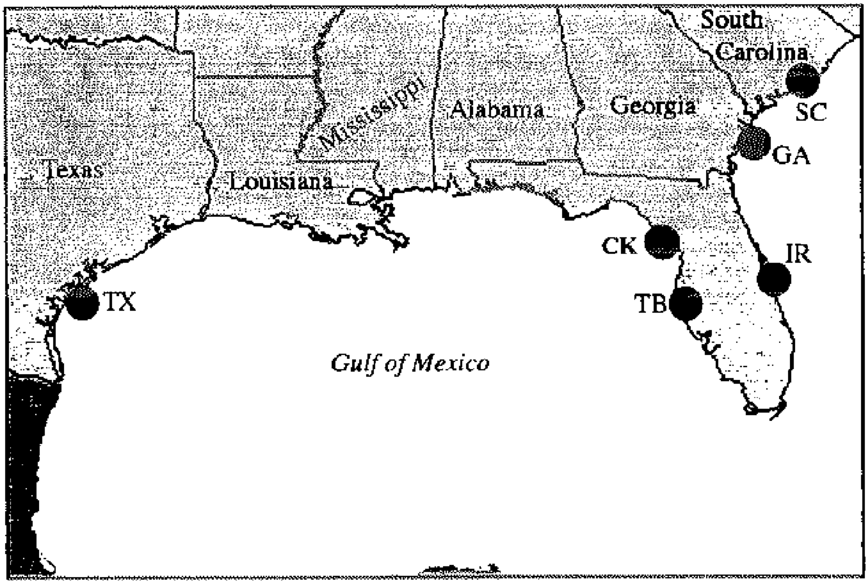


Figure 1. Map showing sampling locations including Texas (TX), Georgia (GA), and South Carolina (SC), and three Florida sites including Indian River (IR), Cedar Key (CK), and Tampa Bay (TB).

Elemental concentrations of the otoliths were determined using a Perkin-Elmer Elan 5000 ICP-MS. Preliminary tests had indicated that several elements were detectable and suitable for ICP-MS analysis (Ba, Ca, Mg, Mn, Sr, and Zn).

Samples were blocked so that one from each location was sampled in turn, with the order within each block randomized. Internal standards for each element were used and referenced against ^{45}Sc, ^{72}Ge, ^{89}Y, and ^{159}Tb. In order to avoid the problem of instrument drift due to build up of Ca on the cones, the cones were conditioned with a solution of 1000 mg/L Ca and 10 µg/L Ba. This solution was analyzed five times and the Ba signal monitored for stability. This process allowed an equilibrium between evaporation and deposition on the cones to be reached, so the cones were conditioned to the matrix, which allowed for a consistent signal. Calcium and Sr were analyzed with a 10x online dilution using a crossflow nebulizer and a double-pass spray chamber. The remaining elements were analyzed undiluted with a Meinhard high-efficiency nebulizer and a baffled cyclonic spray chamber. The limits of detection for each element were calculated from the prepared blanks as 3σ plus the mean blank value and were as follows (in µg/g): Ca 280, Sr 12.1, Ba 0.04, Mg 0.32, Mn 0.04, and Zn 0.07. The limits of detection were well below the observed values for all elements.

Elemental data were converted to molar concentrations and expressed as ratios to Ca because the elements we were working with substitute for Ca in the aragonite matrix of the otolith. Statistical analysis was carried out using both univariate and multivariate approaches. To test for overall significance at all the sites and including all five elemental ratios (Ba/Ca, Mg/Ca, Mg/Ca, Sr/Ca, and Zn/Ca), we performed a MANOVA (multivariate analysis of variance) using the GLM and MANOVA statements in SAS (Littell et al. 1991). This was followed by individual ANOVAs for each of the five elemental ratios to determine which elements were significantly different and by unplanned contrast comparisons (least significant difference method).

To visualize differences between the sampling sites in canonical space, we used the CANDISC function in SAS (SAS 1989) to perform a canonical discriminant analysis. This procedure derives linear combinations of the variables that summarize between-class (i.e., sampling site) variation. Pearson correlation coefficients determined which elemental ratios loaded significantly on each axis. Finally, the DISCRIM procedure (SAS 1989) was used to determine classification accuracy. We used the cross-validation option, which removes each sample in turn from the data set and then classifies it based on the remaining observations.

RESULTS

The MANOVA (Wilks' Lambda = 15.8006; p = 0.0001) showed that there was an overall significant difference between the sites based on otolith chemistry. This difference was examined more closely in the individual ANOVAs of each elemental ratio (Figure 2). Three of the elemental ratios we analyzed were significantly different between sites: Ba/Ca, Mn/Ca, and Sr/Ca.

Elemental ratios of Mg/Ca and Zn/Ca were not significantly different between sites based on univariate analysis, but were left in the remaining multivariate analyses described below and were found to contribute significantly.

The first three canonical axes accounted for 70%, 53%, and 40% of the variation between sites. The only elemental ratio to load significantly on canonical axis 1 was Sr/Ca ($P = 0.0001$), which loaded in a positive direction. Both Ba/Ca ($p = 0.0001$) and Zn/Ca ($p = 0.0012$) loaded significantly in a positive direction on canonical axis 2, and Mn/Ca ($p = 0.0001$) loaded significantly in a negative direction. Lastly, Ba/Ca ($p = 0.0001$) and Mn/Ca $p = 0.0001$) both loaded positively on canonical axis 3, and Mg/Ca ($p = 0.0006$) loaded in a negative direction on this axis. A plot of the first two axes depicts the separation of the sites (Figure 3). The three Florida sites form distinct groups, whereas the other sites are less clearly defined.

The results of the cross-validation procedure show that the classification accuracy of the otoliths from the three Florida sites ranged from 90% to 95%. South Carolina and Texas otoliths had marginal recoveries with 45% and 55%, respectively, while Georgia had a classification accuracy of only 25%. The South Carolina and Georgia otoliths were primarily misclassified with each other, whereas Texas otoliths were scattered throughout the classification (Table 2).

Table 2. Results of the cross-validation procedure showing the classification accuracy for each group based on the elemental signatures of otoliths. The locations are Tampa Bay, FL (TB), Cedar Key, FL (CK), Georgia (GA), Indian River, FL (IR), South Carolina (SC), and Texas (TX). Both the number and percentage of classified otoliths are shown. The number of samples analyzed for each group was 20.

Site	TB	CK	GA	IR	SC	TX
TB	19	0	1	0	0	0
	95%	0%	5%	0%	0%	0%
CK	1	19	0	0	0	0
	5%	95%	0%	0%	0%	0%
GA	0	0	5	1	12	2
	0%	0%	25%	5%	60%	10%
IR	0	1	1	18	0	0
	0%	5%	5%	90%	0%	0%
SC	0	2	8	0	9	1
	0%	10%	40%	0%	45%	5%
TX	2	1	2	2	2	11
	10%	5%	10%	10%	10%	55%

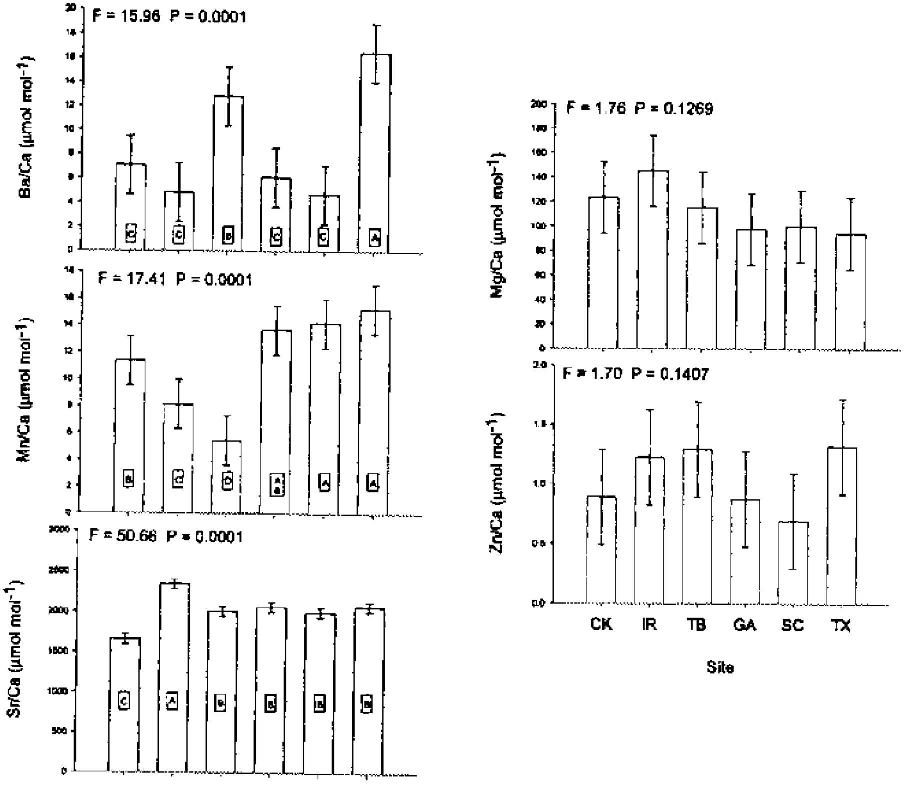


Figure 2. Mean concentrations expressed as a molar ratio to Ca (± SE) of the five elements analyzed in *Sciaenops ocellatus* otoliths (see Figure 1 for sample locations). The F and P values were determined by one-way ANOVAs and are indicated on each graph. The least significant difference contrast groupings are indicated as letters (i.e. A, B, C, D) where overall differences were significant ($p < 0.05$).

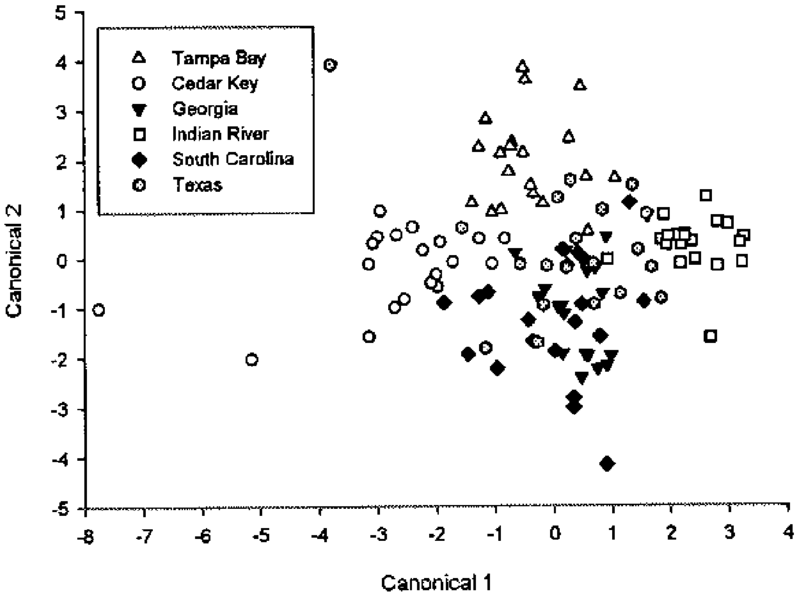


Figure 3. Canonical discrimination of elemental ratios for *Sciaenops ocellatus* otoliths collected from five sampling sites in the Gulf of Mexico and Atlantic Ocean. Only the first two canonical axes are shown here. Tampa Bay is indicated by open triangles, Cedar Key open circles, Indian River open squares, Georgia filled inverted triangles, South Carolina filled diamonds, and Texas gray hexagons.

DISCUSSION

The elemental concentrations of the red drum otoliths we analyzed varied significantly by site. Otoliths from the three sites in Florida were correctly classified more often than otoliths from any other site and formed distinct multivariate groups. Based on the postulations of Gold et al. (1999) concerning the causes of an isolation-by-distance genetic structure for red drum in the Gulf (e.g., limited coastwise movement and female philopatry to natal estuaries), we expected significant differences between the chemistry of the otoliths from the different sites if significant variation in water chemistry existed. Indeed, we were able to find differences between sites on a finer scale than genetic studies have found. Gold et al. (1999) found a geographic neighborhood size relative to genetic migration from an estuary to be 500 to 600 km, but we were able to find a significant difference in otolith chemistry between Tampa Bay and Cedar Key, which are approximately 150 km apart.

Regarding sampling procedures, only Florida otoliths were collected specifically for this study and were collected as young-of-the-year in discrete sampling periods. The South Carolina, Georgia, and Texas samples introduced variability into the analysis that appears to have confounded separation of stocks. The otoliths we were sent from South Carolina and Georgia were collected in several locations, although in each of these states, the sites appear to have been relatively close to each other. It has been demonstrated that otoliths collected only a small distance from each other (i.e., sites within the same river) can differ chemically (Thorrold et al. 1998a), and Texas otoliths, which were collected at sites that spanned the entire coast of that state, were a poor fit to the cross-validation procedure. We concluded that the large spatial distribution of the Texas sites where the otoliths were collected (i.e., Port Aransas, East Matagonda Bay, Sabine Pass, Corpus Christi Bay, Galveston Bay, and Upper/Lower Laguna Madre) introduced variation that reflects very different bay systems within this state.

In this study we have demonstrated that red drum from different sites can be distinguished based on their otolith chemistry, and the results are largely consistent with the genetic models of red drum. Results that deviated from this conclusion were related to sampling and handling methods. Our continuing research with red drum otoliths will focus on whether individuals or consecutive year-classes of fish residing in the same estuary maintain a consistent elemental signature over time. The ultimate objective of this research is to use laser ablation ICP-MS to look at the core region of otoliths from adult red drum from the waters offshore of Tampa Bay and see if their elemental signatures can be used to determine the origin of these fish. Although genetic models have provided a framework for our study, genetic methods might not be sufficient for identifying the source of recruitment to Tampa Bay because only limited genetic exchange is necessary to produce genetic homogeneity (Allendorf and Phelps 1981). If these adult red drum are indeed from waters near Tampa Bay, as research has suggested (Gold et al. 1999), then recovery of the population in this area may have little impact on other sub-populations in the Gulf of Mexico. Such information will be important in deciding future management strategies for this economically important species.

ACKNOWLEDGEMENTS

We would like to thank the fisheries-independent monitoring staff of the Indian River, Cedar Key, and St. Petersburg FMRI laboratories for collecting samples for us. We would also like to thank Bob Colura, Charlie Wenner, and Spud Woodward for donating red drum otoliths to this study. Ron Taylor, Luiz

Barbieri, Jim Quinn, and Judy Leiby made helpful comments that improved the manuscript. Johan Schijf generously allowed us access to his clean room, and Chris Anderson provided assistance with the map. This research was supported under funding from the Department of the Interior, U.S. Fish and Wildlife Service, Federal Aid for Sportfish Restoration, Project Number F-59.

LITERATURE CITED

Allendorf, F.W. and S.R. Phelps. 1981. Use of allelic frequencies to describe population structure. *Can. J. Fish. Aquat. Sciences.* 38:1507-1514.

Beckman, D.W., C.A. Wilson and A. L. Stanley. 1988. Age and growth of the red drum, *Sciaenops ocellatus*, from offshore waters of the northern Gulf of Mexico. ***Fish. Bull.*** **87**:17-28.

Bohlmeyer, D.A. and J.R. Gold. 1991. Genetic studies in marine fishes. II. A protein electrophoretic analysis of population structure in the red drum *Sciaenops ocellatus*. ***Mar. Biol.*** **108**:197-206.

Campana, S.E. and J.A. Gagne. 1995. Cod stock discrimination using ICPMS elemental assays of otoliths. Pages 671-691 in: D.H. Secor, J.M. Dean and S.E. Campana, (eds.) ***New developments in fish otolith research.*** Univ. of South Carolina.

Campana, S.E. and J.D. Neilson. 1985. Microstructure of fish otoliths. *Can. J. Fish. Aquat. Sci.* **42**:1014-1032.

Edmunds, J.S., N. Caputi and M. Morita. 1991. Stock discrimination by trace-element analysis of otoliths of orange roughy (*Hoplostethus atlanticus*), a deep-water marine teleost. *Aust. J. Mar. Freshwater Res.* **42**:1014-1032.

Edmunds, J.S., R.C.J. Lenanton, N. Caputi and M. Morita. 1992. Trace elements in the otoliths of yellow-eye mullet (*Aldrichetta forsteri*) as an aid to stock identification. ***Fish. Res.*** **13**:39-51.

Edmunds, J.S., M. J. Moran, N. Caputi and M. Morita. 1989. Trace element analysis of sagittae as an aid to stock identification: Pink snapper (*Chrysophrys auratus*) in Western Australian waters. *Can. J. Fish. Aquat. Sci.* **46**:50-54.

Farrell, J. and S.E. Campana. 1996. Regulation of calcium and strontium deposition on the otoliths of juvenile tilapia, *Oreochromis niloticus*. *Comp. Biochem. Physiol.* **115**:103-109.

Gold, J.R. and L.R. Richardson. 1991. Genetic studies in marine fishes. IV. An analysis of population structure in the red drum (*Sciaenops ocellatus*) using mitochondrial DNA. ***Fish. Res.*** **12**:213-241.

Gold, J.R., L.R. Richardson and T.F. Turner. 1999. Temporal stability and spatial divergence of mitochondrial DNA haplotype frequencies in the red drum (*Sciaenops ocellatus*) from coastal regions of the western Atlantic Ocean and Gulf of Mexico. *Mar. Biol.* **133**:593-602.

Gold, J.R., L.R. Richardson, C. Furman and T.L. King. 1993. Mitochondrial DNA differentiation and population structure in red drum (*Sciaenops ocellatus*) from the Gulf of Mexico and Atlantic Ocean. *Mar. Biol.* **116**:175-185.

Gold, J.R., T.L. King, L.R. Richardson, D.A. Bohlmeyer and G.C. Matlock. 1994. Allozyme differentiation within and between red drum (*Sciaenops ocellatus*) from the Gulf of Mexico and Atlantic Ocean. *J. Fish. Biol.* **44**:567-590.

Littell, R.C., R.J. Freund and P.C. Spector. 1991. *SAS Systems for Linear Models*, 3rd ed. SAS Institute Inc., Cary, NC. 329 pp.

Murphy, M.D. and R.E. Crabtree. 1999. Age structure of offshore red drum populations in nearshore waters off west-central Florida. MARFIN Final Report NA57FF0296.

Murphy, M.D. and R.G. Taylor. 1990. Reproduction, growth, and mortality of red drum *Sciaenops ocellatus* in Florida waters. *Fish. Bull.* **88**:531-542.

Patterson, H.M., S.R. Thorrold and J.M. Shenker. 1999. Analysis of otolith chemistry in Nassau grouper (*Epinephelus striatus*) from the Bahamas and Belize using solution-based ICP-MS. *Coral Reefs* **18**(2):171-178.

Peters, K.M. and R.H. McMichael Jr. 1987. Early life history of *Sciaenops ocellatus* (Pisces: Sciaenidae) in Tampa Bay, Florida. *Estuaries* **10**(2):92-107.

SAS Institute Inc. 1989. *SAS/STAT User's Guide*, version 6, 4th ed., Cary, NC. 943 pp.

Thorrold, S.R., C.M. Jones and S.E. Campana. 1997. Response of otolith microchemistry to environmental variations experienced by larval and juvenile Atlantic croaker (*Micropogonias undulatus*). *Limnol. Oceanogr.* **42**(1):102-111.

Thorrold, S.R., C.M. Jones, S.E. Campana, J.W. Mclaren and J.W.H. Lam. 1998a. Trace element signatures in otoliths record natal river of juvenile American shad (*Alosa sapidissima*). *Limnol. Oceanogr.* **43**(8):1826-1835.

Thorrold, S.R., C.M. Jones, P.K. Swart and T.E. Targett. 1998b. Accurate classification of juvenile weakfish *Cynoscion regalis* to estuarine nursery areas based on chemical signatures in otoliths. *Mar. Ecol. Prog. Ser.* **173**:253-265.

Wilson, C.A. and D.L. Nieland. 1994. Reproductive biology of the red drum, *Sciaenops ocellatus*, from the neritic waters of the northern Gulf of Mexico. *Fish. Bull.* **92**:841-850.

Preliminary Observations on the Sexual Development of Hogfish, *Lachnolaimus maximus* (Pisces: Labridae)

RICHARD MCBRIDE[1], MICHAEL JOHNSON[2], LEW BULLOCK[3], and
FREDRIK STENGARD[3]
[1]*Florida Marine Research Institute (FMRI)*
Florida Fish & Wildlife Conservation Commission (FWC)
100 Eighth Avenue SE
St. Petersburg, FL 33701 USA
[2]*FMRI, FWC; South Florida Regional Lab; Suite 119*
2796 Overseas Hwy.
Marathon, FL 33050 USA
[3]*FMRI, FWC*
100 Eighth Avenue SE
St. Petersburg, FL 33701 USA

ABSTRACT

Hogfish are the largest and most valuable wrasse species in Florida's waters. Recreational and commercial fishers combined caught an annual average of 400,000 pounds of hogfish during the last 12 years. Declining landings and catch rates, together with rapidly increasing prices per pound, suggest that problems may persist for hogfish populations despite fishery regulations enacted in 1994. Hogfish are hermaphroditic (protogynous) and there is concern that the minimum size limit of 12 inches may be too small to allow females to change to males in heavily fished areas. We are currently reviewing data from the Florida Marine Fisheries Information System, the Marine Recreational Fishery Statistics Survey, and special collections of hogfish. Fish were collected with spears, trawls, and traps. Otoliths were removed for age determination and gonads were fixed for characterization of reproductive biology. Here, we present our methods of reproductive staging through the use of histological preparations. We also present preliminary results on the associations between morphometric and pigmentation characters and reproductive development, and we compare the sizes at first maturity and sexual transformation of fish collected in the northeastern Gulf of Mexico with those of fish from the Florida Keys.

KEY WORDS: Hermaphroditism, protogyny, sexual dichromatism

INTRODUCTION

Hogfish life-history research has been conducted in the Florida Keys (Davis, 1976), Puerto Rico (Colin 1982), and Cuba (Claro et al. 1989). However, the pattern of sexual development of hogfish has received only limited treatment, particularly in comparison to other wrasse species. Wrasses (Labridae) and the related parrotfishes (Scaridae) have complex and diverse mating systems, patterns of sex change, and sexual dichromatism (Robertson and Warner 1978, Warner and Robertson 1978). In many of these species, for example, size at sexual transformation can be explained by adaptive processes of sexual selection.

The purpose of this study is to describe the sexual development of hogfish in a manner comparable to previous studies of the sexual development of other wrasses (e.g., Warner and Robertson 1978). While such information is of interest to evolutionary ecologists, the results are of more than theoretical value. Hogfish are the largest Caribbean wrasse (Randall 1968), and they are harvested as a foodfish throughout Florida's coastal waters, particularly in regions with reef habitats, such as the Florida Keys or the Gulf of Mexico Middle Ground. Both recreational and commercial fishers use spears and hook-and-line gear to capture hogfish. This species is known for remaining in the vicinity of diver activity, a behavior that makes hogfish particularly vulnerable to spearfishing. Because anecdotal information for the Florida Keys indicates that average hogfish size has decreased in recent years (DeMaria 1996), this study will also examine if life-history parameters such as size at maturity have changed relative to previous reports of hogfish life history.

Hogfish are known to be protogynous hermaphrodites, but previous studies had emphasized macroscopic or population-level sexual characteristics, which can occasionally be misleading (Sadovy and Shapiro 1987). In this study, we are examining microscopic characters from histological preparations of gonad tissue. We are specifically interested in sexual characteristics of hogfish such as size at first maturity and size at sexual transformation, and in evidence for monandric versus diandric protogyny. Diandry refers to a pattern in which both primary and secondary males are present in a population, whereas monandry refers to a pattern in which all males in a population have transformed from a female stage.

Histological preparations were made from a portion of tissue excised from the mid-dorsal area of the gonad. Tissue samples were embedded in glycol methacrylate, sectioned along the transverse plane, and stained with PAS/iron-hematoxylin and counterstained with metanil yellow (Quintero-Hunter et al. 1991). Our staging scheme for gonad development was adopted from the system developed by Moe (1969) for the protogynous hermaphrodite *Epinephelus morio* (Serranidae). This system was adapted for a labrid, the protogynous hermaphrodite *Semicossyphus pulcher*, by Warner (1975). Warner had chosen to lump two male classes, but we find it useful to use all ten classes established by Moe (1969). Details of our microscopic criteria will be published elsewhere. We also measured several morphological characters, particularly those relating to head shape, and pigment characters to identify sex-specific allometry and sexual dichromatism. Preliminary results reported here are based on approximately 750 hogfish, about half of our anticipated final sample size.

Our preliminary observations of histological preparations indicate that hogfish are monandric (i.e., all fish begin as females and all males are secondary). We are still assessing evidence for whether sex change by this species is strictly postmaturational (i.e., sexual transition occurs after a functional female stage). We have confirmed several pigment patterns that are characteristic of the terminal male phase: a mask covering the snout, a caudal fin band, and a dash behind the pectoral fin. Sexual dichromatism in hogfish is particularly useful for field identification of sexes; it can be and has been used to observe hogfish harem formation and reproductive behavior (Colin 1982). We can also confirm that dramatic changes in snout morphology are associated with transformation into the terminal phase.

Because fishing pressure on hogfish has increased in the Florida Keys since the 1970s (Bohnsack et al. 1994), hogfish size at maturity could theoretically change over time in response to this fishing pressure. This research question will affect management of this species, but we do not have a final result here because the samples we have examined to date represent an uneven mix of fish sizes, months of collections, and collection localities. Based on our preliminary results, sexual transformation begins at a size (300 mm FL) similar to that reported by Davis (1976) for hogfish collected in the middle Florida Keys during 1973 - 1974. In contrast to this preliminary finding, Cowen (1990) observed that in heavily fished areas, sexual transformation occurred in smaller-than-normal sheephead, *Semicossyphus pulcher*, a temperate labrid. His results were consistent with and offered further insight into both the size-advantage model of Ghiselin (1969) and the sex-ratio model of Shapiro and Lubbock (1980). We are also interested in examining whether sex ratios for hogfish have changed in response to fishing pressure. For example, we plan to examine whether increases in fishing effort have caused harem structure to shift to smaller males and if it has, whether these males are controlling smaller and fewer females. Our final conclusions about the significance of sexual patterns in hogfish will be reported after we have reviewed all the gonadal material available to us.

ACKNOWLEDGMENTS

D. DeMaria, K. DeMaria, T. Dunmire, S. Levett, D. Murie, P. Steele, and J. Styer helped collect hogfish for this study. P. Nagle oversaw histological processing. L. Barbieri, L. French, J. Leiby, H. Patterson, J. Quinn, and P. Thurman helped improve the manuscript. Research funding was provided, in part, by a Marine Fisheries Initiative (MARFIN) Grant (No. NA87FF0422). We thank all of the above.

LITERATURE CITED

Bohnsack, J.A., D.E. Harper, and D.B. McClellan. 1994. Fisheries trends from Monroe County, Florida. *Bull. Mar. Sci.* **54**(3):982-1018.

Claro, R., A. Garcia-Cagide, and R. Fernández de Alaiza. 1989. Caracteristicas biológicas del pez perro, *Lachnolaimus maximus* (Walbaum), en el golfo de Batabanó, Cuba. *Rev. Invest. Mar.* **10**(3):239-252.

Colin, P.L. 1982. Spawning and larval development of the hogfish, *Lachnolaimus maximus* (Pisces: Labridae). *Fish. Bull., U.S.* **80**(4)853-862.

Cowen, R.K. 1990. Sex change and life history patterns of the labrid, *Semicossyphus pulcher*, across an environmental gradient. *Copeia* **1990**(3):787-795.

Davis, J.C. 1976. Biology of the hogfish, *Lachnolaimus maximus* (Walbaum), in the Florida Keys. M.S. thesis. University of Miami, Coral Gables, FL. 86 pp.

DeMaria, K. [1996]. Changes in the Florida Keys Marine Ecosystem based upon Interviews with Experienced Residents. Prepared for the Nature Conservancy and Center for Marine Conservation. 105 p. + 3 Appendices. Unpubl. MS.

Ghiselin, M.T. 1969. The evolution of hermaphroditism among animals. *Quart. Rev. Biol.* **44**:189-208.

Moe, M.A., Jr. 1969. Biology of the red grouper *Epinephelus morio* (Valenciennes) from the eastern Gulf of Mexico. *Fla. Dept. Nat. Res. Mar. Res. Lab. Prof. Pap. Ser.* **No. 10.** 95 pp.

Quintero-Hunter, I., H. Grier, and M. Muscato. 1991. Enhancement of histological detail using metanil yellow as counterstain in periodic acid Schiff's hematoxylin staining of glycol methacrylate tissue sections. *Biotech. & Histochem.* **66**(4):169-172.

Randall, J.E. 1968. *Caribbean Reef Fishes.* T.F.H. Publications. Jersey City, NJ. 318 pp.

Robertson, D.R., and R.R. Warner. 1978. Sexual patterns in the labroid fishes of the western Caribbean, II: the parrotfishes (Scaridae). *Smithson. Cont. Zool.* **No. 255.** 26 pp.

Sadovy, Y., and D.Y. Shapiro. 1987. Criteria for the diagnosis of hermaphroditism in fishes. *Copeia* **1987**(1):136-156.

Shapiro, D.Y., and R. Lubbock. 1980. Group sex ratio and sex reversal. *J. Theor. Biol.* **82**:411-426.

Warner, R.R. 1975. The reproductive biology of the protogynous hermaphrodite *Pimelometopon pulchrum* (Pisces: Labridae). *Fish. Bull., U.S.* **73**(2): 262-283.

Warner, R.R., and D.R. Robertson. 1978. Sexual patterns in the labroid fishes of the western Caribbean, I: the wrasses (Labridae). *Smithson. Cont. Zool.* **No. 254.** 27 pp.

Landings, Value, and Fishing Effort for Halfbeaks, *Hemiramphus* spp., in the South Florida Lampara Net Fishery

RICHARD MCBRIDE
Florida Fish and Wildlife Conservation Commission
Florida Marine Research Institute
100 Eighth Avenue S.E.
St. Petersburg, FL 33701 USA

ABSTRACT

Two halfbeak species, ballyhoo *(Hemiramphus brasiliensis)* and balao (*H. balao*), are targeted by lampara net fishers in south Florida. This report examines data from Florida's Marine Fisheries Information System (MFIS) to assess the status and trends of this bait fishery. Since 1986, when Florida's MFIS began, annual halfbeak landings have been relatively stable (median = 1.2 million lb or 0.54 million kg). In the early 1990s, halfbeak landings peaked temporarily in association with development of halfbeak fishing in Florida Bay. These increases were offset somewhat by declines in landings and effort in Palm Beach County, which occurred in response to Florida's 1995 ban on certain types of fishing nets. Ex-vessel prices have remained relatively stable and suggest that the dockside value of this fishery is more than $ 0.5 million annually. Annual halfbeak fishing trips were negatively correlated with the number of days that named storms (tropical storms or hurricanes) occurred in vicinity of Florida ($r^2 = 0.48$; $p = 0.009$). As a trend, the number of daily fishing trips by halfbeak fishers has declined since 1986. Increased storm activity in the last three years (1996 - 1999) may also explain why landings for these same years were below average. Overall, halfbeak harvest rates have been increasing since 1986, and variations in landings and fishing effort were largely explained by environmental factors. The status of this valuable living marine resource appears very good.

KEY WORDS: Bait fishery, stock assessment, storm effect

INTRODUCTION

Combined landings of ballyhoo, *Hemiramphus brasiliensis*, and balao, *H. balao*, constitute a small but valuable bait fishery in Florida (Berkeley et al. 1975, McBride et al. 1996). The two species are similar in appearance and are marketed together only as 'ballyhoo'. These halfbeak species are harvested with lampara nets in coastal waters of south Florida and sold as bait to anglers seeking gamefishes and foodfishes. Markets for fresh halfbeaks exist throughout the Florida Keys and as far north as Palm Beach County. Frozen, vacuum-packed halfbeaks are marketed widely throughout Florida, the middle Atlantic states, and the Caribbean region. Regional sport anglers also catch their own halfbeaks with cast nets and small (i.e., Sabiki) hooks, and they fish in areas where halfbeaks congregate. Therefore, the status and trends of the halfbeak fishery and its resource populations are of interest to a regional commercial fishery and an international fishing community.

This stock assessment is an extension of previous studies investigating the halfbeak fishery, particularly the trend of higher halfbeak landings in the early 1990s compared to the late 1980s (McBride 1996, McBride et al. 1996). There were also questions about how enactment of Florida's net ban referendum in July, 1995 would affect halfbeak landings and fishing effort. This regulation (s. 16, Art. X of the Florida Constitution) prohibits entangling nets in waters inshore of three miles on the Atlantic coast and nine miles on the Gulf coast of Florida. It also prohibits non-entangling nets larger than 500 ft^2 (such as those nets used by commercial halfbeak fishers), in waters less than 1 mile of Florida's Atlantic coast and three miles of the Gulf coast. Entangling nets are not used in the halfbeak fishery, so their prohibition in coastal waters would not directly affect the halfbeak fishery. But Florida's net ban could affect the ballyhoo fishery indirectly if bait became more valuable and fishers displaced from other fisheries entered the halfbeak fishery. These events prompted a reexamination of Florida's halfbeak fishery.

In this paper, I examine halfbeak landings, value, and fishing effort based on information reported to Florida's Marine Fisheries Information System (MFIS). Since 1986, Florida law requires wholesale transactions of marine organisms landed within the state to be reported to the MFIS. Because a majority of halfbeak landings are reported by piece rather than by weight, I begin with an analysis of how best to render the MFIS halfbeak piece data as pounds landed. Conversion factors were developed from measurements of fish from fishery catches. Also included is an analysis of how storm frequency affects this fishery, and how changes in fishing effort were associated with changes in catch-per-unit-effort. The purpose of this study was to assess the status of this valuable bait fishery in south Florida.

MATERIALS AND METHODS

An initial task of this study was to examine assumptions previously relied upon (e.g., McBride et al. 1996) to convert numbers of fish to pounds. From November 1995 to October 1998, ballyhoo and balao were collected randomly during 73 days of fishing. These fish were collected by a biologist onboard commercial fishing vessels in south Florida. A subsample of fish was collected from the first set completed each day by filling a 5 - gallon bucket with fish as the catch was transferred from the lampara net to holding boxes. A single subsample contained about 100 halfbeaks (approx. range: 50 - 200). Fish were kept on ice and brought back to the laboratory, where whole body weight (0.1 g) was measured for up to 30 fish per sex per species.

Commercial fishery data were available from Florida's MFIS. Although data exist for halfbeak landings since 1950, only data collected since 1986 are reviewed herein because earlier data were collected differently, which makes direct comparisons difficult (McBride et al. 1996). Annual landings were grouped as a fishing year (July - June), because halfbeak landings are highest during winter and grouping landings by a calendar year splits the peak landing period inappropriately (McBride 1996, McBride et al. 1996). Data for the south Florida lampara net fishery represent nearly all halfbeak landings and were kept separate from data for all other fishing locations and gear. Halfbeak landings for counties north of Brevard (east coast) and north of Pinellas (west coast) and for all other commercial gear (e.g., cast net, rod and reel) were less than 5%, and generally less than 1%, of the annual harvest. These minor catches were deleted from most analyses except where noted.

Average price per pound data was estimated for each fishing year from 1993 to 1999 using MFIS data. Halfbeak prices were not required to be reported to Florida's MFIS until January, 1995, so samples sizes prior to that date were small or not available. Price data for 1988 - 1989 and 1989 - 1990 were available to me from notes supplied by fishers. As noted above, because halfbeaks are mostly sold by the piece, most price data had to be converted to price per pound. In the MFIS database values for halfbeak price per pound exist, but this specific parameter refers to bulk fish that are usually damaged or too small to sell individually. A daily mean price per pound for halfbeaks was converted from price per piece data as a weighted average following these steps: Halfbeak numbers were converted to pounds by first multiplying the number of fish in each size category by the median fish weight for that size category (taken from McBride et al. 1996). The fish weight per category per trip was multiplied by the total price for the fish landed in that category for that trip. The weighted price per pound was calculated as the average value of all size categories reported for each trip (Sokal and Rohlf 1981). Finally, annual prices reported here are arithmetic means of all daily weighted price estimates available for each fishing year.

RESULTS AND DISCUSSION

Mean fish weight was generally lower in summer-autumn and higher in winter-spring (Table 1). Most ballyhoo and balao harvested are young-of-the-year (age-0) or have overwintered only once (age-1), and because few individuals live longer than one year, halfbeak numbers and biomass change dramatically from month to month. Young-of-the-year ballyhoo and balao first appear in commercial catches during summer and they appear to be fully recruited to the fishery by October (Berkeley et al. 1975, Berkeley and Houde 1978, McBride et al. in prep.). Thus, one would expect that the largest fish are found during winter-spring. In addition, ballyhoo are larger on average than balao, so the mean fish size in mixed landings depends on the proportions of each species. Because landings are greatest during winter, when large ballyhoo dominate the catches (Berkeley et al. 1975, McBride et al. 1996, McBride and Styer in prep.), using month-specific conversion factors could be more accurate than a single conversion factor.

Table 1. Combined mean fish weight of ballyhoo and balao for each month. Samples were collected by commercial lampara nets in south Florida from November 1995 to October 1998. Sample size refers to number of fish.

Month	Mean (lbs/fish)	Standard deviation	Sample size (n)
January	0.167	0.050	456
February	0.183	0.052	525
March	0.194	0.053	216
April	0.190	0.054	462
May	0.169	0.057	487
June	0.169	0.063	450
July	0.179	0.069	724
August	0.167	0.080	679
September	0.145	0.054	601
October	0.191	0.070	784
November	0.174	0.063	481
December	0.179	0.052	335
All months	0.175	0.064	6200

The average halfbeak, sampled from year-round commercial catches of ballyhoo and balao, weighed 0.175 lb (Table 1). Using this single value, I converted the number of fish recorded in the MFIS database into pounds of fish landed annually, and I compared the result to the annual landings estimated using month-specific conversion factors. Both methods produced annual landings values within 2% of each other. In practice, either method is suitable for examining long-term trends in landings as long as seasonal landings or species ratios do not vary significantly between years. The previous conversion factor used to render numbers of fish into pounds landed (0.1429 lbs/fish) was calculated from a small number of fish from a single sample and was much lower than the factor calculated here. Consequently, annual landings reported here are higher than, but should be more accurate than, those previously reported (e.g., McBride et al. 1996).

Annual halfbeak landings from the south Florida lampara net fishery were relatively stable from 1986 - 1999 (Figure 1). The median value during this period was 1.2 million lb (0.54 million kg), and landings ranged from 0.87 to 1.8 million lb (0.40 - 0.82 million kg). The south Florida halfbeak fishery appears to be the largest halfbeak fishery in the Gulf-Caribbean region. Landings have been reported for Panama (Meek and Hildbrand 1923), Curacao (Zaneveld 1962), South America

(Collette 1978), Virgin Islands (Beets and LaPlace 1991), and Puerto Rico (Kimmel 1987, Holliday 1997), but none approach one million pounds of annual landings. Historically, south Florida's halfbeak landings were low and stable prior to the late 1960s and then increased dramatically in response to increasing demand for bait and to improvements in fishing methods (Berkeley et al. 1975, McBride et al. 1996). During the 1970s and 1980s, landings appeared to be relatively stable at this increased level. In the early 1990s, however, landings increased to the highest levels ever reported for this fishery.

The temporary increase in halfbeak landings in the early 1990s was associated with geographic shifts in fishing effort. A direct analysis of fishing areas was not possible here, because fishers were not required to report fishing area until 1995. The MFIS database has required fishers to report the counties in which the fish were landed, however, and this can be used as a reasonable proxy for fishing location. Since 1986, both harvest and effort have increased in Monroe County relative to other counties (Figure 2). The percentages of the landings and trips that took place in Monroe County were both about 25 - 30% of all halfbeak landings and trips in the mid-1980s. Landings and trips for Monroe County increased rapidly in the early 1990s and have leveled off at about 60 - 70% since the mid-1990s. A previous survey of commercial halfbeak fishing for 1988 - 1991 (McBride et al. 1996) observed fishing activities offshore of Palm Beach, Dade, and Monroe counties, but at that time, there was no record of halfbeak fishing in the middle Keys or in Florida Bay, as there is now. The increased landings and effort that are evident for Monroe County today appear to be largely the result of the development of halfbeak fishing in Florida Bay in the early 1990s.

Implementation of the net ban in 1995 effectively eliminated halfbeak fishing in Palm Beach County, which also had the effect of shifting the halfbeak fishery farther south. Florida's net ban (see Introduction) essentially eliminated halfbeak fishing areas offshore of Palm Beach County because the continental shelf is so close to shore that a boat fishing one mile offshore is in water too deep for halfbeak fishing. These halfbeak fishers have either left the fishery or are fishing in Dade County since the net ban. The net ban had less effect on halfbeak fishing offshore of Dade and Monroe counties because there are numerous shallow areas beyond one mile where halfbeaks congregate in Atlantic waters offshore of these counties. In these counties the main effect of the net ban appears to be that fuel costs have risen on average because the legal fishing areas have moved farther offshore.

At the time of the implementation of the net ban referendum in 1995, there was concern that fishers displaced from other fisheries would shift to the halfbeak fishery and increase fishing pressure further. This concern does not appear to have materialized significantly. McBride et al. (1996) noted that 10 vessels were operating in the south Florida lampara net fishery during 1988 - 1991. During

1995-1998, a total of 14 boats were observed operating in the fishery; however, by 1999, 3 of the 14 had dropped out.

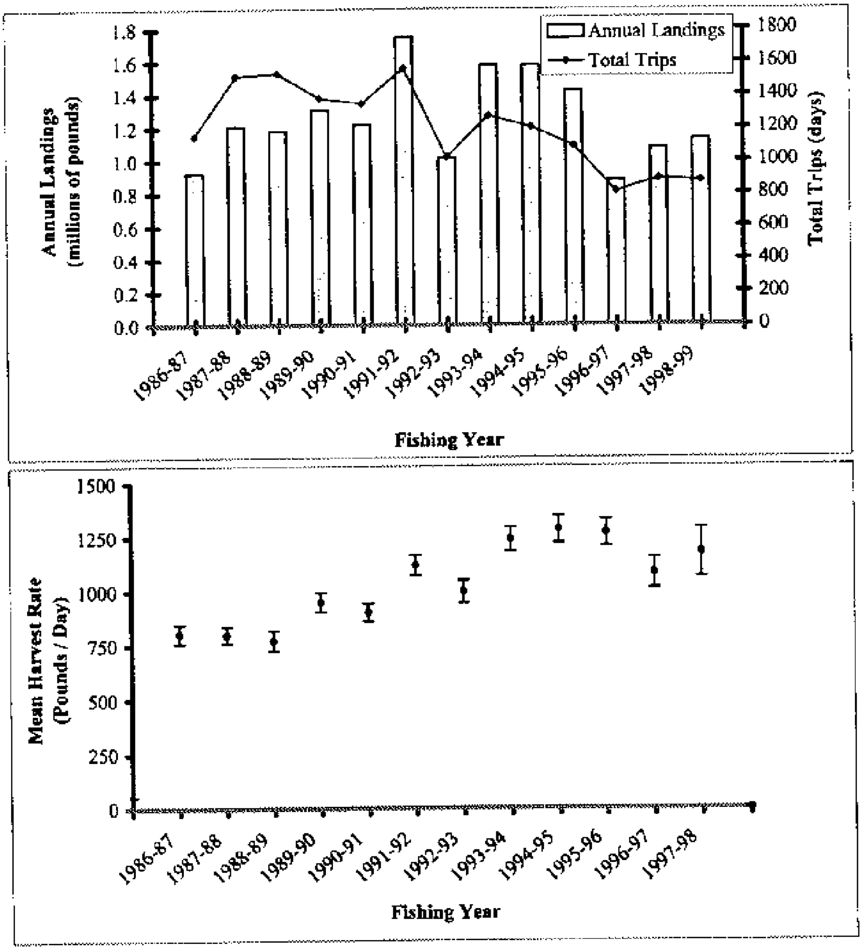


Figure 1. Halfbeak catch-and-effort data for the south Florida lampara net fishery reported to Florida's Marine Fisheries Information System. Annual landings and total number of trips reported separately (top). Mean daily harvest (lbs) ± 95% confidence limits for each fishing year (bottom). Data for 1998 - 1999 are incomplete (Up to April 1999 [Batch #564]).

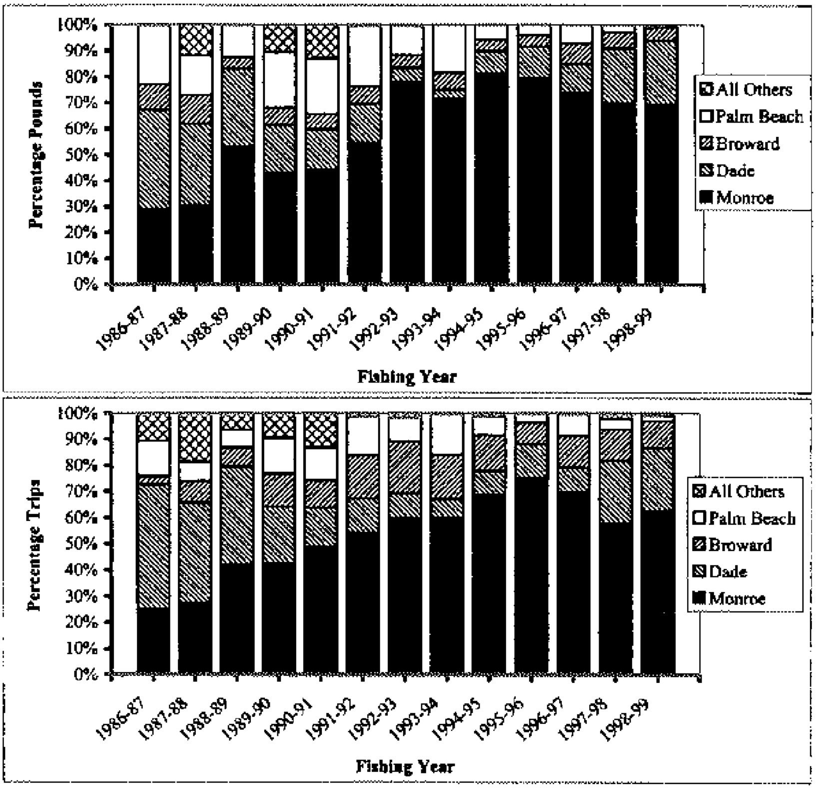


Figure 2. Percentage contribution to halfbeak landings (top) and fishing trips (bottom) per county as reported to Florida's Marine Fisheries Information System. Data includes all gear used statewide. Data for 1998 - 1999 are incomplete (up to April 1999 [Batch #563])

Ex-vessel prices (i.e., those paid directly to fishers) were relatively flat during the 1990s and averaged $0.52/lb (Table 2). This price stability may explain why the number of fishing vessels did not increase beyond one new boat in the late 1990s. In comparison to the monetary value of other bait fisheries in Florida, an average price of $0.52/lb is relatively high, and it suggests that the overall ex-vessel value of the halfbeak fishery is $0.5 - 1.0 million annually. Combined landings of Spanish sardines and Atlantic thread herring totaled more than 5.0 million pounds in 1996, but because these baitfishes were valued at only $0.14-

0.20/lb that year (MFIS unpublished data), their combined value is about equivalent to the value of south Florida's halfbeaks alone. The special onboard processing of halfbeaks and their demand as fresh bait increase their value, at least relative to other baitfishes that are sold frozen, ungraded, and without vacuum-sealed packaging (McBride et al. 1996).

Table 2. The mean ex-vessel (=dockside) price per lb for halfbeaks, ballyhoo and balao, sold from the south Florida lampara net fishery. Sample size refers to the number of trips reporting complete information to Florida's Marine Fisheries Information System. See text for methods of calculating price. Data for 1998 – 1999 are incomplete (up to April 1999 [Batch #564]).

Fishing Year	Mean price per lb (U.S. $)	Standard deviation	Sample size (n)
1988-89	0.69	0.32	4
1989-90	0.58	0.24	17
1993-94	0.41	0.18	16
1994-95	0.50	0.20	18
1995-96	0.46	0.20	117
1996-97	0.41	0.09	119
1997-98	0.60	0.56	210
1998-99	0.50	0.19	220
Overall mean	0.52		

An obvious trend in the MFIS data was that the total number of daily trips declined from 1200 - 1600 during the late 1980s to 800 - 1000 during the late 1990s (Figure 1). If one considers that the 11 boats currently operating in the fishery will fish for approximately 200 working days per year each, then the projected maximum number of total fishing days for this fishery is 2200. In recent years, boats fished less than half of those working days. Because this decline was evident during the early 1990s, it could not be associated with the implementation of Florida's net ban. Instead, this trend appears to be largely the result of weather-related phenomena, particularly the presence of named storms (i.e., tropical storm or hurricane) in the western Atlantic (Figure 3). Named storms in the vicinity of Florida (here the area chosen was bounded by 20°N, 30°N, 70°W, and 90°W) can produce winds and wave action that interfere with fishing in south Florida. A linear, least-squares regression predicted that 27 fishing days were lost for every day that the center of a named storm was present within this area.

This weather vs. fishing effort model does not account for the severity of individual storms, and adjustments for storm severity could improve the model's fit. For example, the most significant hurricane to hit south Florida during this study period was Andrew, which hit Homestead in August 1992 and disrupted local commerce for weeks to months. Both halfbeak landings and trips for 1992 - 1993 were the third-lowest measured, even though the total number of storm-days was only 13 (mean storm-days per year = 12.5; Figure 3). Other weather patterns also affect the number of commercial halfbeak fishing trips and cause variation not explained by this simple linear model. During the El Niño year of 1997 - 1998, when only 12 storm-days occurred, approximately 70 days were too windy or rainy for ballyhoo fishing, according to records supplied by halfbeak fishers. These same records indicate that another 20 - 30 days can be lost to engine or gear repairs (observers witnessed 8 breakdowns at sea during 107 trips). In addition, most fishers do not fish for halfbeaks during August, when catch rates are very low compared to catch rates in other months. At this time, fishers may schedule boat maintenance, vacations, or participate in other types of fisheries. Increased storm activity in recent years caused some fishers to take off earlier in the summer or extend this vacation period, according to interviews with halfbeak fishers.

Among the many variables affecting the number of fishing days, days lost to weather may vary greatly between years, and virtually all south Florida halfbeak fishers are affected by the same bad weather. The number of storm-days explained nearly half the variability in number of fishing days each year (Figure 3). Further improvements to this model could be made with supplemental weather data, but the predictive outcome is already evident: during years of "average" or "better" weather, there will be more fishing days than during years of "poor" weather, but as long as the number of fishing vessels remains stable these increases should not be taken as a sign that the fishery is expanding.

Annual halfbeak landings had been relatively stable even though fishing effort had declined during the period 1986 - 1999, which resulted in increasing daily harvests from 750 - 1000 lbs to 1000-1250 lbs (Figure 1). There has been no recent improvement in fishing methods to account for an increase in harvest efficiency. There is also no evidence or even anecdotal suggestion that halfbeak abundance had increased steadily during this period. No previous study has investigated factors that may control the number of recruits to the halfbeak fishery. Ballyhoo and balao have maximum life spans of about one year (Berkeley and Houde 1978), however, so strong or weak year-classes are unlikely to directly affect population size for more than a single year.

In theory, as the number of halfbeak fishing trips declined, harvest rates could increase because of increases in marginal yield (Gulland 1968). I tested for such a biological response by modeling halfbeak harvest rates as a parabolic function of fishing trips (Schaefer 1954) but the model was not significant (Figure 4). This model attempts to identify rates of fishing that are beyond sustainable levels. This may be difficult to detect for halfbeaks because ballyhoo and balao are highly mobile and their geographic distributions are widespread relative to the distribution of this fishery (Collette 1965). Still, during years of better than average weather when the number of fishing trips can be quite high, local depletions could theoretically occur in certain areas of south Florida, at least until movements by ballyhoo and balao redistribute their populations into exploitable densities. While plausible, modeling the data did not support this.

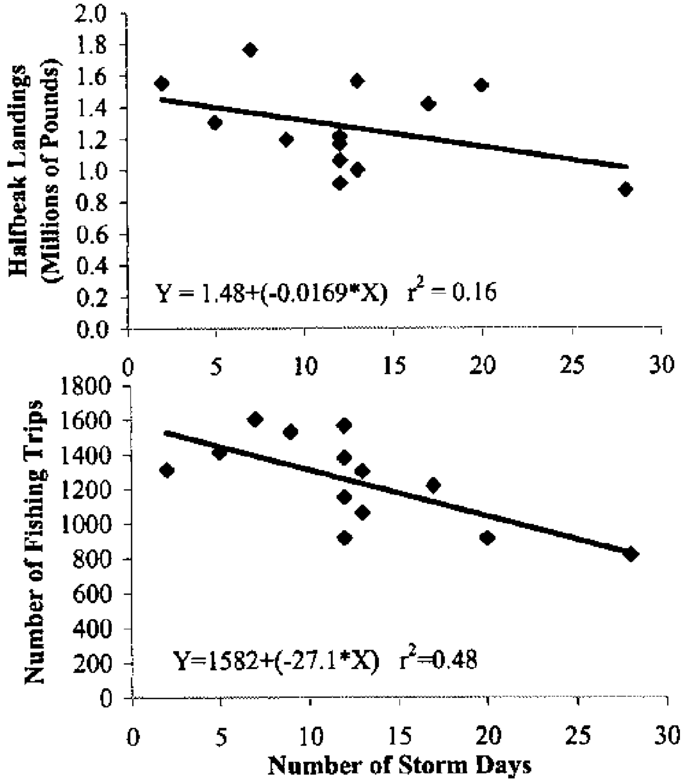


Figure 3. Annual landings (top) and number of halfbeak fishing trips (bottom) by lampara net fishers in south Florida plotted against the number of storm-days in each year, 1986 - 1999. A storm-day is defined as each day the center of a named storm was present within the area bounded by 20°N - 30°N and 70°W - 90°W; storm-days were counted from images obtained from the National Hurricane Center (www.nhc.noaa.gov). The number of storm-days was not significantly correlated with annual landings (p = 0.17), but it did explain a signficant amount of the variability in fishing trips between years (p = 0.009). Data for 1998 - 1999 are incomplete (up to April 1999 [Batch #564]).

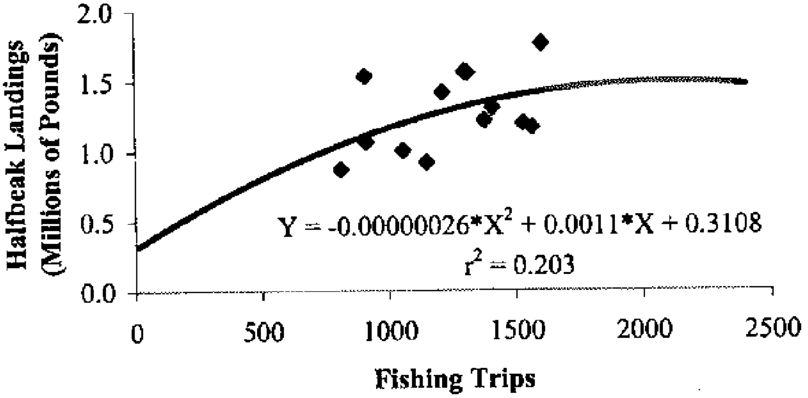


Figure 4. Annual landings as a parabolic function of the number of halfbeak fishing trips by lampara net fishers in south Florida for 1986 - 1999. This model was not significant ($p = 0.32$). Data for 1998 - 1999 are incomplete (up to April 1999 [Batch #564]).

Increasing halfbeak harvest rates relative to a declining number of fishing days could also arise by another process, one that would be relatively unusual compared to other fisheries. During winter and spring, harvest rates are often driven more by demand for halfbeak bait than by exploitable supply, so that realized daily harvests are lower than potential daily catches. Target harvests are set by most fish houses based on the perceived market demand for fresh bait as well as the capacity of these fish houses to sort and package frozen halfbeaks. During winter, halfbeak biomass and numbers are high, and daily harvests are three to four times higher than during summer (McBride et al. 1996). Winter harvests can frequently meet the maximum storage capacity onboard, about 2,600 lbs or 1,200 kg of halfbeaks for most vessels. The higher catch rates during winter also result in shorter fishing days during winter than summer. In this context, a reduced number of trips in years of 'poor' weather may lead to shortfalls in cumulative landings relative to demand for bait, so that in late winter or spring months, while catch rates are still high, fish houses allow higher daily target harvests on average. The relationship between the number of storm days and landings was not significant (Figure 3), which suggests that fishers can increase their harvest rates somewhat when fishing days are reduced.

This process is most likely limited, however, because during most of the year fishers can not catch full boat loads. In addition, interviews with fishers indicated that demand decreased in recent years in certain markets, because increased storm activity also suppresses demand for bait. Notably, frequent storm activity in the last three years (1996 - 1999), for example, may explain why landings for these same years were below average. Market forces certainly affect halfbeak landings and fishing effort to some degree, and they may at least partially explain increasing harvest rates in recent years for the reasons outlined above.

Overall, increasing harvest rates in the halfbeak fishery suggests that the status of this resource is very good. Currently, no impact can be estimated for recreational fishers and charter boat captains who collect halfbeaks for bait, but this examination of Florida's MFIS database showed no indication of negative fishing effects by the commercial halfbeak fishery. Variations in commercial halfbeak landings and fishing effort were largely explained by environmental factors and these should be accounted for in future assessments.

ACKNOWLEDGMENTS

The halfbeak fishers and processors in south Florida voluntarily cooperated by allowing observers onboard and by donating fish samples. T. Brown, J. Hunt, R. Moretti, and D. Snodgrass provided logistical support in the Florida Keys. R. Beaver, K. Krumm, E. Robillard, J. Styer, and J. Whittington assisted in fish collecting and processing. M. Norris provided data from Florida's Marine Fisheries Information System. M. Palavido assisted with hurricane data. L. Barbieri, B. Mahmoudi, M. Murphy, and M. Norris offered helpful critiques to improve this manuscript. This work was funded in part by Saltonstall-Kennedy funds (NOAA Grant# NA77FD0069). I am grateful for everyone's help in this study.

LITERATURE CITED

Beets, J. and J. A. LaPlace. 1991. Virgin Islands baitfish survey: October to December 1986. *Proc. Gulf Carib. Fish. Inst.* **40**:55-63.

Berkeley, S. A. and E. D. Houde. 1978. Biology of two exploited species of halfbeaks, *Hemiramphus brasiliensis* and *H. balao* from southeast Florida. *Bull. Mar. Sci.* **28**(4):624-644.

Berkeley, S. A., E. D. Houde and F. Williams. 1975. Fishery and biology of ballyhoo on the southeast Florida coast. *Univ. Miami Sea Grant Prog., Spec. Rep.* **4**, 15 pp.

Collette, B. B. 1965. Hemiramphidae (Pisces, Synentognathi) from tropical west Africa. *Atl. Rep.* **8**:217-235.

Collette, B. B. 1978. Hemiramphidae. Unpaginated in: W. Fischer. (ed.) *FAO Species Identification Sheets for Fishery Purposes, Western Central*

Atlantic (Fishing Area 31). Food and Agriculture Organization of the United Nations. Rome.

Gulland, J.A. 1968. The concept of the marginal yield from exploited fish stocks. *J. Con. Int. Expl. Mer.* **32**:256-261.

Holliday, M. C. 1997. *Fisheries of the United States, 1996.* Current Fishery Statistics No. 9600. 14 pp.

Kimmel, J. J. 1987. A fisheries dependant [sic] survey of baitfishes in Puerto Rico. *Proc. Gulf Carib. Fish. Inst.* **40**:64-81.

McBride, R. S. [1996]. Progress Report on Florida's Halfbeak Fishery. Florida Department of Environmental Protection, Florida Marine Research Institute. Report to the Florida Marine Fisheries Commission. 10 July 1996. 12 pp. Unpubl. MS.

McBride, R.S., L. Foushee and B. Mahmoudi. 1996. Florida's halfbeak, *Hemiramphus* spp., bait fishery. *Mar. Fish. Rev.* **58**(1-2):29-38.

McBride, R.S. and J. Styer. Harvest rates, species composition, and size structure for the south Florida lampara net fishery. In prep.

McBride, R.S., J. Styer and P. Thurman. Age, growth, reproductive biology and spawning grounds of ballyhoo and balao in south Florida. In prep.

Meek, S.E. and S.F. Hildebrand. 1923. *The marine fishes of Panama.* Field Museum of Natural History, Publication No. 215. **15**(Zoological Series). 330 pp.

Schaefer, M.B. 1954. Some aspects of the dynamics of populations important to the management of the commercial marine fisheries. *Bull. Inter-Am. Trop. Tuna Comm.* **1**:27-56.

Sokal, R.R. and F.J. Rohlf. 1981. *Biometry.* Second Edition. W.H. Freeman and Co., New York, NY. 859 pp.

Zaneveld, J.S. 1962. The fishery resources and fishery industries of the Netherlands Antilles. *Proc. Gulf Carib. Fish. Inst.* **14**:137-171.

Aspects of Reproduction, Age and Growth of the Lane Snapper, *Lutjanus synagris* (Linnaeus, 1758) in Jamaican Coastal Waters

KARL A. AIKEN
Department of Life Sciences
University of the West Indies, Mona campus
Kingston, Jamaica

ABSTRACT

The lane snapper, *Lutjanus synagris* (Linnaeus), 1758 is the most important commercial species taken by the large and growing seine and trammel net fishery on the south shelf of Jamaica. Samples from the commercial net fishery between February, 1996 and June, 1999 showed a size range from 150 – 430 mm FL (fork length) with a mean of 232.1 mm FL (± 52.4, N = 1,094), both sexes combined. Most fishes in samples were already gutted by beachside vendors. Mean sizes for males and females were 219.5 (± 72, N = 99) and 220.0 mm FL (± 9.82, N = 235), respectively. Weight range for both sexes combined were 75 - 1,100g. Mean size at maturity was 268 mm FL (± 0.27) and 221 (± 1.85) in males and females respectively. Sex ratio (F:M) was 2.6:1.0 in the size range 150 - 310 mm FL, and above that 100% female. Prolonged spawning was found with a maximum in July for both sexes. Sagittal otoliths were examined whole and by thin (300 micrometres) section and revealed opaque and hyaline bands. Whole otolith ages produced a maximum of six years but sections showed maximum ages of at least 12 years and that whole otolith analysis alone could seriously underestimate age. Marginal increment analysis showed one opaque zone was laid down in July and one hyaline in March annually. The relationship between otolith weight and fork length was described by the regression equation $Y = 0.0403X - 1.543$ ($r^2 = 0.9407$) for males, and for females $Y = 0.0405X - 1.1716$ ($r^2 = 0.9711$). ELEFAN routines produced $K = 0.25$, $L_{\infty} = 320$ mm FL and to = -0.0001, for males and for females $K = 0.070$, $L_{\infty} = 538.7$ mm FL, to = -3.795, phi prime = 4.346.

KEY WORDS: *Lutjanus synagris*, population dynamics, Jamaica, otoliths

INTRODUCTION

The family Lutjanidae is the most valuable finfish component of Jamaican commercial landings. The fishery on the island shelf is dominated by trap gear, but nets are the second most common fishing gear (Sahney 1983, Munro 1983). As a group, frequency and mean size of fishes from trap catches have generally been in decline for years, probably due to overfishing (Aiken and Haughton 1987). There is a trend towards the use of nets such as seines and tangle nets, away from traps in order to avoid trap theft. The lane snapper (*Lutjanus* *synagris*, Linnaeus, 1758) is one of the more commonly taken seine and tangle net fishes. However, one exception to the trend of declining catches is thought to be the lane snapper, which seems to have produced consistently modest catches for many years. This species is taken almost exclusively by beach seine and tangle (called "Chinese") monofilament nets. Lane snappers are common over nearshore seagrass beds, and muddy, soft substrates, particularly on the south coast (Figure 1). The importance of net gears has greatly increased in the period since at least 1980. The abundance of this species may be a function of it being relatively unaffected by the intense trap fishery which has decimated reef fishes. The lane snapper has never been the focus of a dedicated study, and thus little is known of the biology in national waters, almost nothing of age and growth on this species in Jamaica. Munro and Thompson (1983) collected some Jamaican data on lane snappers but only as a small part of a much larger study on reef fishes, and additionally, their findings were all length-based. Some extraterritorial comparative data on age, growth and reproduction of lane snapper are available from Cuba (Claro and Reshetnikov 1981, Claro 1983, Rubio 1986, Rubio et al. 1985, Pozo et al. 1991, Pedroso and Pozo 1991, Salahanghe 1981), Puerto Rico (Acosta and Appeldoorn 1992), Mexico (Torres-Lara and Chavez 1987, Torres-Lara and Salas-Marquez 1990, Torres-Lara et al. 1990, Torres-Lara et al. 1991), Trinidad & Tobago (Manickchand-Dass 1987), and the United States (Manooch and Mason 1984).

The objectives of this study were to investigate the use of otoliths (earstones) as a method to provide age and growth information which would be new to Jamaica. Also an objective was to refine earlier conclusions on lane snapper reproductive biology as well as to investigate the ease with which marks in otoliths could be assessed for aging fish as a routine procedure.

STUDY AREA AND METHODS

The study area of this small-scale fishery-dependent investigation was the central south island shelf in the centre of which was the main sampling site, Old Harbour Bay fishing beach, the largest such beach in the island (Figure 1). Data collected there from February 1996 to June 1998 included fork length (mm), weight (g), sex and maturity stage (if whole), and the pair of sagittal otoliths. All fish had to be quickly returned to suppliers so this requirement determined that the method of otolith collection had to be the least destructive "up-through-the gills" method. Sexual maturity was recorded as either immature, active or ripe, a modification of the technique Munro (1983) successfully used for Jamaican reef fishes. Whole otoliths were viewed in a watchglass in glycerine using binocular microscopy. Selected otoliths were mounted in epoxy resin, then sectioned with a high-speed diamond saw, ground to 300 micrometeres on a lapping machine. Afterwards each was mounted on a small microscope slide for viewing with a compound microscope. Otolith radius was measured with an ocular micrometer where one ocular micrometer unit was equal to 14.7 micrometers (μ). Validation of otolith periodicity was determined by marginal increment analysis (analysis of monthly distribution opaque-edged (fast growth) otoliths in randomly selected specimens from fishes within the commercial size range).

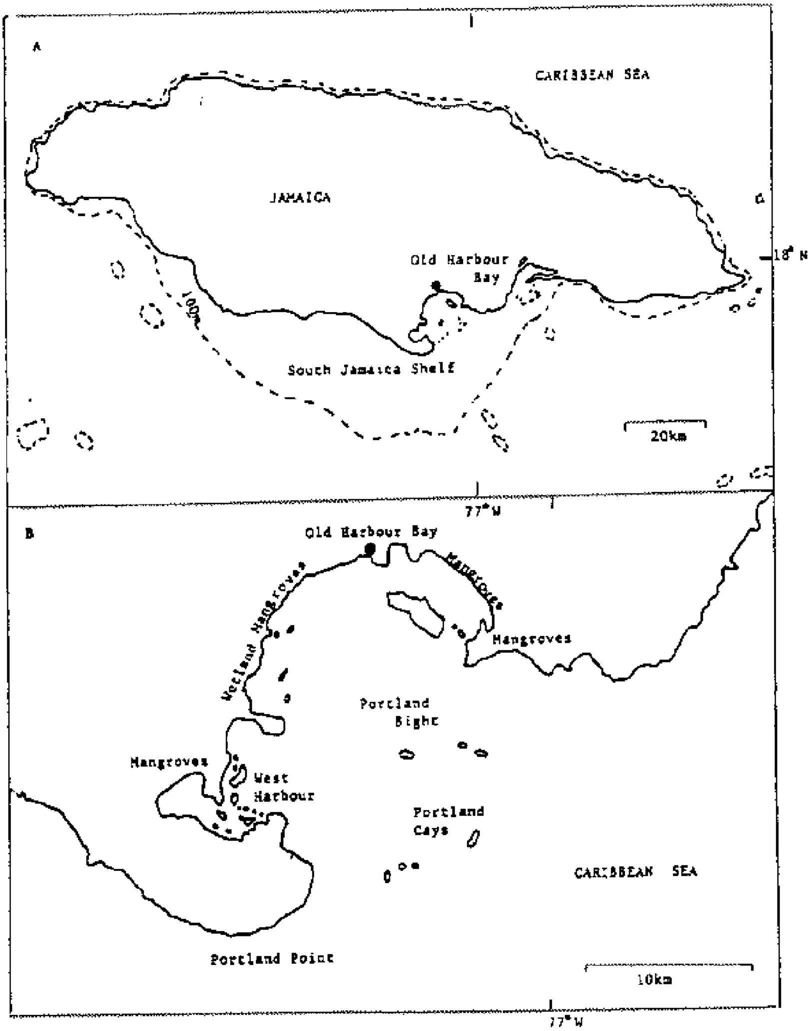


Figure 1. A map of Jamaica showing the 100 m depth contour and proximal oceanic banks (A), and Portland Bight and Old Harbour Bay sampling sites

RESULTS

Population Structure

Investigation of population structure showed that the size range (sexes combined) landed from the commercial seine and tangle net fishery on the south shelf was from 150 – 430 mm FL (fork length) with a mean of 232.1 mm FL (± 52.4 mm, N = 1,094) Figure 2. Mean size for males was 219.5 mm FL (± 3.72mm, N = 99) and for females 220.0 mm FL (± 52.4 mm, N = 235), Figures 3 and 4, respectively. Sex ratio (F:M) was found to be 2.6:1.0 in the size range 150 mm – 310 mm FL. Above 310 mm FL, there was 100% females. Table 1 lists the sex ration results and compares it to other findings elsewhere for the species.

Table 1. Sex ratios in *Lutjanus synagris* by geographic location

Location	Population Ratio (F:M)	Ratio at Length	Reference
Cuba	1.3:1.0 (N = 1,640)	ND	Claro, 1981
Cuba	1.9:1.0 (N = 1,441)	80 - 420 mm FL 50 - 70% F > 420 mm FL, 100% F	Rodrigues-Pino, 1962
Colombia (Caribbean)	1:1.3 (N = 2,472)	100 - 400 mm F 40 - 60% F	Erhardt, 1997
Jamaica	1.0:1.0 (N = 99)	170 - 350 mm FL	Thompson and Munro, 1983
Jamaica	2.6:1.0 (N = 326)	150 - 320 mm FL >320 100% F	This Study

Reproduction

Female lane snapper achieved first sexual maturity at 170 mm FL while males did this at 150 mm FL. Female mean size at maturity was 268 mm FL (± 0.27) while that for males was 221 mm FL (± 1.85 mm). Table 2 presents these results and compares them with those for Cuba, the nearest (160 km) Caribbean neighbour with this species. Study of the spawning season showed that ripe and spent individulas could be found throughout the year indicating prolonged spawning. Spawning maxima for both males and females were found from July to August annually. Figures 5 and 6 show the monthly distribution of immature, active and ripe specimens. Arrows indicate maxima for ripe individuals. Table 3 compares these results with periods of maximum spawning elsewhere.

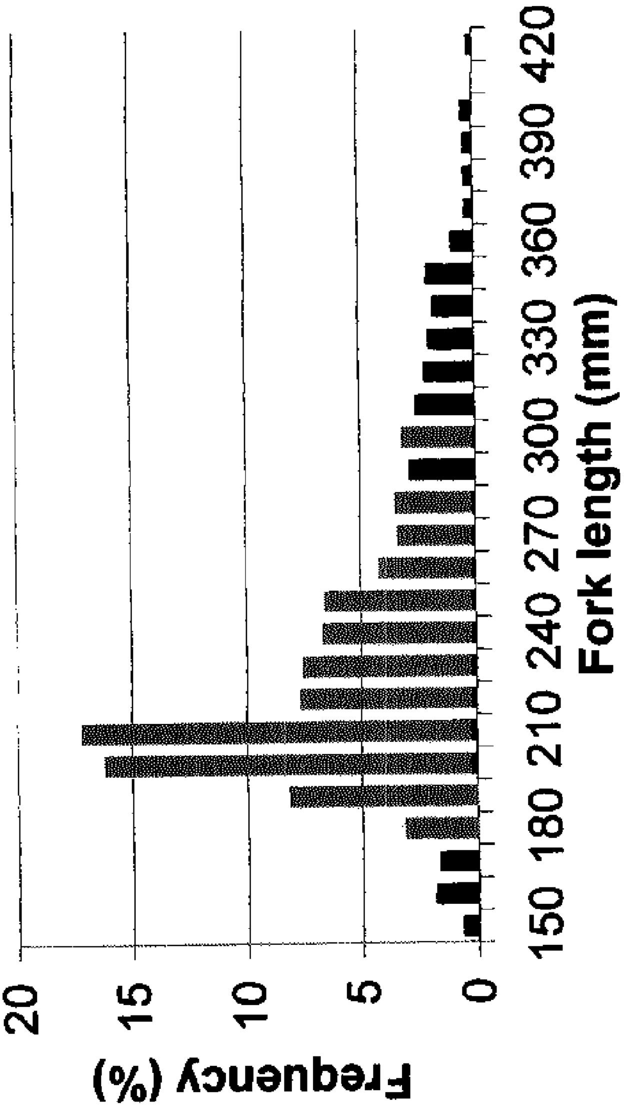


Figure 2. Cumulative length-frequency of male and female *L. synagris* during 1996 - 1998 at Old Harbour Bay, Jamaica. (N = 1,094; Mean = 232.1 mm FL)

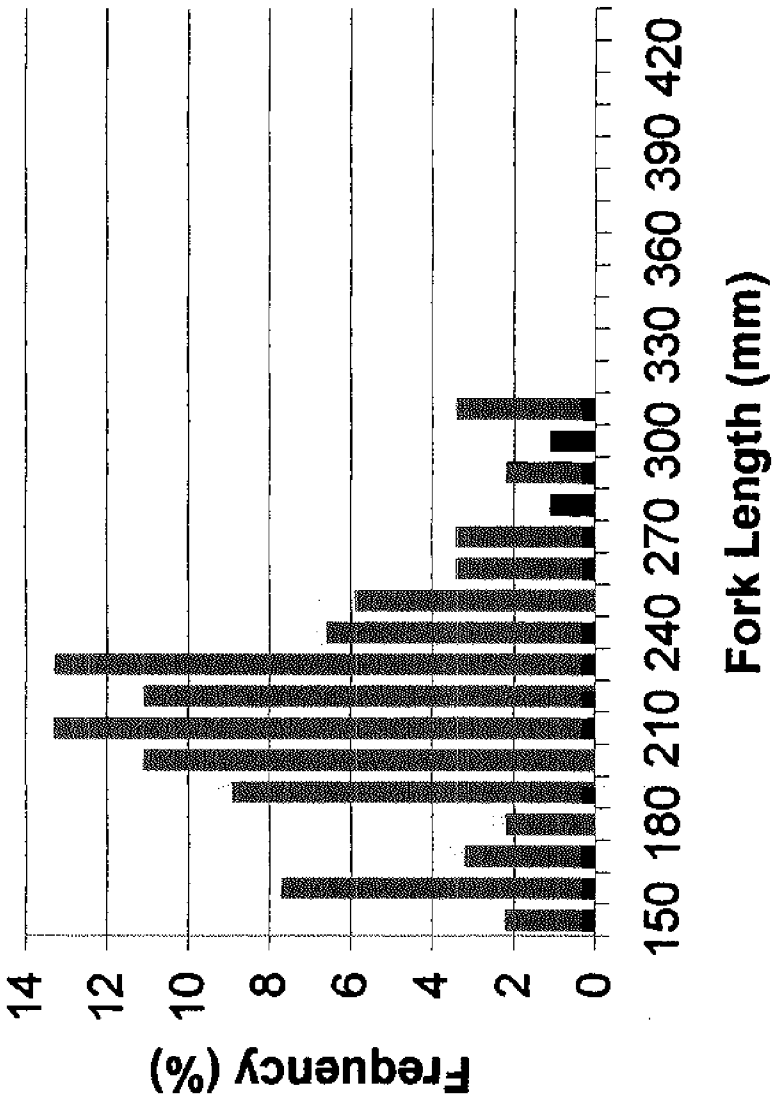


Figure 3. Length-frequency of male *L. synagris* during 1996 - 1998 at Old Harbour Bay, Jamaica. (N = 90; Mean = 219.5 mm FL)

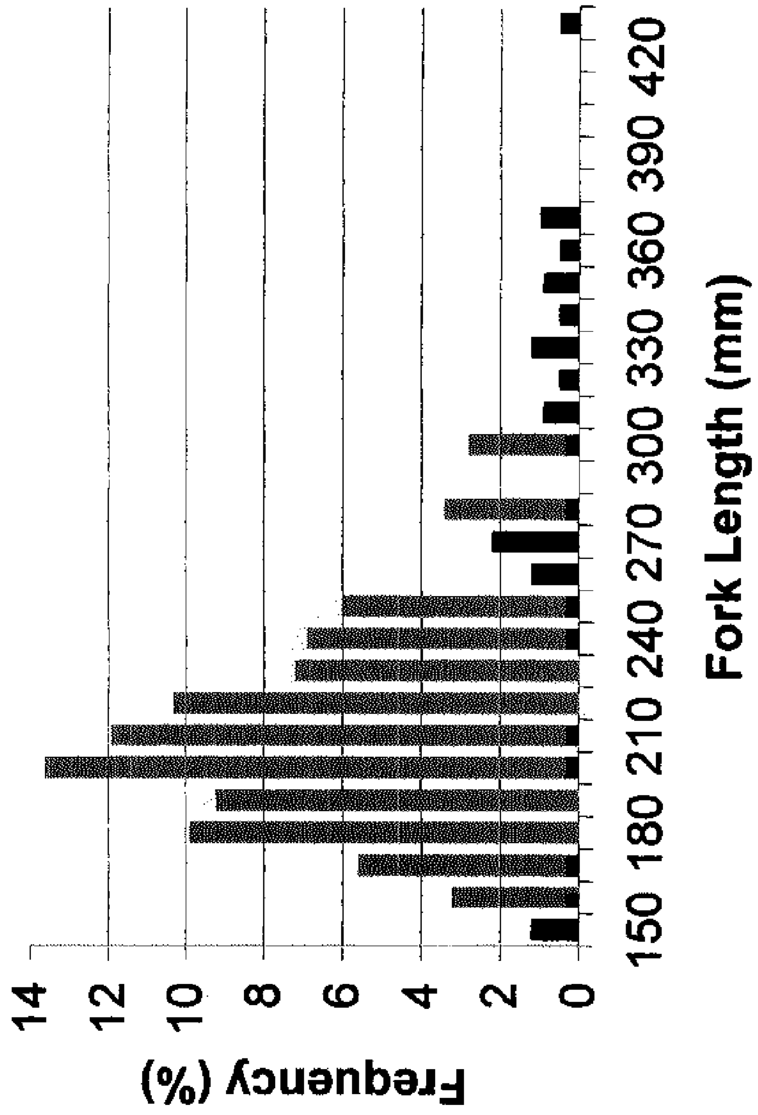


Figure 4. Length-frequency of female *L. synagris* during 1996 - 1998 at Old Harbour Bay, Jamaica. (N = 26; Mean = 220.0 mm FL)

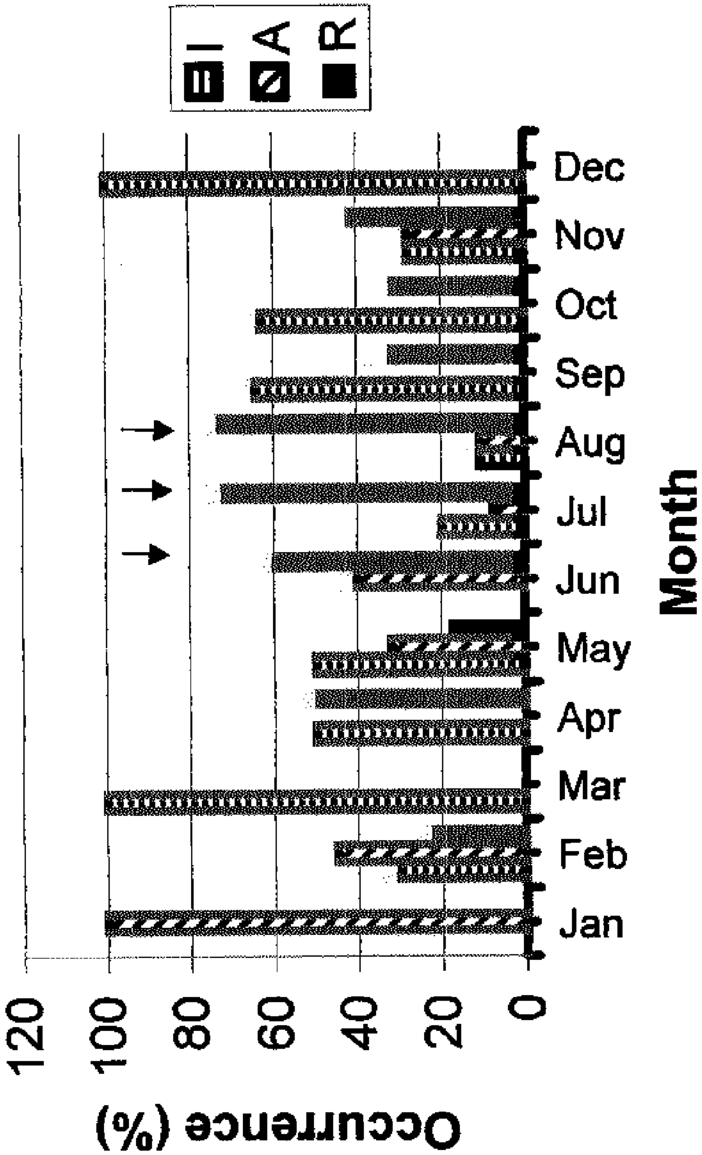


Figure 5. Frequency of occurrence of immature (I), active (A) and ripe (R) male *L. synagris* during 1996 - 1998 at Old Harbour Bay, Jamaica (arrows indicate spawning season)

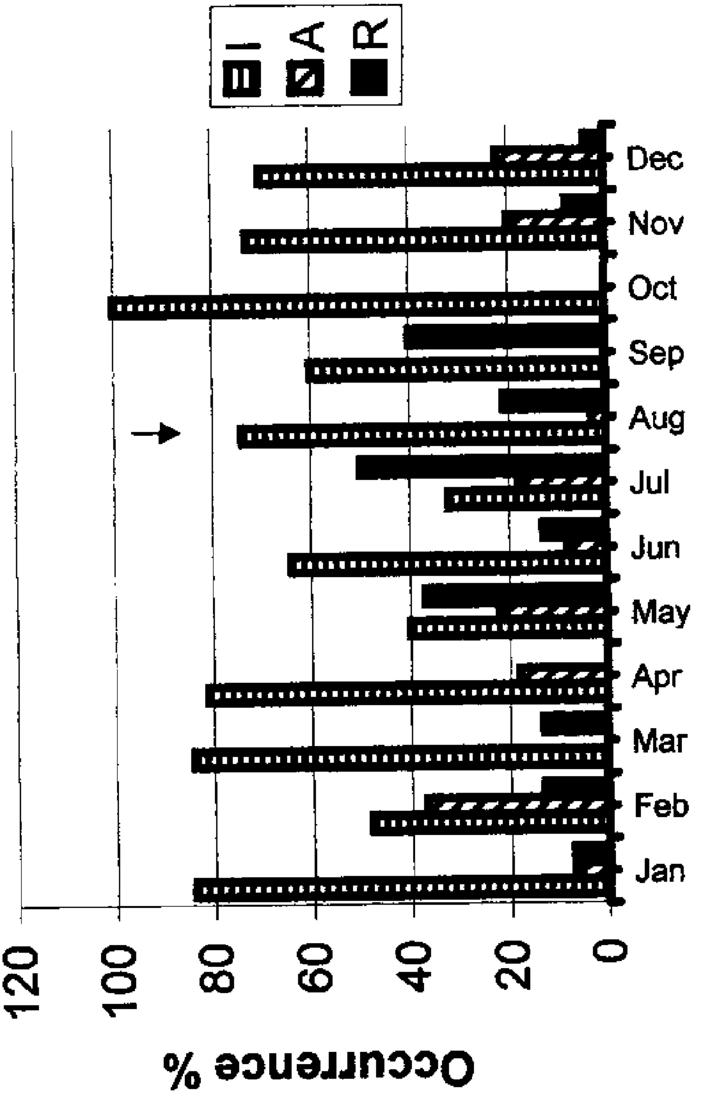


Figure 6. Frequency of occurrence of immature (I), active (A) and ripe (R) female *L. synagris* during 1996 - 1998 at Old Harbour Bay, Jamaica (arrows indicate spawning season)

Table 2. Size at maturity in *Lutjanus synagris* by geographic location

Location	Sex	Length at first maturity (mm)	Maximum length in Population (mm)	Percent of max. length at first maturity	Reference
Cuba	M	91 TL	300 TL	22.8	Reshetnikov and Claro, 1976
	F	91 TL	400 TL	22.8	
Cuba	M	85 FL	275 FL	30.9	Ridriguez-Pino, 1962
	F	85 FL	310 FL	27.4	
Jamaica	M	183 FL	410 FL	44.6	Thompson and Munro, 1983
	F	176 FL	410 FL	48.4	
Jamaica	M	150 FL	320 FL	48.4	This study
	F	170 FL	430 FL	39.5	

Table 3. Spawning season of *L. synagris* by geographic region

Location	Maximum spawning time (months)	Analysis Type V = visual H = histological	Reference
Cuba	March - Sept.	V/H	Ridrigues-Pino, 1962
Cuba	April - Sept.	V	Reshetnikov and Claro, 1976
USVI	March - May Aug.	V	Erdman
Jamaica	March - April	V	Thompson and Munro, 1983
Jamaica	July - Aug.	V	This study
Trinidad	Jan. - Dec. (ripe)	V	Manckchand-Dass, 1987
Colombia (Caribbean)	Jan. - June Sept. and Dec.	V	Erhardt, 1997

Age and Growth

Examination of sagittal otoliths revealed clear patterns of marks in both whole and sectioned specimens. Generally, these were clearly defined opaque (fast growth) and hyaline (slow growth) zones. Marginal increment analysis of 300 sagittal otoliths showed that opaque (fast growing) margins were deposited in July and August. Hyaline (slower growth) zones were laid down in March annually. Thus, lane snapper had one opaque and one hyaline zone deposited each calendar year. Figure 7 shows the periodicity of deposition of these zones each year. Examination of whole otoliths showed a maximum age of six years in the range 150 -360 mm FL (N = 78). However, sectioned otoliths in the size range 150 - 390 mm FL showed a maximum age of 14 years (a female, N = 94). Figures 8 and 9 show the maximum age found in whole and sectioned lane snapper otoliths. Age analysis of sectioned otoliths is taken to be a more accurate indicator of true age, given validation by marginal increment analysis which proves that only one opaqe/hyaline zone pair is deposited annually. Length-frequency analysis and age-frequency analysis suggests that the majority of the females landed are 2 - 5 years in age while the males are from 1 to 3 years old. Also, the data suggest that males do not live as long as females.

Analysis of otolith radius versus length data showed a significant relationship where fork length (both sexes combined) was related to otolith radius by the regression equation $Y = 1.5262X - 3.9574$ ($r^2 = 0.08452$; N = 54) and is shown in Figure 10. Otolith weight changes with fork length analyses showed a significant relationship in male lane snapper related by the regression equation $Y = 0.04303X - 1.1543$ ($r^2 = 0.9407$, N = 15) and shown in Figure 11. For otoliths from females, weight was related to fork length by the equation $Y = 0.0405X - 1.1716$ (r2 = 0.9711, N = 24) and shown in Figure 12.

Growth patterns estimated from ELEFAN analyses suggest that males and females have different growth rates and the von Bertalannfy growth parameters derived were; males K = 0.25-yr, L_{oo} = 320 mm FL and t_o = -0.0001 (Figure 13) and for females, K = 0.076-yr, L_{oo} = 538.7 mm FL, and t_o = -3.970 and phi prime value = 4.346 (Figure 14).

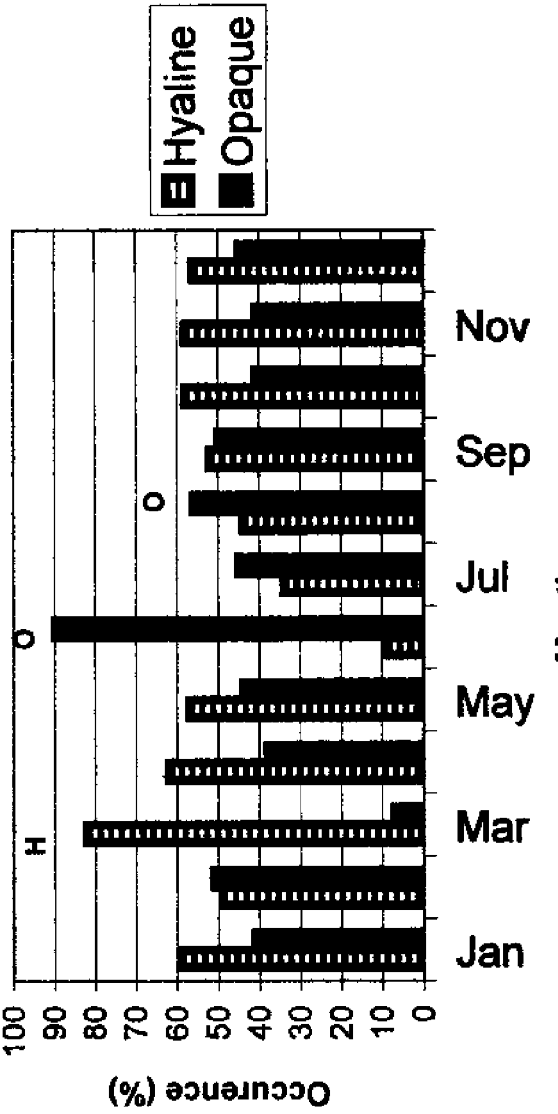


Figure 7. Marginal increment analysis, by month, of the otoliths of *L. synagris* showing hyaline vs. opaque margins

Figure 8. Maximum age found in whole otoliths of *L. synagris* from Jamaica

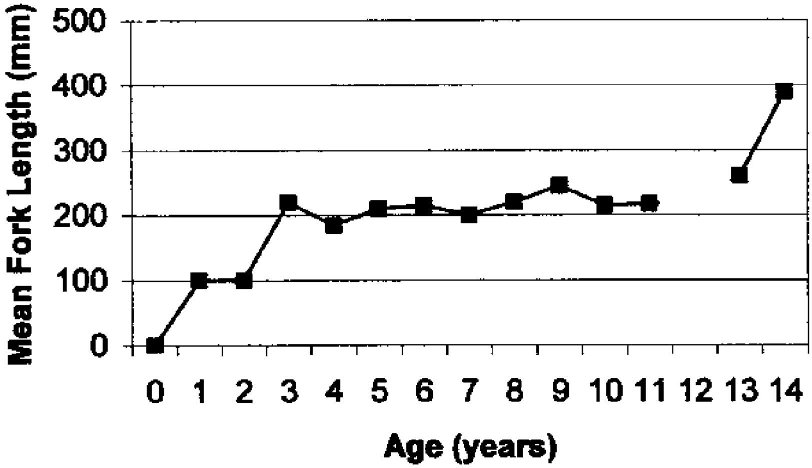


Figure 9. Maximum age found in sectioned otoliths of *L. synagris* from Jamaica

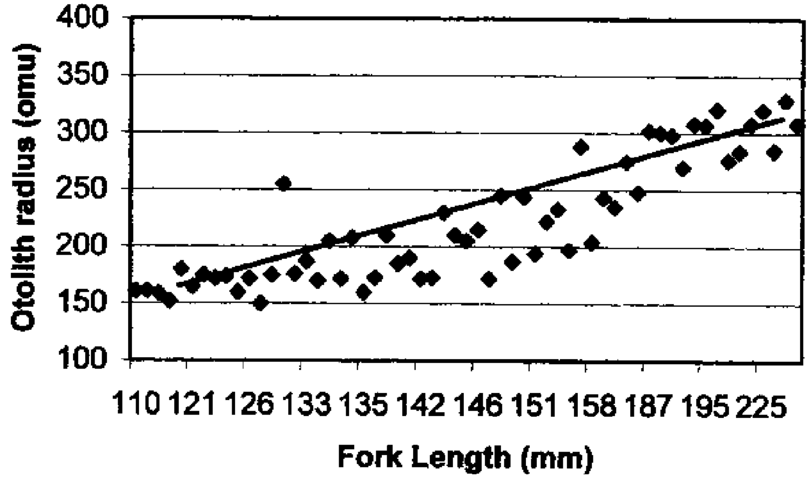


Figure 10. Fork length vs. otolith radius of *L. synagris* from Jamaica. $y = 1.5262x - 3.9574$; $R^2 = 0.8452$

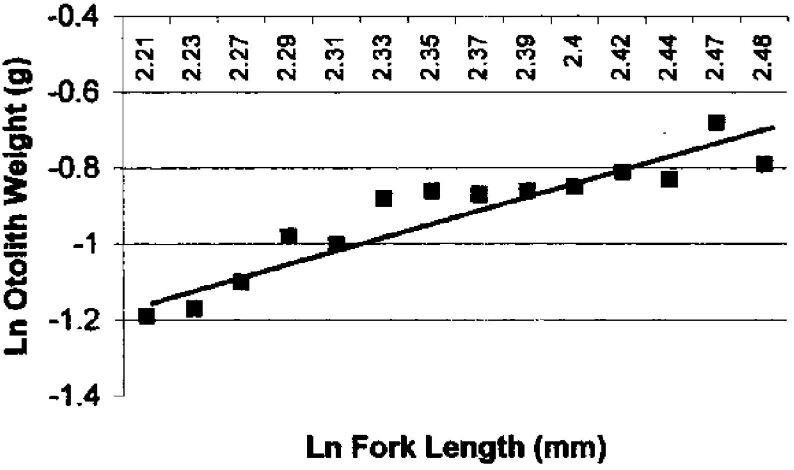


Figure 11. Linear regression for fork length vs. otolith weight of male *L. synagris* from Jamaica. $y = 0.0403x - 1.1543$; $R^2 = 0.9407$

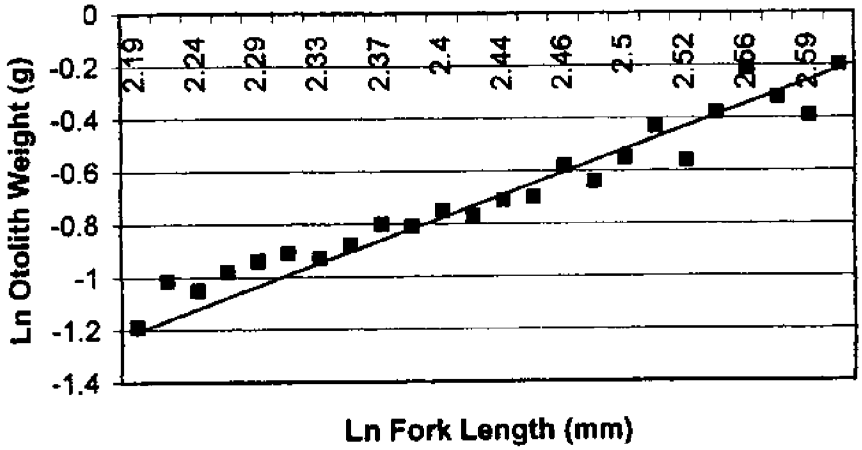


Figure 12. Linear regression for otolith weight vs. fork length of female *L. synagris* from Jamaica. $y = 0.0405x - 1..1715$; $R^2 = 0.9711$

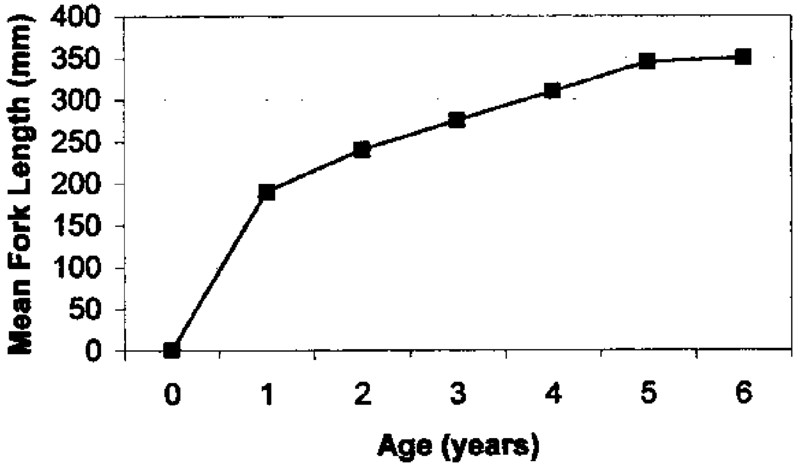


Figure 13. Growth patterns estimated from ELEFAN analyses for male *L. synagris* from Jamaica

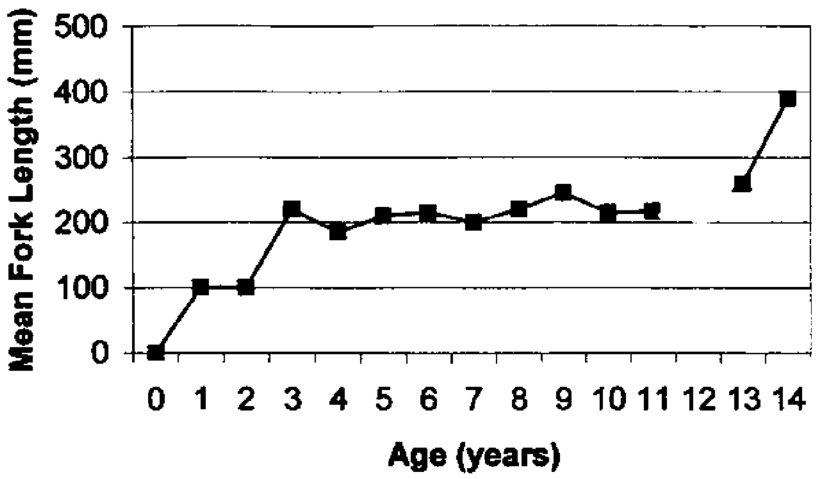


Figure 14. Growth patterns estimated from ELEFAN analyses for female *L. synagris* from Jamaica

DISCUSSION

The present study showed in part that lane snapper in Jamaican waters could attain an asymptotic size of at least 430 mm FL. The maximum size to which this species grows in other areas was not greatly different. Sex ratio (male to females) increases with increasing age. The largest lane snappers are female (>320 mm FL). Age at sexual maturity for males and females is approximately two years for males and three years for females. Males arrive at sexual maturity and spawn brfore females of the same age. This means that it is possible that young viable males could spawn with more fecund slightly older females. Estimates of mean percentage of otoliths with opaque zones at growing edges suggest that these zones are produced in summer at the time of maximum reproductive activity. Validation through marginal increment analysis showed that one opaque and one hyaline zone is produced in sagittal otoliths each year. Whole otolth analysis produced for both sexes a maximum age of six years. Sectioned otoliths produced maximum ages around 12 years. Lane snapper in Jamaican waters therefore attain a greater age than previously known. This is close to the 10 years maximum age in Florida (Manooch and Mason 1984) and greater than most other areas except for Bermuda with 19 years (Luckhurst, pers.

comm. 1998). Male fish gow more slowly than females and are outlived by females. These latter can achieve at least 12 years, but males appear not to survive beyond approximately six years. Otolith sections provided more accurate ages than did whole otolith. The present study suggests that the use of whole otolith age analysis alone could markedly underestimate age in lane snapper. It is notelworthy that many of the studies in the literature did not use sections but relied on somle ageing of whole otoliths. This general preference for whole otolith ageing may be due to the higher cost and time consumption of section preparation. Perhaps the most significant finding is that lane snapper can attain a much greater age than previously expected in Jamaican waters. This suggests that if this is typical of other reef fishes, then the already seriously overfished stocks may take even longer to recover (within an overall management plan) than thought earlier. Removal of larger (>320 mm FL) fishes will direcly impact females fishes more than males, and will have significant implications for future fish stocks

With training and practice, otolith analyses could become part of the fish aging methodology in Jamaica. It is recommended that a combination of analytical methods always be used in ageing investigation rather than one method.

ACKNOWLEDGEMENTS

I am grateful to the Fisheries Division of the Ministry of Agriculture, which assisted in establishing contacts at various early trial data collection sites. This study could not have been done without the continuous cooperation of vendors at the Old Harbour Bay landing site, St. Catherine, particularly Isola Simpson and Merlene Chastri.

LITERATURE CITED

Acosta, A. and R.S. Appeldoorn. 1992. Estimation of growth, mortality and yield per recruit for *Lutjanus synagris* (Linnaeus) in Puerto Rico. *Bull. Mar. Sci.* **50(2)**:282-291.

Aiken, K.A. and M.O. Haughton.1987. Status of the Jamaica reef fishery and proposals for its management. *Proc. Gulf Carib. Fish. Instit.* **38**:469-484.

Claro, R. 1981. Ecologia y ciclo de vida de la biajaiba, *Lutjanus synagris* (Linnaeus), en la plataforma Cubana. II. Biologia pesquera. Informe Cientifico-Technico No. 177, Academia de Ciencia de Cuba, Habana. 54 pp.

Claro, R. 1982. Ecologia y ciclo de vida de la biajaiba, *Lutjanus synagris* (Linnaeus), en la plataforma Cubana IV. Reproduccion. Reporte de

Investigacion del Instituto de Oceanologia. No.5. Academia de Ciencias de Cuba, La Habana. 36 pp.

Claro, R. and Y.S.Reshetnikov. 1981. Ecologia y ciclo de vida de la biajaiba, *Lutjanus synagris* (Linnaeus), en la plataform Cubana. 1. Formacion de marcas de crecimiento en sus estructuras. Informe Cientifico-Technico No. 174, Academia de Ciencias de Cuba, La Habana. 28 pp.

Manickchand-Dass, S. 1987. Reproduction, age and growth of the lane snapper, *Lutjanus synagris* (Linnaeus), in Trinidad, West Indies. *Bull. Mar. Sci.* **40**(1):22-28.

Manooch, C.S. III and Mason, D. 1984. Age, growth and mortality of lane snapper from southern Florida. *Northeast Gulf Science.* 7(1):109-115.

Munro, J.L. (ed.).1983. *Caribbean coral reef fishery resources.* ICLARM Studies and Reviews No. 7, Manila. 276 pp.

Munro, J.L. and R.W. Thompson. 1983 The biology, ecology and bionomics of the snappers, Lutjanidae. Pages 94-109 in: Munro, J.L. (ed.)*Caribbean coral reef fishery resources.* ICLARM Studies and Reviews No. 7, Manila.

Pedroso, B.L. and E.Pozo. 1991. Edad y crecimiento de la biajaiba (*Lutjanus synagris* Linnaeus, 1758) en el Golfo de Batabano, Cuba. FAO Fish. Rept. 431 Supplement, FIP /R431 Suppl., Rome. 94-110.

Pozo, E., Pedroso, B. and M.H. Obregon. 1991. Edad y crecimiento de la biajaiba (*Lutjanus synagris* Linnaeus, 1758) en el Golfo de Ana Maria, Cuba. FAO Fish. Rept. No. 431 Supplement, FIP/R431 Suppl., Rome. 78-93.

Rubio, R. 1986. Estudio sobre la edad y el crecimiento de la biajaiba (*Lutjanus synagris* Linnaeus, 1758), Pisces Lutjanidae), en la plataforma noroccidental de Cuba. 5to. Foro Cient., Cent. Invest. Pesq., La Habana.

Rubio, R., Salahanghe, P. and M. Betancourt. 1985. Relacciones de la edad con el pargo de la biajaiba (*Lutjanus synagris*) de la plataforma suroccidental de Cuba. *Rev. Cub. Invest. Pesq.* **10**(3-4):77-90.

Sahney, A.K. 1983. Sample survey of the fishing industry in Jamaica – 1981. FAO Fish. Rep. No.278 Suppl., Rome. 255-275.

Salabanghe, P. 1981. Edad y crecimeinto de la biajaba (*Lutjanus synagris* Linnaeus). Resum.I. Congr. Nac. Cienc. Biol., La Habana.

Torres, R. and E. Chavez. 1987. Evaluacion y diagnostico de la pesqueria de rubia (*Lutjanus synagris* L.) en el estado de Yucatan. *Ciencias Marinas* **13**(1):7-29.

Torres-Lara, R. and S. Salas-Marquez. 1990. Crecimiento y mortalidad de la rubia *Lutjanus synagris* de las costas de Yucatan durante las temporadas de pesca 1983-1985. *An. Inst. Cienc. Mar. y Limnol. Univ. Natl. Auton. Mecico.* **1**7(2):205-214.

Torres-Lara, Ayala-Perez, L.A. and S. S.Salas-Marquez. 1990. Determinacion de algunos parametros de la reproducion de la rubia, *Lutjanus synagris,* en las costas de Veracruz en 1983. *An. Inst. Cienc. Mar. y Limnol. Univ. Nal. Auton. Mexico.* **1**7(2):157-161.

Torres-Lara, R. Salas-Marquez, S., Mexicano-Cintora G.and M.A. Cabrera-Vazquez. 1991. Sinopsis de la dinamica poblacional de cinco especies de la familia Lutjanidae en la costa de Yucatan. *An. Inst. Ciencias Mar. y Lmnol. Univ. Nal. Auton. Mexico.* **1**8(2):189-198.

Diferenciación Sexual e Inicio de la Madurez de la Cachicata Blanca, *Haemulon plumieri* (Pisces: Haemulidae), en La Parguera, Puerto Rico

JEANNETTE G. MATEO and RICHARD S. APPELDOORN
Department of Marine Sciences
University of Puerto Rico
Mayagüez, Puerto Rico 00681-9013 USA

La cachicata blanca, *Haemulon plumieri*, es una especie dominante y un componente importante de las pesquerías deportivas y comerciales en la región del Caribe. Por ser muy suceptible a ser capturada por todas las artes predominantes (nasas, redes, arpones, compresores) esta especie está en riesgo de ser sobrepescada. Los estudios reproductivos necesarios para el manejo de la especie son escasos y generalmente dirigidos hacia individuos adultos. Con el fin de caracterizar las etapas tempranas del desarrollo reproductivo y determinar la talla en la cual se inicia la diferenciación y la madurez sexual en *H. plumieri*, se realizó un estudio de febrero a mayo de 1997. Se anotaron detalles individuales de comportamiento, talla, ubicación dentro o fuera de la costa y se determinó el estadio gonádico correspondiente. Se encontró que la talla mínima de diferenciación sexual fue de 72 mm LH en las hembras y de 86 mm LH en los machos, mientras que la madurez sexual se inicia a 91 mm LH en las hembras y 86 mm LH en los machos. El cincuenta por ciento de las hembras maduran a 120 mmLH y los machos desde 80 mm LH. Estos valores son menores que los reportados anteriormente para la misma especie.

PALABRAS CLAVE: *Haemulon plumieri*, diferenciación sexual, primera madurez.

Sexual differentiation and First Maturation of the White Grunt, *Haemulon plumieri* (Pisces: Haemulidae), in La Parguera, Puerto Rico

The white grunt, *Haemulon plumieri*, is a dominant species and an important component of the sport and commercial fisheries in the Caribbean region. Because it is very susceptible to capture by all dominate gears (traps, nets, spears, hooks) this species is at risk of being overfished. The reproductive studies needed for management of this species are scarce and generally directed toward larger adults. A study was conducted from February to May of 1997 to characterize the early stages of reproductive development and to determine the lengths at differentiation and first maturation in *H. plumieri*. Individual details of behavior, length, and inshore-offshore distribution were recorded and the corresponding gonadic stage was determined. The minimal observed lengths at sexual differentiation was 72 mm FL in females and 86 mm FL in males, while sexual maturity started at 91 mm FL for females and 86 mm FL for males. Fifty percent of females were mature at 120 mm FL, 80 mm FL for males. These values are less than those previously reported for this species.

KEY WORDS: *Haemulon plumieri*, sexual differentiation, first maturation

INTRODUCCION

La cachicata blanca (*Haemulon plumieri*) es un pez arrecifal de gran importancia comercial y recreacional en las pesquerías de nasa del oeste del Atlántico y en las aguas del Caribe (Manooch 1976, Danmann 1980; McFarland et al., 1979). Es explotada en Puerto Rico e Islas Vírgenes (McFarland, 1979 Darcy 1983); en la República Dominicana (Aquino e Infante 1990) y en toda la plataforma cubana (Claro et al. 1994). Por ser abundante y alcanzar una de las mayores tallas dentro de su familia, la especie se encuentra bajo riesgo de sobrepesca (Appeldoorn y Lindeman 1985). Los estudios reproductivos necesarios para su manejo adecuado son escasos y orientados generalmente a descripciones histológicas.

No se ha observado dimorfismo sexual ni el desove en condiciones naturales en *H. plumieri*, pero se ha descrito el desarrollo de huevos (Sacksena y Richards 1975) y larvas en cautiverio (Lindeman 1986). Se sabe que es una especie gonocorística (Román Cordero 1991), con camadas fraccionales, múltiples o de serie (Bagenal y Braum 1981, Román Cordero 1991, García-Cagide 1987).

La talla de primera reproducción en *H. plumieri* ha sido calculada por diversos autores (Mota y Pessoa 1973, Billings y Munro 1974, Darcy 1983, García Cagide y Claro 1983, Román Cordero 1991, García Cagide et al. 1994) pero en la mayoría de los casos se han contemplado sólo los adultos, por lo cual, una parte importante de la población potencialmente reproductora queda fuera de los análisis

Teniendo en consideración la escasez de datos en relación al tamaño en que *H. plumieri* comienza a madurar y la necesidad de actualizar y enriquecer la información existente sobre el desarrollo sexual de la cachicata blanca, los objetivos particulares de este trabajo fueron:

i) determinar la talla a la cual *H. plumieri* se diferencia sexualmente y,
ii) registrar la talla de primera madurez sexual en hembras y machos de la especie.

METODOLOGIA

Se colectaron doscientos ocho especímenes de *H. plumieri* de varios tamaños en La Parguera, suroeste de Puerto Rico desde Febrero hasta Mayo de 1997. Los peces fueron capturados con arpón, y el comportamiento previo a la captura fue anotado para cada individuo y sus gónadas, así como los valores de longitud y peso.

Las gónadas fueron procesadas histológicamente siguiendo la metodología propuesta por Luna (1968) y ligeramente modificada por Polofski (c.p.) que conlleva deshidratación e inmersión en parafina y luego aclaramiento y tinción con hematoxilina de Harris y eosina. Los cortes histológicos se analizaron por triplicado y aquellas en donde no se pudo establecer el sexo fueron catalogadas como indeterminadas. En la identificación de las distintas etapas celulares, del sexo y grado de desarrollo se siguió a Román Cordero (1991), Asoh (1991), Claro (1996) y Posada (1996). La característica principal que permitió clasificar los especímenes en los inicios del desarrollo fue la presencia de ovocitos primarios en las hembras y en los machos, los espermatocitos primarios (Figura 1). Las hembras con proliferación de ovocitos terciarios y los machos con espermatocitos terciarios y espermatozoides fueron catalogados como maduros inactivos.

RESULTADOS

Treinta y tres de los individuos colectados no pudieron ser identificados debido al mal estado de las gónadas o al sexo irreconocible por estar en una etapa muy temprana del desarrollo gonádico. La proporción total fue de machos:hembras =1:2.27 (N = 160). En el caso de los especímenes maduros la proporción sexual hallada fue de machos:hembras = 1:2.94 (N = 69). La talla mínima de diferenciación sexual encontrada para la cachicata blanca fue de 86 mm LH en los machos y de 72 mm LH en las hembras, mientras que el tamaño mínimo en el cual los primeros alcanzaron la madurez fue a los 86 mm LH y las hembras a los 91 mm LH.

Al tomar en consideración las diferentes etapas de desarrollo y madurez de las gónadas en este trabajo, se encontró que el 50% de las hembras de *H. plumieri* estaban en período de madurez inactiva a partir de 120 mm LH; maduras activas desde 160 mm LH y desovadas desde 190 mm LH. Se halló un 100% de individuos inmaduros entre 70 y 80 mm LH; maduros activos a partir de 180 mm LH y desovados a 210 mm LH. En el caso de los machos, los resultados de este trabajo indican que el 50% de los mismos están maduros inactivos desde los 80mm LH, pero que es posible encontrar especímenes inactivos con tallas por encima de los 220 mm LH. El 50% de especímenes maduros activos solo fue encontrado a partir de 230 mm LH.

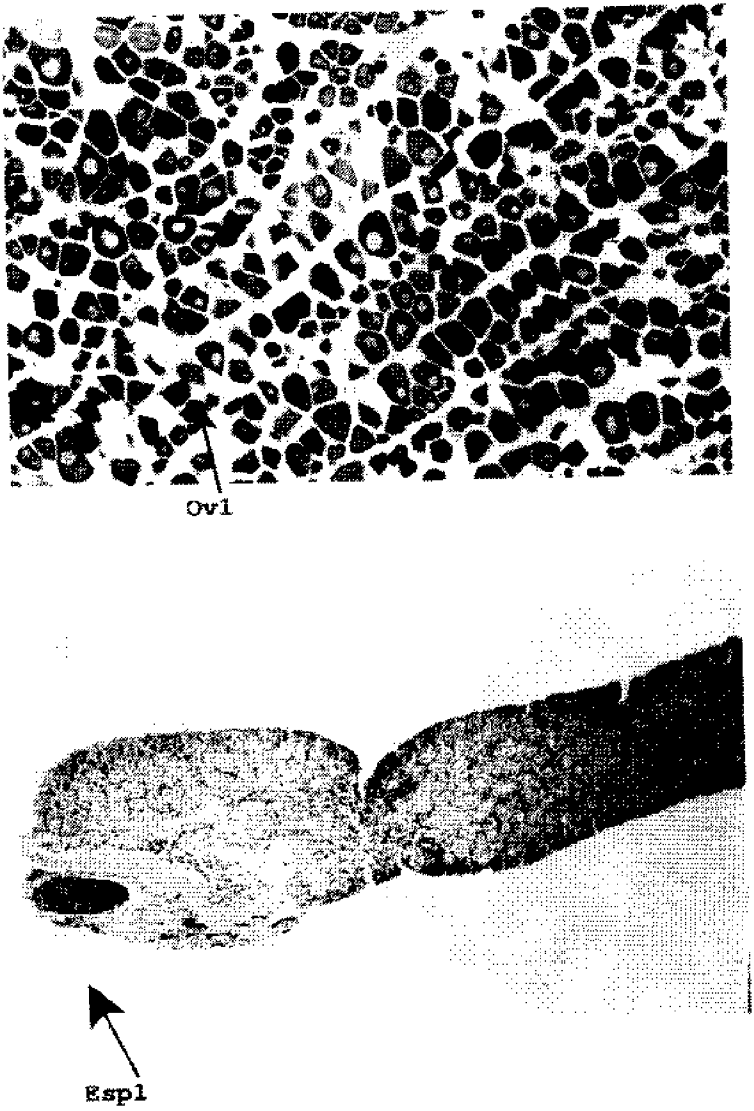


Figura 1. A) Primeras etapas en la formación de los ovocitos en una hembra inmadura de *Haemulon plumieri* (Ovi1 = ovocito primario, longitud horquilla = 114 mm, aumento = 40x). B) Macho en los inicios del desarrollo sexual (Esp1 = espermatocitos primarios, longitud horquilla = 129 mm, aumento = 100x).

DISCUSION

La razón sexual total y en especímenes maduros hallada en el presente estudio, difiere de la proporción encontrada por Román Cordero (1991) para la especie en Puerto Rico (machos:hembras = 1:1.05) y del valor teorético esperado en las poblaciones (1:1). Estos resultados sugieren que las hembras en la población no son un recurso limitado. Diferentes autores han propuesto que la diferencias entre proporciones de sexo son debidas a diferencias sexuales en el crecimiento, mortalidad o disponibilidad de alimento. García Cagide et al. (1994) plantean que la proporción en que se encuentran los individuos está sujeta a gran variabilidad y está relacionado con el carácter adaptativo de la especie y con el hábitat ocupado por cada sexo y éste puede variar durante el ciclo anual y la ontogenia.

Las tallas mínimas de diferenciación sexual encontradas para la cachicata blanca en el presente trabajo (hembras = 72 mm LH, machos = 86 mm LH) son ligeramente menores que las indicadas por otros autores (Tabla 1). En estos trabajos se evidencia una ligera tendencia en las hembras a diferenciarse sexualmente a una talla menor que en los machos, aunque éstos alcancen tamaños mayores. Nagahama, (1983) indica que, usando las células germinales en meiosis como criterio, se ha hallado que la diferenciación sexual ocurre primero en hembras. Esto ha sido demostrado en el pez medaka, *Oryzias latipes* y en la tilapia, *Sarotherodon mossambicus* (Hoar et al. 1983).

Tabla 1. Tallas de diferenciación e inicio de la maduración en la cachicata blanca, *Haemulon plumieri* en La Parguera, Puerto Rico. Los valores representan las tallas de longitud horquilla en milimetros (DS= diferenciación sexual, IM= inicio de la madurez, L50= longitud de clase en la cual el 50% de los especímenes están maduros). M = machos; H = hembras

Autores	DS	IM (Lmin)	L_{50}
Mateo (1999)	h = 72, m = 86	h = 91, m = 86	h = 120, m = 80
Román Cordero (1991)	h = 74, m = 97	h = 165, m = 160	h = 210, m = 165
García Cagide (1987)	90 ??	h = 110, m = 120	
García Cagide y Claro (1983)		140 - 160	
Gaut y Munro (1983)			h h = 220, m = 200
Billins y Munro (1974)		h = 143, m = 145	
Mota y Pessoa (1973)		h = 110, m = 120	

La definición de las tallas de diferenciación y de maduración sexual pueden ser elementos importantes al momento de clasificar las especies e individuos de acuerdo a su estadío ontogenético, en donde, usualmente, sólo se toman en cuenta el tamaño promedio de la especie y su conducta. Nuestros resultados, al ser comparados con otros sugieren que la cachicata blanca obtiene su diferenciación sexual aún en tallas propias de juveniles.

El proceso de diferenciación sexual de las gónadas ha sido estudiado en diferentes peces teleósteos y se ha encontrado que este período varía de acuerdo a la especie y al sexo de los individuos. Redding y Patiño (1993) explican que la diferenciación sexual en peces es un proceso controlado por genes específicos en donde las células germinales bipotenciales podrían diferenciarse como espermatogonias en respuesta a andro-inductores y otras como ovogonias en presencia de gino-inductores, sin embargo la naturaleza de esos agentes no se conoce de manera definitiva.

Las tallas de inicio de la madurez en hembras y machos de *H. plumieri* registradas en este estudio fueron 91 y 86 mm LH, respectivamente, y son menores que las indicadas por otros autores para la misma especie (Tabla 1).

Al calcular la talla en la cual el 50% de los individuos alcanzan la madurez sexual, se encontró que los machos pueden estar maduros inactivos a partir de los 80 mm LH, mientras que las hembras maduraron a una talla más grande (120 mm LH). Estos valores de tamaños de maduración son mucho menores que los obtenidos por Román Cordero (1991) y Gaut y Munro (1983). Las tallas más pequeñas observadas en este estudio podrían estar relacionadas con varios factores. Primero, en este estudio se hizo énfasis en colectar los individuos dentro de la amplitud de tamaño en en la cual se espera que tengan lugar la diferenciación y maduración. Segundo, los peces fueron colectados en áreas diferentes. En los estudios previos la mayoría de las muestras fueron tomadas de la pesquería y, por lo tanto, estuvieron prejuiciadas hacia los individuos más grandes de las áreas de pesca comercialmente importantes. De esta manera, nuestro estudio ha debido tener una mayor facultad para determinar con más precisión los limites de tallas a las cuales el desarrollo gonádico se inicia.

Por otro lado, estos tamaños menores pueden ser el resultado de diferencias en las condiciones ambientales entre los estudios. Numerosos estudios (Pauly 1984, Stearns y Crandall 1984) han mostrado que las variaciones en los factores ambientales pueden afectar el tiempo de desarrollo gonádico en peces. Se ha sugerido (Roff 1984, Stearns y Crandall 1984, Stearns y Koella 1984, Schaefer 1987, Appeldoorn 1989) que tales variaciones son adaptaciones (genéticas o fenotípicas) hacia condiciones ambientales que afectan diferencialmente los patrones de crecimiento y mortalidad. En particular, cuando la mortalidad en adultos es alta y el crecimiento en las etapas tempranas es rápido, se debería esperar una maduración temprana. Esto ocurre en condiciones de elevada presión de pesca.

García Cagide et al. (1986) sugieren, específicamente, que esto podría afectar el desarrollo gonádico en las cachicatas. Así, es también posible que las tallas menores de diferenciación y maduración observadas sean un reflejo de que, al momento de nuestro estudio, la cachicata blanca haya estado sometida a una mayor presión pesquera en relación con los estudios previos. Es de notar que los estudios pasados en Puerto Rico han mostrado que el recurso está siendo sobreexplotado (Stevenson 1978, Appeldoorn y Lindeman 1985, Dennis,1988), y no hay una base para esperar que la presión por pesca haya hecho otra cosa que incrementarse.

En un estudio asociado, 35 cachicatas blancas fueron mantenidas en un tanque grande con fondo natural y alimentadas diariamente. Bajo examinación histológica estos individuos presentaron un desarrollo gonádico más avanzado que otros individuos de tallas similares en el campo (Mateo Pérez 1999). Esto demuestra claramente el potencial para la variabilidad inducida por el ambiente en el inicio y tasa de desarrollo gonádico en esta especie. García Cagide y Claro (1983), García Cagide (1986b, 1987) argumentan que en peces de pequeño a mediano tamaño como son las cachicatas en su mayoría, la diferenciación sexual ocurre antes de la madurez sexual, la cual tiene lugar a temprana edad y esto provee un mecanismo de respuesta a las condiciones ambientales lo cual que se traduce como un mayor avance en el desarrollo gonádico y en la talla de madurez.

Si se toman en consideración los resultados obtenidos por diferentes autores (Román Cordero 1991, Billings y Munro 1974, Mota y Pessoa 1973, García Cagide y Claro 1983, García-Cagide 1987) respecto al tamaño mínimo de maduración en hembras y machos de la cachicata blanca, es difícil asegurar con exactitud cúal de los sexos madura primero ya que las tallas obtenidas varían de lugar a lugar. Es importante destacar, sin embargo, que los tamaños indicados para Puerto Rico (Román Cordero 1991, el presente trabajo) permiten sugerir que los machos inician la madurez sexual a tallas ligeramente inferiores que las hembras; mientras que en los informes de Brasil, Jamaica y Cuba, son las hembras las que parecen madurar primero.

Los resultados de este trabajo pueden ser aplicados en el diseño de planes de manejo de la especie y deben ser tomados en consideración a la hora de establecer tallas mínimas de captura con fines de evitar la sobrepesca del reclutamiento.

AGRADECIMIENTOS

Este trabajo fue apoyado por el Programa de Colegio Sea Grant de la Universidad de Puerto Rico (R/FA-06-3-94). En las colectas de los peces participaron E. Molina, M. Rosado, H. Ruíz, G. López y B. Shank. Queremos agradecer a S. Polofski y al Laboratorio de Histopatología Animal de la Universidad de Rhode Island por su asistencia en el procesamiento de las secciones histológicas.

LITERATURA CITADA

Appeldoorn, R.S. 1989. Phenotypic life-history strategies as an alternative to *r*-*K*-type selection. Pages 149-158 in: J.C. Aldrich (ed.). *Phenotypic Responses and Individuality in Aquatic Ectotherms*. JAPAGA, Ashford, Ireland.

Appeldoorn, R.S. y K.C. Lindeman. 1985. Multispecies assessment in coral reef fisheries using higher taxonomic categories as unit stocks, with analysis of artisanal haemulid fishery. *Proc. of the Fifth Int. Coral Reef Congr.* 5:507-514.

Aquino, C., y J. Infante. 1990. Composición de especies y volúmenes de capturas de nasas Antillanas en Juan Esteban, Barahona, República Dominicana. En: *Reportes del Propescar-Sur. 1994.* **1**:9-1, 9-25.

Asoh, K. 1992. Sexual pattern and reproductive ecology of the fairy basslet, *Gramma loreto*. Thesis MS. Department of Marine Science. Univ. Puerto Rico. 202 pp.

Bagenal, T. y E. Braum. 1981. Eggs and early life history. Pages 165-201 in: *Methods for assessment of fish production in fresh water. I. B. P. Handbook #3*. Blackwell Publications, London. 3rd Ed.

Billings, V.C. y J. Munro. 1974. The biology, ecology and bionomics of Caribbean reef fishes: Pomadasyidae (Grunts). Part 5. *Res. Rep. Zool. Dep. Univ. West Indies*, **3**:1-128.

Claro, R. (ed.). 1994. *Ecología de los peces marinos de Cuba*. Inst. Ocean.; Acad. Ci. Cuba. y Centro de Investigaciones de Quintana Roo. México. 525 pp.

Dammann, A. 1980. Caribbean reef fish: Fish traps and management. *Proc. Gulf Carib. Fish Invest.* **32**:100-105.

Dennis, G.D. 1988. Commercial catch length-frequency data as a tool for fisheries management with an application to the Puerto Rico trap fishery. *Mem. Soc. Cien. Nat. La Salle* **48** (Supl. 3): 289-310.

Darcy G.H. 1983. Synopsis of Biological Data on the Grunts *Haemulon aurolineatum* y *H. plumieri* (Pisces: Haemulidae). *FAO Fisheries Synopsis No. 133*. NOOA Technical Report NMFS cicular 448. 36 pp.

García Cagide, A. 1986. Características de la reproducción del ronco amarillo, *Haemulon sciurus*, en la oriental del Golfo de Batabanó, Cuba. *Rep. Invest. Inst. Oceanol. Acad. Cienc. Cuba* **48**:1-28.

García Cagide, A. 1987. Características de la reproducción del ronco arará, *Haemulon plumieri* (Lacépede), en la región oriental del Golfo de Batabanó, Cuba. *Rev. Inv. Mar.* **8**(3):39-55.

García Cagide, A. y R. Claro. 1983. Datos sobre la reproducción de algunos peces comerciales del Golfo de Batabanó, Cuba. *Rev. Invest. Mar.*, Ser. 8, **3**:39-55.

García Cagide, A. y R. Claro. 1994. Reproducción. Páginas 187-209 en: Claro, R. (ed.). Ecología de los peces marinos de Cuba. *Instituto de Oceanología de la Academia de Ciencias de Cuba y Centro de Investigaciones de Quintana Roo.* Mexico.

Gaut, V.C. y J.L. Munro. 1983. The biology, ecology and bionomics of the grunts, Pomadasyidae. Pages 110-141 in: J. L. Munro (ed.) *Caribbean Coral Reef Fishery Resources. ICLARM Studies And Reviews.* ICLARM, Metro Manila, Philippines.

Hoar, W.S., D.J. Randall, y E.M. Donaldson. (ed.). 1983. Reproduction. Part A. Endocrine Tissues and hormones. Pages 223-224 in: *Fish Physiology.*

Lindeman, K. C. 1986. Development of larvae of the frech grunt, *Haemulon flavolineatum,* and comparative development of the twelve species of Wester Atlantic Haemulon (Percoidei: Haemulidae). *Bull. Mar. Sci.* **39**:673-716.

Luna, L. G. (ed.). 1968. *Manual of Histologic Staining Methods of the Armed Forces Institute of Pathology.* Third Edition. McGraw-Hill Book Company. 258 pp.

Manooch, C.S., III. 1976. Age, growth and mortality of the white grunt, *Haemulon plumieri* Lacépede (Pisces: Pomadasyidae) fron North Carolina and South Carolina. *Proc Annu. Conf. Southeast. Assoc. Fish Wildl.Agencies.***30**:58-70.

Mateo Pérez, J.G. 1999. Función de la madurez sexual en el control de las migraciones ontegenéticas de la cachicata blanca, *Haemulon plumieri* (Pisces: Haemulidae), en La Parguera, Puerto Rico. M.S. Thesis. University of Puerto Rico, Mayagüez, Puerto Rico.

McFarland, W.N., J.C. Ogden and J.N. Lythgoe. 1979. The influence of light on the twilight migration of grunts. *Env. Biol. Fish.* **4**:9-22.

McFarlan, W. N. and Z.M. Hillis. 1982. Observations on agonistic behavior between members of juvenile french and white grunts-family Haemulidae. *Bull. Mar. Sci.* **32** (1):255-268.

Mota, M.I. y L .Pessoa. 1973. Maturacao sexual da biquara, *Haemulon plumieri* Lacepede (Pisces: Pomadasyidae). *Arq. Cien. Mar.* **13**(2):69-76.

Nagahama, Y.1983. The functional morphology of teleost gonads. Pages 223-224 in: W. S. Hoar, D. J. Randall and E. M. Donaldson (eds). *Fish Physiology.*

Pauly, D. 1984. A mechanism for the juvenile-to-adult transition in fishes. *J. Cons.* **44**:280-284.

Posada, J. M. 1996. *Life-history, reproductive biology and sexual pattern of the Creole fish Paranthias furcifer (Valenciennes, 1828) (Pisces: Serranidae).* Ph.D. Dissertation. University of Puerto Rico, Mayaguez, Puerto Rico. 210 pp.

Redding , J. M. y R. Patiño. 1993. Pages 503-537 in: Evans, D.H. (ed.) *The Physiology of Fishes.*

Roff, D.A. 1984. The evolution of life-history parameters in teleosts. *Can. J. Fish Aquat. Sci.* **41**:989-1000.

Román Cordero, A. M. 1991. Estudio sobre la dinámica reproductiva de la cachicata blanca *Haemulon plumieri* (Lacépede, 1802) (Pisces: Pomadasydae). MS tesis. Universidad de Puerto Rico, Mayagüez, Puerto Rico. 120 pp.

Saksena, V. P. y W. J. Richards. 1975. Description of eggs and laboratory- reared white grunt, *Haemulon plumieri* (Lacépede) (Pisces: Pomadasyidae). *Bull. Mar. Sci.* **25**:532-536

Schaefer, K. 1987. Reproductive biology of black skipjack, *Euthynus linaetus*, an Eastern Pacific Tuna. *Inter-Am. Trop. Tuna Comm. Bull.* **19**(2):260 pp.

Stearns, S.C., and R.E. Crandall. 1984. Plasticity for age and size at sexual maturity: a life-history response to unavoidable stress. Pages 13-33 in: G. Potts and R.J. Wootton, (eds.) *Fish Reproduction.* Academic Press, New Work.

Stearns, S.C., and J.C. Koella. 1986. The evolution of phenotypic plasticity in life-history traits: predictions of reaction norms for age and size at maturity. *Evolution* **40**:863-913.

Stevenson, D.K., 1978. Management of a tropical pot fishery for maximum sustainable yield. *Proc. Gulf Carib. Fish. Inst.* **30**:95-115.

Methods and Lessons Learnt in the Application of Ultrasonic Telemetry to Coral Reef Fish Movement Studies

NEWTON ERISTHEE[1], IAN POPPLE[2], HAZEL OXENFORD[1],
and WAYNE HUNTE[1, 2]
[1]*Marine Resource and Environmental Management Programme*
University of the West Indies
Cave Hill, Barbados
[2]*Bellairs Research Institute of McGill University*
St. James, Barbados

ABSTRACT

There is increasing interest in the use of acoustic telemetry to track the movement of medium-sized marine fishes, particularly coral reef species, as a result of recent improvements in technology which have allowed reductions in the size of both receivers and transmitters. However, acoustic tagging and tracking techniques are still at a developmental stage. In this study we use six medium-size (12 - 46 cm FL) reef fish species (*Clepticus parrae*, *Cephalopholis cruentata*, *C. fulva*, *Lutjanus mahogani*, *Ocyurus chrysurus* and *Kyphosus sectatrix)* to investigate appropriate non-injurous capture and *in situ* retention methods, examine suitable transmitter tag placements (internal versus external) using dummy tags, and test the feasibility of simultanously tracking up to 20 individuals in a reef environment. The most appropriate capture methods were species-specific and included use of an Antillean fish trap, barbless hook and line, and small mesh net. Clove oil proved to be an effective and fast-acting anaesthetic. Surgical implantation of transmitters into the body cavity proved more successful than external attachment to the dorsal musculature. Super Glue® was found to be effective in closing the incision wound, with full healing in as little as five days. The use of coded transmitters operating on the same frequency allowed for tracking of many individuals simultaneously. However, when several transmitters were in close proximity, the ability of the receiver to decipher individual transmitter codes was greatly reduced.

KEY WORDS: Acoustic tagging methodology, reef fish movement

INTRODUCTION

Patterns of movement and space utilisation by fishes are important to the understanding of population distribution and community structure (Zeller 1997). Such information is also critical in predicting and understanding the effectiveness of marine reserves (MRs) in conserving adult fish biomass, and in providing adult fish biomass to adjacent fisheries (Russ and Alaca 1996).

Information on the pattern of movement and home range size of adult fish is also important for the successful design of MRs. For example, placement of the reserve relative to the position of different habitats; reserve size; and area to boundary ratio will all affect the degree to which adult fish are conserved or 'leaked' to adjacent fishing areas (Holland et al. 1996, Kramer amd Chapman 1999). Furthermore, movement patterns are likely to vary among different species.

The need for information on adult fish movement is particularly urgent in coral reef areas where the use of MRs is now being favoured over the more traditional management tools for the effective long-term management of sustainable coral reef fisheries (e.g. Bohnsack 1993, Alaca and Russ 1990, Rowley 1994, Russ and Alaca 1996). However, the majority of information on fish movement and habitat utilisation, by coral reef fish species of commercial importance to fisheries, is inferred from conventional mark and recapture studies which involve the use of fishing gear to recapture marked individuals, or rely on visual observations of marked individuals by SCUBA (visual recaptures) (e.g. Corless *et al.* 1997, see Appeldoorn 1997 for review). The numerous constraints of conventional mark and recapture studies are well documented (e.g. Parker 1990, Watson *et al.* 1993, Appeldoorn 1997). Observer presence and experience may greatly effect the data by means of failure to notice, recognise and record the correct code and position of both the tag and/or the individual fish. Additionally, the recapture of a tagged individual in a particular area at a particular point in time does not preclude its presence elsewhere at any time between release and recapture (Winter & Ross 1982). Consequently this method will always provide an underestimation of actual movement (Appeldoorn 1997). Perhaps because of this constraint, adult reef fish are generally considered to be site-attached with low potential for supplementing fished areas adjacent to MR boundaries (Bardach 1958, Corless et al. 1997, Holland et al. 1996). Clearly, more complete data on fish movements are needed before any firm conclusions can be drawn on the extent to which MRs will supply adult fish biomass to adjacent fisheries.

Ultrasonic telemetry, which allows continuous tracking of marked individuals, is an ideal tool with which to address the movement and activity patterns of fishes (Winter and Ross 1982, Zeller in press). Until relatively recently, this technique was used exclusively on large pelagic species because of the large size and high cost of the early acoustic transmitter tags (Nelson 1990; cited in Zeller in press). However, improved technology allowing the production of much smaller and relatively less expensive transmitters now means that this technology can be used on much smaller species, including a wide range of reef fishes. Recent studies have successfully applied ultrasonic telemetry to a number of medium-sized coral reef fishes including *Mulloides flavolineatus* (Holland et al. 1993), *Heamulon plumieri* (Tulevech and Recksiek 1994), *Plectropomus leopardus* (Zeller 1997, 1998, in press, Zeller and Russ 1998) and *Caranx melampygus* (Holland et al. 1996).

This paper outlines the methods and the lessons learned in the application of ultrasonic telemetry to the study of movement of several other medium-sized coral reef fish species.

METHODS AND RESULTS

Capture Methods

Successful tagging studies require efficient, non-injurous methods of capture. Several methods were investigated in this study to capture medium-sized (12 - 46 cm FL) reef fishes (i.e. creole wrasse, *Clepticus parrae*; grasby, *Cephalopholis cruentata;* coney, *C. fulva*; mahogany snapper, *Lutjanus mahogani*; yellowtail snapper, *Ocyurus chrysurus*; and Bermuda chub, *Kyphosus sectatrix)* with the least damage and stress to both the specimen and the environment. The capture methods and results are described below.

Traps — Traditional un-baited Antillean Z-shaped traps (2 m long x 1 m wide x 0.6 m high, with 1.5" wire mesh) set on sand patches within the reef (5 - 15 m deep) were used in an attempt to capture the target species.

This passive mode of capture was unsuccessful for *C. parrae,* which tend to school above the reef, but was successful for the more benthic reef dwellers *C. cruentata, C. fulva, L. mahogani,* and *K. sectatrix* and was also occasionally successful for *O. chrysurus* in areas where densities were high. However, repeated soaks in the same area (i.e. > 4 times in 7 days) resulted in *O. chrysurus* and *K. sectatrix* displaying an avoidance behaviour towards the traps.

Once avoidance behaviour was observed, a more active mode of trap fishing was employed for these species. This involved baiting the trap underwater with stale bread placed close to the entrance, and having a diver lying as still as possible, approximately 2 m from the trap. Bread was then crushed and allowed to float in the water near the trap to attract the fish. Once attracted, the diver then slowly moved closer to the trap, eventually rushing the final 0.5 m with arms wide open. This presented the opening of the trap as an avenue of escape, into which the fish fled. This method was very successful for *K. sectatrix* but did not work for *O. chrysurus.*

Fish were removed promptly from traps, since prolonged retention caused additional stress and often resulted in injury.

Nets — A miniature cast net (2 m diam. 1 cm mesh size) was spread in an area in which the targeted fish were frequently seen. The net was either held by two divers or was left hanging at the mouth of a trench in the reef or between two large coral heads. Two to three divers using SCUBA then attempted to shepherd the fish in the direction of the net. Once a fish swam into the net, the net was folded over by a diver to prevent escape. This method was successfully used to capture *C. parrae* and *L. mahogani* in highly rugose areas of the reef. The rate of success of this method decreased markedly as rugosity of the reef declined. Attempts to capture *K. sectatrix*, *O. chrysurus* and *C. cruentata* using this method were unsuccessful.

Hook and line — Small barbless No. 9 hooks (2/0 O'Shaughnessy) were baited with strips of fresh ballyhoo (*hemiramphus* sp.), flyingfish (*Hirundichthys* sp.) or clupeids, and dangled near the target species by divers equipped with SCUBA. This was the most successful method of capture for *C. cruentata*. Little success was achieved when this method was employed in the capture of the other target species, and the bait was often rapidly removed from the hooks by schools of *Abudefduf saxatilis* (sergeant major) and *Melichthys niger* (black durgon).

Suction gun — A suction gun was constructed from the cylindrical barrel of an old Van Dorn sampling device (10 cm diam, 75 cm length) fitted with a manual plunger constructed from a circular Perspex disc (9.3 cm diam.) and a 1/2" (1.25 cm) PVC pipe handle that fitted snuggly to the interior of the tube, but was free to move up and down. The mechanism of operation is such that when the plunger is depressed and then pulled underwater, a strong suction force is produced which draws the fish into the barrel, facilitating capture. This device, although successful in capturing small-sized benthic reef fishes, was not successful in the capture of any of the target species in this study because of the difficulty in getting close enough to the medium-sized fish for the suction force to be effective.

Retention of Fish

Ex situ retention — Following initial trials of different capture techniques, fish were transported from the reef in sampling coolers containing 40 L seawater and placed in concrete flow-through seawater tanks (0.43 m^3) for investigations of anaesthetic and transmitter tag attachment.

Fish were held without feeding for 24 - 48 hr prior to tagging. During this period they were observed for signs of injury from capture and handling, and a crude index of stress was measured by dividing the concrete tank into 4 imaginery sectors and recording the degree of movement between sectors during periods of 20 minutes for each fish, before and after tagging.

None of the three species investigated (*C. cruentata*, *C. parrae* and *L. mahogani*) showed significant differences in their behaviour before and after tagging. However, the death of two *L. mahogani* and two *C. parrae* before tagging was attributed to stress-related trauma which could not be visually detected.

In situ retention — Once initial trials had been completed, fish were no longer removed from the field, but were retained in underwater shade cages (Figure 1) placed in the reef, for a period of observation and starvation (of atleast 24 hrs) prior to tagging. Only one fish was retained in each shade cage at any one time.

Two different sized shade cages (small or large) were used depending on the size of the fish to be held. These followed the basic design of P. Sikkel[1] and were constructed of green or black shade cloth (the type used to provide 80% shade in plant nurseries) covering a nylon rope and 1/2" (1.25 cm) PVC pipe frame measuring 0.5 m x 0.35 m x 0.35 m for small cages and 1 m x 0.70 m x 0.70 m for large cages. A door comprising black Perspex® was placed in the centre (measuring 0.38 m x 0.18 m for small cages and 0.76 x 0.36 m for large cages) of the top face of the cage and this was hinged with plastic cable ties and fastened with nylon rope.

The shade cage served as a means of isolating the fish from its environment (ensuring a period of starvation) as well as providing a safe and secure place to rest (thereby allowing recovery from the stress of capture). The soft material also served to reduce abrasions or injury to the fish when moving suddenly. The shade cages were weighted down with four 1 kg lead weights as well as tied to the traps to prevent relocation by strong currents, and kept in the reef during use. The shade cages worked well for all species tried (i.e. *K. sectatrix, O. chrysurus* and *C. cruentata).* However removal of *O. chrysurus* from the cages without losing them proved to be very difficult, even with the use of a safety net placed around the door, owing to their powerful acceleration and strong swimming ability.

Anaesthetic Procedure

Ex situ trials — Two anaesthetics, Quinaldine and clove oil (active ingredient 4-ally-2-methoxy-phenol; Anderson *et al.* 1997) were used in *ex situ* trials with *C. cruentata, C. parrae and L. mahogani* in the laboratory holding tanks. Quinaldine was used at a concentration of 0.1 ml/L (2 ml Quinaldine to 20 L of seawater). Clove oil was first dissolved in 70% ethanol (2.4 ml clove oil to 24 ml ethanol) to give a 10% stock solution. This was then added to 20 L of seawater.

Both clove oil and Quinaldine performed equally well at these concentrations in inducing a rapid (in < 2 min) deep anaesthesia in fish of size range 12 - 46 cm FL, from which they recovered in less than 3 minutes.

---

[1] Dr. Paul Sikkel, Department of Biology, Murray State University, KY

However, clove oil was chosen for use in the field because of its non-toxic nature to the environment.

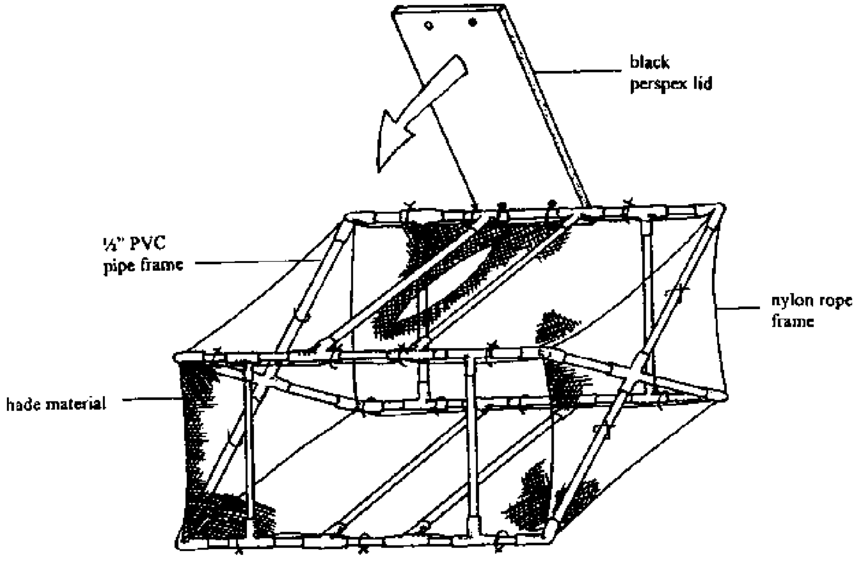
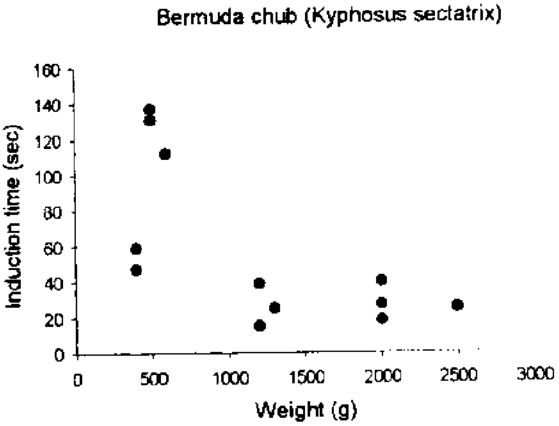


Figure 1. Diagram showing design of the shade cage used for *in situ* retention of reef fish (after P. Sikkel)

Field procedure — Two species, *K. sectatrix* and *C. cruentata* were routinely anaesthetised in the field using the following procedure. The shade cage was first slowly brought to the surface (from depths not exceeding 15 m) by a SCUBA equipped diver, and placed on board the tagging boat (a 5 m inflatable dingy). The fish was then removed from the shade cage and immediately exposed to the anaesthetic by placing it in a sampling cooler containing 20 L of seawater and 24 ml of 10% Clove oil stock solution.

The three induction stages described by Hikasa et al (1986) in Anderson et. al (1997) (stage 1: onset of rapid opercula movement; stage 2: erratic swimming and partial loss of equilibrium; stage 3: total loss of equilibrium) were clearly observed in both species and the onset of each stage was recorded by stopwatch. The time to full induction (stage 3) varied among individuals but was negatively correlated with weight in *K. sectatrix*, (Log_{10} induction time (sec) = 4.035 -

0.802 Log_{10} weight (g); $r^2 = 0.542$, $P = 0.006$) and not significantly correlated to length or weight in *C. cruentata* ($P > 0.05$ in both cases) (Figure 2).



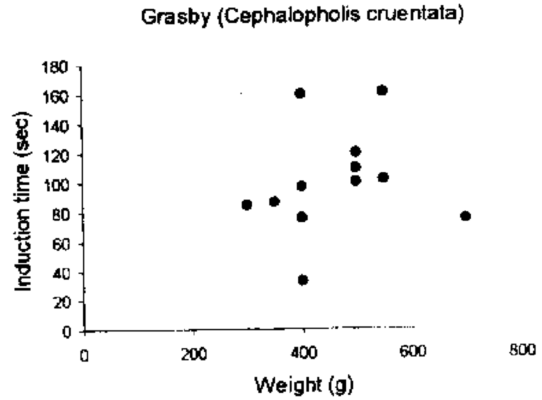


Figure 2. Relationship between fish weight and anaesthetic induction time for the Bermuda chub (*Kyphosus sectatrix*) and grasby (*Cephalopholis cruentata*).

Tag Attachment and Recovery

Dummy tag trials — Both external attachment, through the dorsal musculature, and internal surgical implantation methods of tag attachment were investigated using dummy transmitter tags in initial laboratory-based trials. Dummy tags were made from PVC plastic tubing in two sizes (8 and 11 mm diam., and 18 and 35 mm long respectively) weighted with linseed oil putty and capped at both ends with epoxy resin. The size and weight of the dummy tags approximately mimiced those of the Lotek CAFT8-1 (8 mm) and CAFT11-2 (11 mm) transmitter tags.

After induction to the anaesthetic, external attachment was tried using *L. mohagani*, *C. cruentata*, and *C. parrae*. This involved strapping the tag, with 1 mm (diam.) nylon thread, flush to the body of the fish parallel to the dorsal fins as described by Holland et. al. (1996). Total time for both external and internal tag placements never exceeded seven minutes, inclusive of induction time to anaesthetic.

Based on both laboratory and field observations of *L. mohagani*, swimming ability was not diminished by the externally attached dummy transmitter tag. However, repeated attempts were made to remove the tag by chaffing against rocks, and we recorded a 50 % tag loss rate in as little as four days after release into the field. *C. cruentata* showed obvious signs of distress from external attachment of the dummy transmitter tag, manifested by rapid swimming movements and repeated attempts to remove the tag by chaffing it along the side of the holding tank. *C. parrae* showed good recovery from the handling and tagging procedure, but displayed great difficulty swimming. Although they quickly rejoined a school on release into the field, they were unable to keep up with the rest of the school and were soon left behind.

Surgical implantation of dummy transmitters in the body cavity was tried using L. *mahogani*, *C. fulva* and *C. cruentata* (*C. parrae* were considered too small for internal implantation). This procedure involved making an incision immediately posterior to the right pelvic fin, through which the tag was inserted into the body cavity followed by closure of the wound using Super Glue®.

Internally tagged *L. mahogani*, and *C. cruentata* showed good recovery after surgery, with full healing of incision wounds observed in as little as five days. Swimming ability and feeding both appeared unaffected by the operation. The single *C. fulva* used in these trials survived only for a few hours after the surgery owing to the rupture of an extended stomach during implantation. Internal implantation of transmitters was therefore selected for the field study of fish movement.

Field procedure — Acoustic transmitter tags were surgically implanted into 11 individuals of *C. cruentata* and 11 individuals of *K. sectatrix* in the field using the following procedure.

Firstly, the anaesthetised fish was placed on a clean, flat, shaded surface in preparation for the surgery. Then one end of a 3 m long transparent plastic hose (1 cm diam.) was inserted into the mouth and held in place. The other end of the hose was connected to the drain of an elevated cooler containing fresh seawater, thereby allowing a continuous flow of seawater over the gills during the surgical procedure. The fish was also periodically bathed in seawater to prevent dehydration. In the event that the fish showed signs of recovery before the end of the operation, it was re-exposed to the anaesthetic until complete loss of equilibrium was again observed. The cooler was kept full and details of the operation were recorded.

An incision, slightly larger than the diameter of the transmitter tag, was made equidistant from the ventral fins and the anus and slightly to one side of the central line. This was done with the aid of a size 22 scalpel and a blunt ended seeker. The transmitter was then coated with BNT antibiotic cream and inserted into the body cavity through the incision. The wound was closed with a small quantity of Super Glue®. The fish was subsequently externally marked with a Floy® anchor tag (inserted into the dorsal musculature and anchored between the dorsal spines, midway along the dorsal fin) for easy recognition by divers.

After surgery, the tagged fish was placed in a cooler containing fresh seawater to recover from the anaesthetic and finally transferred to the shade cage, which was subsequently replaced on the reef, using SCUBA. This procedure required a team of three people for smooth running of the operation. Surgical latex gloves were always worn when handling the specimens.

The tagged fish was kept in the shade cage for a further 24 hr during which time the health of the fish could be monitored and the functioning of the transmitter verified. After this the fish was released and tracked.

The species tagged using this method (*K. sectatrix* and *C. cruentata)* showed no apparent discomfort, exhibited fast recovery and retained the transmitters for the duration of the experiment (maximum observed 70 days).

Electronic Equipment

The electronic equipment used in this field study was manufactured by Lotek Marine Technologies Inc., Newmarket, Ontario, Canada.

Transmitter tags — The acoustic transmitter tags were cylindrical-shaped Lotek CAFT 11-2 tags weighing 7.5 g and measuring 11 x 42 mm. Each tag was factory programmed, with a distinct electronic numeric identification code and set to emit an electronic micro pulse every four seconds, once the magnet switch is activated. Lotek estimated transmitter longevity (battery-life) at 90 days.

Recievers — The pulses (acoustic signals) from the transmitter tags were picked up by a directional HPA-O hydrophone fitted with a DAB 45 noise baffle and converted into 76.8 kHz radio frequency signals via a 150 MHz ultrasonic upconverter (UUC). A SRX 400 manually operated telemetry receiver then deciphered these radio signals and displayed signal strength data.

The noise baffle consisted of a windowed sleeve attached over the end of the hydrophone and tightened by two screws. The exposed portion of the hydrophone was sensitive to acoustic signals while the covered portion was dampened by the sleeve. The noise baffle served to focus the sensitive portion of the hydrophone in a cone-shaped pattern, providing directionality to the otherwise omnidirectional hydrophone, thereby increasing the ability of the observer to determine the position of the transmitter tag.

There was little problem in following instructions for the assembly of the equipment, but the operations manual for the equipment could only be deciphered by persons knowledgeable in acoustic technology. However, the customer relations department of Lotek Engineering proved excellent, with a trained technician on call to give a step by step tutorial in the operation of the equipment.

Release and Tracking

Following surgical implantation of the transmitter tag and the recovery period, the fish was released, and tracking commenced immediately. Tracking was conducted from the same 5 m inflatable dinghy that the implantation surgery was performed in. The electronic equipment was assembled in the dingy prior to commencement of tracking (as shown in Figure 3), and dissembled and taken ashore at the end of each tracking session.

The hydrophone was operated by securely fastening it to an oar, which was in-turn then secured to the starboard pontoon of the dingy. The attachment was such that it allowed the hydrophone to be manually rotated 360 degrees when held vertically or 90 degrees when held horizontally. Thus giving maximum directional coverage for detection of acoustic tag signals both in mid-water (vertical position) and within the reef structure (horizontal position).

Once the receiver equipment was set up and the hydrophone was in place, the SRX receiver was set at five gain steps for initial detection of transmitter signals. When a signal was detected, the hydrophone was rotated to find the direction of maximum signal strength (determined either by loudspeaker output or by the signal strength meter on the SRX receiver). The boat was then maneuvered in the direction of the signal whilst the gain was gradually reduced towards zero. A reduction of the gain effectively reduced the maximum detectable range of the tag. Increased signal strength allowed for distance to an individual fish to be determined so that its position could be recorded within a few metres accuracy.

During tracking sessions, the geographical location of the fish was recorded every 15 minutes using visual hand-held compass fixes on at least three known landmarks. In the event that a fish spent a long time at one position a visual check on the habitat was made using snorkel gear or SCUBA. For the more sedentary *C. cruentata*, the fish position was recorded directly onto a field map of the study area. The field map was divided into 20 x 20 m grid squares and showed landmarks, reef depth contours and subsurface features which allowed easy recognition of the different grid squares. For night tracking, landmarks that remained lit throughout the night were used as reference points for compass and sextant readings. Light sticks attached to marked reef features were also used to help in the estimation of fish positions. The initial use of a hand-held Magellan GPS was discontinued since the error incurred by selective satellite availability proved to be approximately 20 m (a distance that could place a transmitter in a separate grid square).

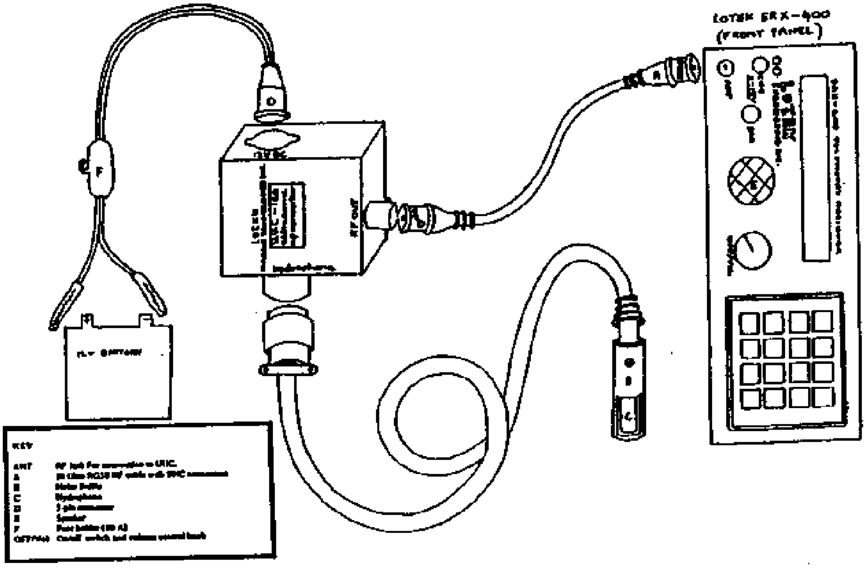


Figure 3. Diagrammatic representation of the assembly arrangement for telemetric receiver equipment used in tracking reef fishes.

Fish were tracked continuously during the first 24 hours after their release. This labour intensive technique involved using two separate teams of researchers, each undertaking six hour shifts.

Subsequently, each of up to 22 fish were tracked for a minimum of four hours every 24 hours. Those four hours were split between two tracking sessions occurring randomly throughout any 24 hour period. Several different fish were often tracked simultaneously on the same frequency, since each transmitter had a unique identification number.

The tracking procedure generally worked well and signals were detected from tags at depths of up to 30 m and over distances of up to 250 m. This was however considerably less than the 500 m horizontal distance given in equipment specifications. Signal strength was found to diminish drastically when fish entered crevasses in the reef.

Simultaneous tracking of several fish on the same transmitter frequency allowed for the investigation of schooling behaviour. However, signal overlap during schooling was high. The acoustic receiver was often unable to identify individual signals and as a result a large number of error readings were displayed. Another constraint was that signal strength appeared to degrade with battery age. In cases where a transmitter tag was activated three to four weeks before another, operation on the same frequency resulted in the signal of the newer tag masking the signal of the older tag even when separated by distances of up to 100 m.

CONCLUSIONS

Effective, non-injurous methods of capture were species-specific, with baited hook and line being most effective for *C. cruentata*, while Z-shaped Antillean traps were more effective for *K. sectatrix, L. mahogani* and *C. fulva*, and a hanging cast net was most effective for the more pelagic *C. parrae*.

Clove oil proved to be an effective anaesthetic with the option of immediate release into the marine environment. Full recovery from the anaesthetic occurred in less than three minutes as reported by Anderson et al. (1997).

External attachment of transmitter tags has been used successfully in several studies (e.g. Tulevech and Recksiek 1994, Begout and Legardere 1995, Holland et al. 1996, Josse et al. 1998). However, it was rejected in favour of surgical implantation in this study after laboratory trials of both methods. Surgical implantation into the body cavity has the advantages of greatly reduced tag loss compared with external attachment, allows a better distribution of the weight near the fish's centre of gravity, and does not create drag forces when the fish is swimming (Mellas et al. 1985). Surgical implantation of transmitter tags into the body cavity proved to be effective in this study for long-term tag attachment in *K. sectatrix* and *C. cruentata*. Only one incident of tag induced mortality was observed out of the 22 fish in which transmitter tags were implanted, although a number of individuals were lost in the latter stages of the tracking study (i.e. after more than 21 days at large). The probability of stomach and/or swim bladder rupture during tag implantation was significantly decreased by ensuring a period of starvation, and a slow accent to the surface prior to tagging.

Successes with surgical tag implantation have been reported for several fresh water and diadromous species (e.g. Mellas et al. 1985, Moore et al. 1990); and for coral reef-associated species (e.g. Colton et al. 1983, Zeller in press). However, in the reef-associated species, immediate release of fish after surgery resulted in mortality due to predation, and therefore a 2 - 3week post-surgery recovery period was suggested to reduce this risk (Zeller in press). The predation-mortality may have occurred because of the relatively substantial incision wounds (2 - 3 cm long) closed by sutures or staples. In this study we found Super Glue® to be effective in closing the small (1 cm) incision wounds without the aid of sutures, and suggest that fish may be safely released 24 hours after tagging. This surgical procedure avoids the problem reported by Tulevech and Recksiek (1994) of exhausting the battery of short-life transmitter tags during post-surgery recovery. Placement of the transmitter tags in the gut was not considered in this study as a result of reported tag losses of up to 60% with this method (Colton and Alevison 1983, Moser et al. 1996).

The use of a mesh shade cage, as suggested by Moore et al. (1990), proved to be a successful means of short-term fish retention. This eliminates the need for seawater flow-through aquarium facilities while in the field and also allowed *in situ* testing of the transmitter tag before release of the fish.

Tracking large numbers of acoustically tagged fish simultaneously proved a monumental task in terms of manpower. Furthermore, transmitter tag signal interference was common with so many tagged fish in relatively close proximity. Reliability of the electronic receiver equipment was also a major issue. Working from a small open boat presented significant challenges for keeping the sensitive equipment dry. Mounting the SRX receiver and UUC in a protective box was not always successful in protecting the equipment from rain and sea spray. This emphasises the need for back-ups at this stage of developing technologies and the need for development of waterproof units.

ACKNOWLEDGEMENTS

This forms part of a larger study receiving funding from numerous sources including a UWI Research Grant to HO, a UWI Postgraduate Research Grant to NE, a NSERC Research Grant to WH, MAREMP, MacArthur/CCA, University of Puerto Rico Sea Grant College Programme (MPRD-12), and CFRAMP. We are extremely grateful for the assistance and co-operation of the St. Lucia Fisheries Division, K. Wolf and his staff at the SMMA, the Anse Chastenet Dive Shop, Frogs Dive Shop, assistants K. Dwyer, C. Wabnitz, D. Pakes, C. Popple, M. Slack and J. Lariviere. We are particularly grateful to B. Wetherbee and M. Sisak for their assistance and technical expertise.

LITERATURE CITED

Alcala, A.C. and G.R. Russ. 1990. A direct test of the effects of protective management on the abundance and yield of tropical marine resources. *J. Cons. int. Explor. Mer* **46**:40-47.

Appeldoorn, R.S. 1997. Dispersal rates of commercially important coral reef fishes: What do tagging studies tell us about potential emigration from marine fisheries reserves? *Proc. Gulf Carib. Fish. Inst.* **49**:54-63.

Bardach, J.E. 1958. On the movement of certain Bermuda reef fishes. *Ecology* **39**(1):139-146.

Begout, M-L and J-P Legardere. 1995. An acoustic telemetry study of seabream (*Sparus aurata* L.): first results on activity rhythm of the environmental variables and space utilisation. *Hydrobiologia* **300/301**:417-423.

Bohnsack, J.A. 1993. Marine reserves: they enhance fisheries, reduce conflicts, and protect resources. *Oceanus* **36**:63–71.

Colton, D.E. and W.S. Alevizon. 1983. Movement patterns of bonefish, *Albula vulpes*, in Bahamian waters. *Fish. Bull.* **81**(1):148-154.

Corless, M., B.G. Hatcher, W.Hunte and S. Scott. 1997. Assessing the potential for fish migration from marine reserves to adjacent fished areas in the Soufriere Marine Management Area, St. Lucia. *Proc. Gulf Carib. Fish. Inst.* **49**:74-98.

Diana, J.S., D.F. Clapp, E. M. Hay-chemielewski, G. Schnicke, D. Siler, W. Ziegler, and R.D. Clark (Jr). 1990. Relative success of telemetry studies in Michigan. *Am. Fish. Soc. Symp.* **7**:346-352.

Holland, K.N., C.G. Lowe, B.M. Wetherbee. 1996. Movements and dispersal patterns of blue trevally (*Caranx melampygus*) in a fisheries conservation zone. *Fish. Res.* **25**:279-292.

Holland, K.N., J.D. Peterson, C.G. Lowe and B.M. Wetherbee. 1993. Movements, distribution and the growth rates of the white goatfish *Mulloides flavolineatus* in a fisheries conservation zone. *Bull. Mar. Sci.* **52** (3):982-992.

Josse, J., P. Bach, and L. Daghorn. 1998. Simultaneous observations of tuna movements and their prey by sonic tracking and acoustic surveys. *Hydrobiologia* **371/372**:61-69.

Kramer, D.L. and M.R. Chapman. 1999. Implications of fish home range size and relocation for marine reserve function. *Environ. Biol. Fish.* **55**:67-79.

Mellas, E.J. and J.M. Haynes. 1985. Swimming performance and behaviour of rainbow trout (*Salmo gairdneri*) and white perch (*Morone americana*): effects of attaching telemetry transmitters. *Can. J. Aquat. Sci.* **42**:488-493.

Moore, A., I.C. Russel and E.C. Potter. 1990. The effects of intraperitoneally implanted dummy acoustic transmitters on the behaviour and physiology of juvenile Atlantic salmon, *Salmo salar* L. *J. Fish Biol.* **37**:713-721.

Moser, M.L., A.F. Olson and T.P. Quinn. 1990. Effects of dummy ultrasonic transmitters on juvenile coho salmon. *Am. Fish. Soc. Symp.* **7**:353-356.

Nelson, D.R. 1990. Telemetry studies of sharks: a review, with application in resource management. Pages 239-256 in: H.L. Pratt, S.H. Gruber, and T. Taniuchi. (eds.) *Elasmobranchs as living resources: advances in biology, ecology, systematics, and the status of the fisheries.* NOAA Tech. Rep. NOAA-NMFS-SWFSC-90.

Parker, R.O. (Jr). 1990. Tagging studies and diver observations of fish populations on live-bottom reefs of the U.S. southeastern coast. *Bull. Mar. Sci.* **43**(3):749-760.

Rowley, R.J. 1994. Case studies and reviews: marine reserves in fisheries management. *Aquat. Cons. Mar. Freshw. Ecosys.* **4**:233-254.

Russ, G.R. and A.C. Alcala. 1996. Do marine reserves export adult fish biomass? Evidence from Apo Island, central Philippines. *Mar. Ecol. Prog. Ser.* **132**:1-9.

Tulevech, S.M. and Recksiek, C.W. 1994. Acoustic tracking of adult white grunt, *Haemulon plumieri*, in Puerto Rico and Florida. *Fish. Res.* **19**:301-319.

Watson, R.A., G.M. Carlos and M.A. Samoilys. 1993. Bias introduced by the non-random movement of fish in visual transect surveys. *Ecological Modeling* **77**:205-214.

Winter, J.D. and M.J. Ross. 1982. Methods of analysing fish habitat utilisation from telemetry data. Pages 273-279 in: N. Armantrout, (ed.) *Proc. Of the symposium on acquisition and utilisation of aquatic habitat inventory information.* Am. Fish. Soc. West Div., Portland, Oregon USA. October 1981.

Zeller, D.C. *Ultrasonic telemetry: Application in coral reef fisheries research. Fish. Bull.* In Press.

Zeller, D.C. 1997. Home range and activity patterns of the coral trout *Plectropomus leopardus* (Serranidae) as determined by ultrasonic telemetry. *Mar. Ecol. Prog. Ser.* **154**: 65 - 77.

Zeller, D.C. and G.R. Russ, 1998. Marine reserves: patterns of adult movement of the coral trout *Plectropomus leopardus* (Serranidae). *Can. J. Fish Aquat. Sci.* **55**: 917-924.

The Age and Growth of Southern Flounder, *Paralichthys lethostigma*, in Louisiana Waters

ANDREW J. FISCHER and BRUCE A. THOMPSON
Coastal Fisheries Institute
Louisiana State University
Baton Rouge, LA 70803 USA

ABSTRACT

Morphometric measurements and otoliths were collected from southern flounder *Paralichthys lethostigma* from a variety of sample sources from Louisiana waters. Transverse sections of otoliths (n = 1286) were examined and opaque zones validated to form once a year in the winter months. Maximum observed age for males was 4 years while that among females was 8 years. Growth was expressed by Vonbertalanffy's growth model as $L_t = 325.65\{1 - e^{[-1.33(t + 0.01)]}\}$ for males and $L_t = 520.14\{1 - e^{[-0.74(t + 0.14)]}\}$ for females where t is age in years and L_t is total length at age t. Growth was shown to be significantly different between males and females with males displaying a faster growth rate than females but a much smaller L_∞.

KEY WORDS: Otolith, growth model, southern flounder

INTRODUCTION

The southern flounder, *Paralichthys lethostigma*, is the largest member of the family Paralichthyidae (Hensley et al. 1984) in the Gulf of Mexico (Henderson-Arzapalo et al. 1988). Southern flounder are distributed from Albemarle Sound, North Carolina, to the Loxahatchee River on the lower eastern coast of Florida. They are absent on the southern peninsular tip of Florida, but occur in the Caloosahatchee River and up the western coast of Florida and around the Gulf of Mexico to northern Mexico (Hoese and Moore 1998, Manooch 1984).

Southern flounder is an important species throughout Gulf coast region. Commercial and recreational landings of *Paralichthys* along Louisiana coasts for 1997 were estimated to be 94,898 lbs. and 319,607 lbs. (personal communication from the National Marine Fisheries Service, Fisheries Statistics and Economics Division). It is the dominant targeted flatfish in the region and is fished mainly using hook and line, gigging, and by trawl. Commercial landings for southern flounder in Louisiana have fluctuated since the 1950s with the highest landings in the mid-1990s at 0.97 million pounds (Louisiana Department of Wildlife and Fisheries 1998). Substantial restrictions have been put on the southern flounder fishery in recent years leading to a decrease in those landings.

The age structure and longevity of a species is important in formulating a proper fisheries management strategy. It is essential to obtain the age structure of fish populations being harvested to effectively monitor the status of those stocks (Williams and Bedford 1974). Age estimation can be accomplished through the use of otoliths. The sagitta, the largest of the three pairs of teleost otoliths is arrowhead shaped in the flounder. Although growth of the otolith is not uniform along all axes, it grows in a radial fashion forming layers of opaque and translucent zones. These zones, or annuli, are often utilized for age estimation; but in order to do so, the periodicity of the annuli must be validated. Validation of this periodicity of annuli formation is as important in fisheries biology as standardizing solutions or calibrating instruments are in other sciences (Beamish and McFarlane 1983).

The few studies conducted on the age and growth of southern flounder suggest that they are a short-lived species. Nall (1979) reported a maximum age of 10 years using whole otoliths, but did not validate his methods. Stokes (1977) used whole otoliths and reported a maximum age of five years for southern flounder. Wenner et al. (1990) used whole otoliths and reported a maximum age of seven years using length frequency data and marginal increment analysis to validate the use of whole otoliths. Interpretation of length frequency data for validation may be suspect, however, because overlapping size classes of cohorts beyond two years (Ross 1988) can complicate age-class designations. Music and Pafford (1984) assigned a maximum age of six years using scales. They attempted to use otolith counts to document the validity of increment counts on scales, although it is unclear whether they used whole or sectioned otoliths.

It appears that a number of flatfish exhibit sexual dimorphism in age and growth rates. Solomon et al. (1987) found that the growth of male and female *Limanda yokohamae* differed; females exhibited a higher growth rate than males of the same age. The same observation was reported for stone flounder, *Kareius bicoloratus* (Uehara and Shimizu 1996) with females reaching a greater size and living longer. Lux (1973) reported that female winter flounder, *Pleuronectes americanus*, grew faster then males after the second year. Lux and Nichy (1969) also stated a similar pattern of growth in the New England yellowtail flounder, *Limanda ferruginea*. Gilbert (1986) reported that there is evidence that *Paralichthys* females reach a larger size then males. Stokes (1977) stated that male southern flounder grew slower than females and did not exceed 320 mm total length where Miller et al. (1991) reported a difference in maximum size between male and female southern flounder with male maximum size at only 68% of females at the same age. Therefore, it may be necessary to generate separate growth curves by sex to properly manage the fishery.

The objectives of this study are to describe the age and growth of southern flounder in Louisiana waters through examination of sagittal otoliths. Growth will be modeled using the VonBertalanffy growth model. Males and females will be compared to determine if southern flounder displays sexual dimorphism in age and growth and if separate models are required.

METHODS AND MATERIALS

Southern flounder used in this study came from a variety of sample sources from Louisiana waters or the Gulf of Mexico off the coast of Louisiana. Multiple sources provided the most reasonable cross section of the estuaries near shore population. Samples were collected at commercial docks in Grand Isle and Leeville, LA from October 1997 to January 1998 (n = 146). The Louisiana Department of Wildlife and Fisheries at the St. Amant Marine Laboratory also collected samples from Grand Terre with the use of a pound net during November and December 1997 (n = 125). An existing flounder data set compiled by Dr. Bruce Thompson of the Coastal Fisheries Institute of Louisiana State University was also used in the analysis (n = 1,134). This data set contains samples from 1987 to 1998 from a variety of sources with the large majority from commercial fish docks (n = 565) and hook and line fishing rodeo tournaments (n = 421).

Fish were weighed (mg), measured (total and standard lengths in mm), sex determined, and otoliths removed. Otoliths were stored in ethyl alcohol to preserve until they were returned to the laboratory for analysis. Otoliths were cleaned of any extraneous tissue and air dried for at least twenty-four hours. Right and left otoliths were then weighed (± 0.01 mg).

A length – weight regression was calculated on $\log_{10}$ transformed data using the model $\log_{10}$ (weight, g) = slope $\log_{10}$ (TL, mm) + intercept. Linear regressions were also calculated for otolith weight (mg) – age using the model otolith weight = age (slope) + intercept. Analysis of Covariance was used to compare sexes for both regressions.

Fish were aged through processing and analysis of 1,286 sagittal otoliths. A number of fish in the data set were not aged due to missing or broken sagittae. Due to the morphological differences between right and left saggitae, The left was chosen for embedding. Otoliths were embedded in a mixture of five parts araldite 8702 epoxy resin to one part hardener 8,700 and left to harden for twenty-four hours. Two transverse sections near the core of the otolith were taken and glued on to glass slides. The better of two sections was polished and inscribed with an identification number, and aged. Sections were read along the medial side of the section along the ventral side of the sulcus groove (Figure 1).

Reader variability was also evaluated; otoliths were viewed and aged by of two independent readers without the knowledge of the date of capture or sample source. Ages were assigned based on annulus count and edge condition.

Edge condition was recorded as opaque or translucent using the criteria of Beckman et al. (1991). Ages were assigned based on a January 1 birth date from Wenner et al. (1990) and data from this study.

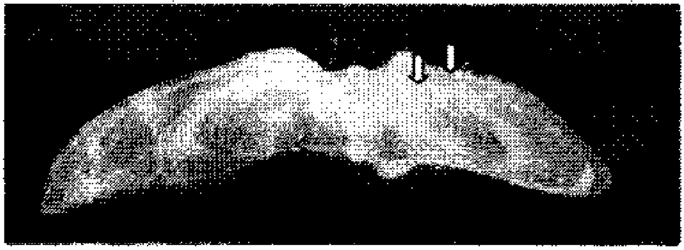


Figure 1. Photomicrograph of a transverse section near the core of a southern flounder otolith. Arrows point to opaque zones or "annuli" counted for age estimation.

Marginal increment analysis and a plot of edge condition by month were used to determine the periodicity of annulus formation in southern flounder otoliths. In addition, length-frequency distributions were plotted by month for young of the year (YOY) and yearlings with and without opaque zones on their otoliths to determine age of first annulus formation.

Length frequency distributions were examined for males and females. Distributions were plotted in 20mm intervals. A Komolgorov-Smirnov two-sample test (Tate and Clelland 1957) was used to test for differences between sexes.

Sex specific VonBertalanffy growth equations were derived from total lengths using nonlinear regression (SAS Institute Inc., 1985) based on the formula:

$$L_t = L_\infty\{1 - e^{[-k(t-t_0)]}\}$$

where t is age in years, and L_t is total length at age t, L_∞ is the theoretical maximum length, k is the growth coefficient, and t_0 is age at which length is zero. Individual fish were not included in the analysis if age or length data was not available. Each model also included 22 unsexed juveniles to provide points at the lower end of the curve. These juveniles ranged in size from 68 mm to 214 mm total length.

The resultant models fitting parameters for both males and females were then combined into one full six-parameter model and compared to a reduced model on the pooled data in which sex was not considered. A likelihood ratio test of the six-parameter and the pooled data models was used to test for differences in the models. Plots of residuals were used to test for normality of the data.

RESULTS

Fourteen hundred and five southern flounders (139 males, 1,201 females, 22 juveniles, and 43 unsexed) were sampled from August 1987 through January 1998. Fish came from numerous sources including commercial and recreational catches from the northern Gulf of Mexico, Louisiana Offshore Oil Port (LOOP) trawls, and the Louisiana Department of Wildlife and Fisheries which provided samples of flounder taken from Barataria Bay. Because not all parameters could be measured for each fish, the numbers of fish included in the different analyses vary.

Total length (TL) frequency distributions plotted by sex were significantly different ($p < 0.05$) (Figure 2). Males ranged in size from 68 mm to 414 mm TL. Males were most abundant at the 280 mm interval with fifty-three percent of all males ranging from 260 mm to 300 mm TL. Females were more abundant at much larger sizes ranging from 189 mm to 764mm TL and were most abundant at the 390 mm interval. Fifty-three percent of all females ranged from 380 mm to 440 mm TL. As expected, males also had a much lower range in body weight then females ranging from 19 g to 936 g. Females ranged in weight from 61 g to 5,953 g.

Regression equations of $\log_{10}$ transformed data were calculated to predict total weight at total length for males and females. Analysis of covariance (ANCOVA) showed no statistical difference between sexes ($p > 0.05$) for slopes; $p > 0.05$ for intercepts). Therefore, a combined length–weight regression was fit for males and females:

$$\log_{10} \text{weight (g)} = 3.21\log_{10}(\text{TL, mm})-5.46 \quad (r^2 = 0.98;\ n = 1236)$$

The slope of 3.21 was significantly different then 3 ($p < 0.0001$).

Opaque rings are easily distinguishable on both the ventral and dorsal sides of the sulcus groove in cross section of southern flounder otoliths. Marginal increment analysis and a plot of otolith edge condition were used to determine the seasonal periodicity of annulus formation. Opaque margins were found in fish caught from the months of January and May and the margins of nearly all samples taken from August through December were translucent. The plot of edge condition corresponded with marginal increment analysis showing the progression of opaque zones from January through May and translucent zones from March through December. Length-frequency distributions for young of the year (YOY) and yearlings indicated first annulus formation as early as 200 mm in length and up to 330 mm. The first annulus appeared on YOY otoliths between the months of January and March.

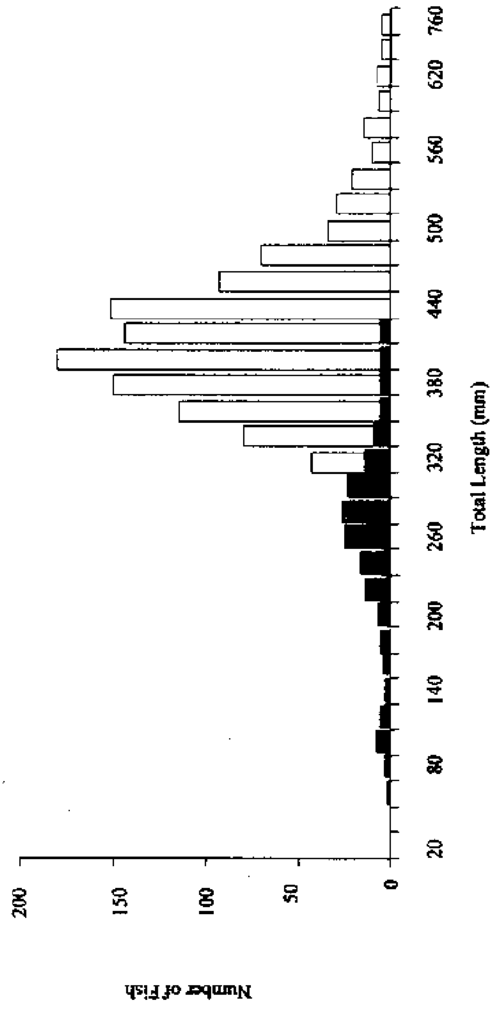


Figure 2. Length frequency distribution of male, female and juvenile southern flounder caught from 1987 to 1998. Males range in size from 68mm to 414mm and are most abundant at 280mm interval. Females range in size from 189mm to 764mm and are most abundant at 390mm interval. Juveniles range from 68 to 309mm and are most abundant at 100mm interval. Males are shown in shaded pattern, females are shown in white, and juveniles are shown in solid black.

Ages were assigned through analysis of 1,286 otoliths. Seven otoliths were excluded from the analysis due to lack of agreement between the two readers. Each of the seven age estimates differed by one year. The two readers agreed on all other otolith annulus counts (N = 1279) or 99.5% of age estimates. The mean coefficient of variation (V) was 0.0011. The mean index of precision (D) was 0.00081 indicating an average error of 0.08 annuli per one hundred counts (Beckman 1989).

A large number of the fish collected were estimated to be two years of age (Figure 3). Forty-six percent of females and thirty-six percent of males fell into this age class. The oldest female was 8.5 years in age and the oldest male was found to be 4.13 years.

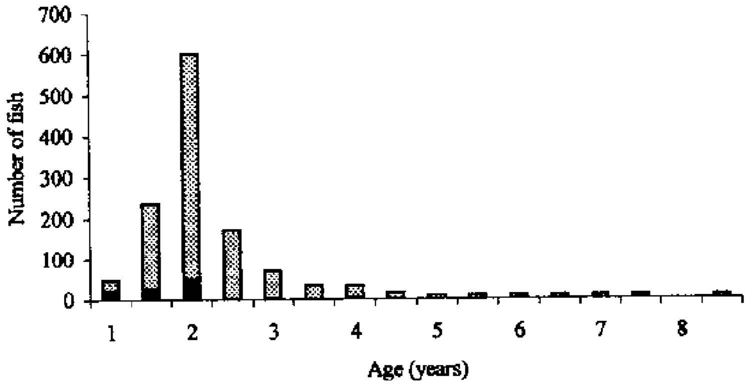


Figure 3. Age frequency distribution for male and female southern flounder sampled from 1987 through 1998. Males reached a maximum age of 4 years and females reached a maximum age of 8.5 years.

Data were fit to a Von Bertalanffy growth model and compared. A likelihood ratio test indicated that there was a significant difference between a full six-parameter Von Bertalanffy growth model and the pooled data growth model ($p < 0.0001$). Therefore, separate growth models were fit for each sex (Figure 4). The Von Bertalanffy growth models derived from total lengths are:

$$\text{Male: } L_t = 325.65\{1 - e^{[-1.33(t+0.01)]}\} \quad (r^2 = 0.68)$$
$$\text{Female } L_t = 520.14\{1 - e^{[-0.74(t+0.14)]}\} \quad (r^2 = 0.52)$$

Plots of residuals indicated normal distribution of the data.

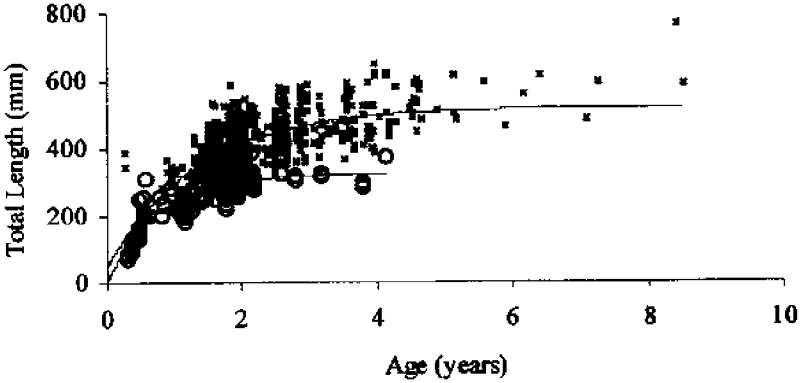


Figure 4. Von Bertalanffy growth models fit for male and female southern flounder sampled from 1987 through 1998. Each model includes 22 unsexed individuals ranging from 68 mm to 214 mm. Gray crosses represent females and hollow circles represent males.

DISCUSSION

The hypothesis that southern flounder display sexual dimorphism in age and growth is supported by length frequency distributions for males and females. Females had a distribution mode of 390 mm and reached up to 764 mm TL while males had a mode of only 280 mm and reached a maximum size of 414 mm. Shepard (1986) reported similar modal lengths with female mode at 358mm and a male mode of 247 mm. Wenner et al. (1990) did not report modal lengths by sex but stated that forty-four percent of aged females were greater than 300mm and few aged males were above 300 mm.

The length-weight regression of $\log_{10}$ transformed data indicated the slope was significantly different from 3.0 ($p = 0.0001$) indicating growth is allometric; weight of the fish increases in relation to its size as length increases. Length-weight regressions on southern flounder produced slopes of 3.14 in South Carolina, 3.09 in Georgia, 3.10 in Florida, and 3.13 in Texas. These slopes were not tested to see if they were significantly different then 3 so it is unclear if there is significant allometric growth in southern flounder from different regions. The higher slope suggests Louisiana southern flounder are more robust then those in previous studies.

Marginal increment analysis and plot of edge condition indicate that one opaque zone is deposited on the otolith between the months of January and May and that annuli on sectioned otoliths may be utilized for accurate age estimation (Barger

1985). A peak of fifty-six percent of individuals with an opaque zone at the growing edge of the otolith is consistent with that of Beckman and Wilson (1995). In a review of 49 studies on north latitude temperate populations using sectioned otoliths, Beckman and Wilson (1995) reported a mean percentage of 65% of individuals with an opaque zone at the growing edge of the otolith.

Analysis of sectioned otoliths of young of the year (YOY) and yearling southern flounder indicated they formed their first annulus as early as 200 mm and up to 330 mm TL. The first annulus began to form in January with all yearlings completing their first annulus by March. These findings are consistent with Powell (1982) who found first annulus formation in the summer flounder to take place between January and March. Stokes (1977) also reported lengths of up to 300 mm by first annulus formation in southern flounder. Wenner et al. (1990) detected no delayed or "lost" first annulus. Therefore this variability in size at first annulus formation is most likely due to differential growth among individuals, which Fitzhugh et al. (1996) found accounted for the broad dispersion of lengths occurring in the first year.

Females live longer than males. Females reached a maximum age of just over eight years while males reached only four years. These findings are close to Wenner et al. (1990) who reported a maximum age of seven years for females and three years for males. Although they employed the use of whole otoliths, their validation techniques of evaluating the edge condition of whole otoliths were similar to this study producing similar maximum age estimates for each sex. Music and Pafford's (1984) maximum age of six years for a female came from a data set of only 198 fish. It is not unexpected that they found a lower maximum age considering that only 11 out of 1,286 (0.009%) aged fish in our data set were five years or older. Nall (1977) reported a maximum female age of 10 years. However, this age estimate seems unlikely when taking into account his invalidated use of whole otoliths. Williams and Bedford (1974) stated that the main source of difficulty in using whole otoliths to age fish is the presence of secondary checks or rings that could be perceived as additional annuli and thus increase your age estimation. In all studies on southern flounder cited here, males have never been aged above three years.

Growth parameters from sex specific curves suggest rapid growth to age two for males and to age three for females. Maximum theoretical size was calculated at 326 mm for males and 520 mm for females. This study predicts more rapid growth and smaller maximum sizes for males and females then reported on southern flounder from South Carolina by Wenner et al. (1990). These parameters and the contrast of the Louisiana and world record suggest that Southern flounder occupying the cold temperate waters of the Atlantic appear to reach greater maximum sizes then those in the warm temperate waters of the Gulf of Mexico. These differences in sizes between the Atlantic and Gulf of

Mexico populations suggest zoogeographic variation in population dynamics of southern flounder. Such variation has been suggested for red drum (Matlock 1987) and Atlantic croaker (White and Chittenden 1977).

These sex specific growth models included 22 unsexed young of the year fish ranging from 68 mm to 214 mm total length. Music and Pafford (1984) stated that sex could not be determined before 130 mm for females or 232 mm for males. Stokes (1977), however, reported that sexual differentiation was not possible for either sex before 170 mm. Juvenile southern flounder have exhibited a capacity for high growth rate relative to other fishes (Fitzhugh 1993). The addition of unsexed juveniles into the growth models may account for the high growth coefficient (k).

ACKNOWLEDGMENTS

We would like to thank Jane Black and all of the employees of USA Fish (Leeville) for providing us access to their commercial flounder catch. We would also like to thank The Louisiana Department of Wildlife and Fisheries for providing us with samples from The St. Amant Marine Laboratory on Grand Terre and to Louisiana Offshore Oil Port (LOOP) for providing samples as well. This research was funded by the Louisiana Department of Wildlife and Fisheries.

LITERATURE CITED

Barger, L.E. 1985. Age and growth of Atlantic croakers in the northern Gulf of Mexico, based on otolith sections. *Transactions of the American Fisheries Society* **114**:847-850.

Beamish, R.J. 1979. New information on the longevity of Pacific ocean perch (*Sebastes alutus*). *Journal Fisheries Research Board of Canada* **36**: 1395-1400.

Beamish, R.J. and G.A. McFarlane. 1983. The Forgotten requirement for age validation in fisheries biology. *Transactions of the American Fisheries Society* **112**:735-743.

Beckman, D.W. 1989. *Age and growth of red drum, Sciaenops ocellatus, and black drum, Pogonias chromis, in the northern Gulf of Mexico.* Louisiana State University unpublished Ph.D. dissertation. 159 pp.

Beckman, D.W., A.L. Stanley, J.H. Render, and C.A. Wilson. 1991. Age and growth-rate estimation of sheepshead, *Archosargus probatocephalus*, in Louisiana waters using otoliths. *Fishery Bulletin* **89**:1-8.

Beckman, D.W. and C.A. Wilson. 1995. Seasonal timing of opaque zone formation in fish otoliths. Pages 27 – 43 in: D. H. Secor, J. M. Dean, S. E. Campana (eds.) Recent Developments in Fish Otolith Research. University of South Carolina Press, Columbia SC. 1995.

Fitzhugh, G.R. 1993. *An individual-based approach to understanding patterns of differential growth and population size structure in juvenile southern flounder (Paralichthys lethostigma).* Ph.D. dissertation, North Carolina State University, Raleigh, North Carolina.

Fitzhugh, D.R., L.B. Crowder, and J.P. Monaghan, Jr. 1996. Mechanisms contributing to variable growth in juvenile southern flounder (*Paralichthys lethostigma*). *Canadian Journal of Fisheries and Aquatic Sciences* **53**:1964-1973.

Gilbert, C.R. 1986. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (South Florida)—southern, gulf, and summer flounders. U. S. Fish Wildl. Serv. Biol. Rep. 82(11.54). U. S. Army Corps of Engineers, TR EL-82-4. 27 pp.

Henderson-Arzapalo, A., R.L. Colura, and A.F. Maciorowski. 1988. Temperature and photoperiod induced maturation of southern flounder. Texas Parks and Wildlife Department, Management Data Series Number 154, 20 pp.

Hensley, D.A. and E.H. Ahlstrom. 1984. Pleuronectiformes: Relationships. Pages 670-687 in: H. Moser, W. Richards, D. Cohen, M. Fahay, A. Kendall, and S. Richardson (eds.) *Ontogeny and systematics of fishes.* Special Publication Number 1. American Society of Ichthyologists and Herpetologtsts. Allen Press. Lawrence, KS.

Hoese, H.D., and R.H. Moore. 1998. *Fishes of the Gulf of Mexico: Texas, Louisiana, and Adjacent Waters*. Second edition, Texas A & M Press, College Station. 422 pp.

Louisiana Department of Wildlife and Fisheries 1998. 1998 Southern flounder stock assessment. 23 pp.

Lux, F.E., 1973. Age and Growth of the Winter Flounder, *Pseudopleuronectes americanus*, on Georges Bank. *Fishery Bulletin* **71**(2): 505-512.

Lux, F.E., and F.E. Nichy. 1969. Growth of yellowtail flounder, *Limanda ferruginea* (storer), on three New England fishing grounds. *International Commision on Northwest Atlantic Fisheries, Research Bulletin* **6**:5-25.

Manooch, C.S., III. 1984. *Fisherman's Guide: Fishes of the southeastern United States*. N. C. State Museum of Natural History. Raleigh. 362 pp.

Matlock, G.C. 1987. Maximum total length and age of red drum off Texas. *Northeast Gulf Science* **9**(1):49-52.

Miller, J.M., J.S. Burke, and G.R. Fitzhugh. 1991. Early life history patterns of Atlantic North American flatfish: likely (and unlikely) factors controlling recruitment. *Netherlands Journal of Sea Research* **27** (3/4):261-275.

Music, J.L., Jr. and J.M. Pafford. [1984]. Population dynamics and life history aspects of major marine sportfishes in Georgia's coastal waters. Georgia Department of Natural Resources, Contribution Series Number 38, 382 pp.

Nall, L.E. 1979. Age and growth of the southern flounder (*Paralichthys lethostigma*) in the northern Gulf of Mexico with notes on *Paralichthys albigutta*. M.S. thesis, Florida State University, Tallahassee. 58 pp.

Powell, A.B. 1982. Annulus formation on otoliths and growth of young summer flounder from Palmico Sound, North Carolina. *Transactions of the American Fisheries Society* **111**:688-693.

Reznick, D., E. Lindbeck, and H. Bryga. 1989. Slower growth results in larger otoliths: an experimental test with guppies (*Poecilea reticulata*). *Canadian Journal of Fisheries and Aquatic Sciences* **46**:108-112.

Ross, S.W. 1988. Age, growth, and mortality of Atlantic Croaker in North Carolina, with comments on population dynamics. *Transactions of the American Fisheries Society* **117**:461-473.

SAS Institute Inc. *SAS User's Guide: Statistics, Version 5 Edition*. Cary, NC: SAS Institute Inc., 1985.

Shepard, J.A. 1986. Spawning peak of southern flounder, *Paralichthys lethostigma*, in Louisiana. Louisiana Department of Wildlife and Fisheries, Technical Bulletin. No. **40**:77-79.

Solomon, G., M. Sano, M. Shimizu, and Y. Nose. 1987. Age and growth of the Pleuronectid flounder *Limanda yokohamae* in Tokyo Bay, Japan. *Nippon Suisan Gakkaishi* **53**(5):711-716

Stokes, G.M. 1977. Life history studies of southern flounder (*Paralichthys lethostigma*) and gulf flounder (*Paralichthys albigutta*) in the Aransas Bay area of Texas. Texas Parks and Wildlife Department 1977, Technical Series #25

Tate, M.W. and R.C. Clelland. 1957. *Non-parametric and Shortcut Statistics in the Social, Biological, and Medical Sciences.* Interstate Printers and Publishers, Inc. Danville, Illinois. P. 93-94.

Uehara, S. and M. Shimizu. 1996. Age and growth of stone flounder *Kareius bicoloratus* in Tokyo Bay, Japan. *Fisheries Science* **62**(6):897-901.

Wenner, C.A., W.A. Roumillat, J. E. Moran, Jr., M. B. Maddox, L. B. Daniel, III, and J. W. Smith. 1990. Southern flounder: *Paralichthys lethostigma*. Pages 2-1 – 2-35 in: *Investigations on the life history and population dynamics of marine recreational fishes in South Carolina: Part 1*.

White, M.L. and M E. Chittenden, Jr. 1977. Age determination, reproduction, and population dynamics of the Atlantic croaker *Micropogonias undulatus*. *Fishery Bulletin* **75**:109-123.

Williams, T. and B.C. Bedford. 1974. *The use of otoliths for age determination.* The Proceedings of an international symposium on the aging of fish. Unwin Brothers Limited.

Distribución y Abundancia de Larvas Velígeras de *Strombus gigas* en Banco Chinchorro Quintana Roo, Mexico

ALBERTO DE JESÚS NAVARRETE
Departamento de Pesquerías Artesanales
El Colegio de la Frontera Sur-Unidad Chetumal
A. P. 424, Chetumal
Quintana Roo, Mexico

ABSTRACT

To determine the distribution and abundance of *S. gigas* veligers, duplicate plankton samples were collected bimonthly from August 1997 to July of 1998 in six sites at Chinchorro Bank. Plankton tows were realized with a conical net 0.5 m diameter and 202 μm mesh opening. 58.52% of veligers were found in rainy season, 35.46% in cold season and only 6.02% were collected in dry season. Higher abundance occurred at stations Penelope, Centro Key and Centro West Key. 89.08% of veligers corresponded to stage I, 3.76% to stage II, 0.25% to stage III, 6.52% to stage IV, and only 0.38% were competent, and suggest that Chinchorro Bank is an important source of veligers. Larval density varied from 0.00093 veligers/10 m^3 at Lobos key in May to 7.42 veligers/10 m^3 at Penelope in August. Larvae abundance was related with juvenile "zero-phase" abundance in the bottom.

KEY WORDS: Caribbean, Chinchorro Bank, distribution, larvae, queen conch, *S. gigas*.

INTRODUCCIÓN

El caracol rosado (*Strombus gigas* Linnaeus, 1758) es un molusco de importancia económica en el Caribe, su captura representa la segunda pesquería, después de la langosta espinosa (*Panulirus argus* Latreille, 1804) (Appeldoorn 1994). Actualmente se reconoce una crisis regional del recurso, derivada principalmente de la sobrexplotación (Stoner et al. 1992). El Convenio sobre Comercio Internacional de Especies en Peligro de Extinción (CITES) vigila el comercio del *S. gigas*, ya que se incluyó en la lista de especies comercialmente amenazadas desde 1992 (Stoner et al. 1996a).

Aún cuando se han implementado diversas políticas internacionales de manejo para el control del recurso, incluso el cierre total de la pesquería en algunas áreas, los resultados no muestran una recuperación sustancial de las poblaciones (Stoner y Ray 1996, Stoner et al. 1996, Stoner et al. 1996a). Este problema se agrava debido a la falta de datos sobre la especie en algunas áreas del Caribe, principalmente aquéllos sobre abundancia de juveniles y adultos y distribución y abundancia de larvas.

Con la disminución de la abundancia de *S. gigas* en el Caribe, muchos laboratorios comenzaron a investigar sobre la generación de larvas a partir de masas de huevo recolectadas en el medio natural, así como sobre la depredación de juveniles (Creswell 1984, Jory e Iversen 1988). Esos trabajos permitieron entre otras cosas, elaborar descripciones detalladas de las larvas en distintos estadios de desarrollo (D´Asaro 1965, Davis et al. 1993). Con estas descripciones se logró identificar a las larvas del plancton del ambiente natural, sin embargo, las evaluaciones de la abundancia y distribución de las larvas de *S. gigas* son recientes e incompletas si se considera lo amplio de la cuenca (Stoner et al. 1992, Posada y Appeldoorn 1994, Stoner y Davis 1997a, 1997b).

Este trabajo es la primera investigación sobre abundancia de larvas en el Caribe mexicano, y pretende probar que Banco Chinchorro es un lugar importante en la generación de larvas.

MATERIALES Y MÉTODOS

Area de estudio

Banco Chinchorro es un complejo arrecifal coralino, (18°36'12"N, 87°18'28"W) que se encuentra dentro de la zona económica exclusiva de México. Se efectuaron colectas bimensuales de agosto de 1997 a julio de 1998, que cubren un ciclo estacional: lluvias de junio a septiembre, nortes octubre a enero y secas de febrero a mayo. Los muestreos se realizaron en seis sitios de la laguna arrecifal: Cayo Lobos (18°23’45.0’’N, 87°21’20.9’’O), Isla Che (18°30’ 12.3’’N, 87°26’13.1’’O), Cayo Centro (18° 33’ 32.7’’N, 87°18’ 24.5’’O), Cayo Centro oeste (18°33’24.1’’N, 87°24’56.6’’O), Penélope (18°42’ 47.6’’N, 87°14´55.5’’O) y Cayo Norte (18° 45’ 28.1’’N, 87°47’ 01.1’’O)(Figura 1).

En cada sitio se efectuaron dos arrastres superficiales de plancton con una red cónica de 0.50 m de diámetro de boca, y una abertura de malla de 200 μm. Los arrastres se efectuaron a una velocidad constante (100 m/s) y tuvieron una duración de 15 minutos. El volumen de agua filtrada por la red se estimó utilizando un flujómetro mecánico General Oceanics, que se ató a la boca de la red.

En el laboratorio se separaron los gastrópodos del resto del plancton y se identificaron las larvas de *S. gigas* de acuerdo con lo planteado por Davis et al. (1993); los valores de abundancia se estandarizaron a 10 m^3. Las larvas se separaron en clases de tamaño, considerando los siguientes estadios: estadio I que corresponde a larvas recién eclosionadas (protoconcha), estadio II caracterizado por la presencia de un pico en la concha de la larva y con aproximadamente 5 días de vida, estadio III velígeras de 530 - 600 μm de longitud de concha de una edad aproximada de diez días, estadio IV los picos de la concha han disminuido y la larva tiene un mayor tamaño, en las larvas competentes carecen de pico y han alcanzado el tamaño para su metamorfosis (Davis et al. 1993). Las larvas se contaron y se midieron utilizando un micrómetro previamente calibrado.

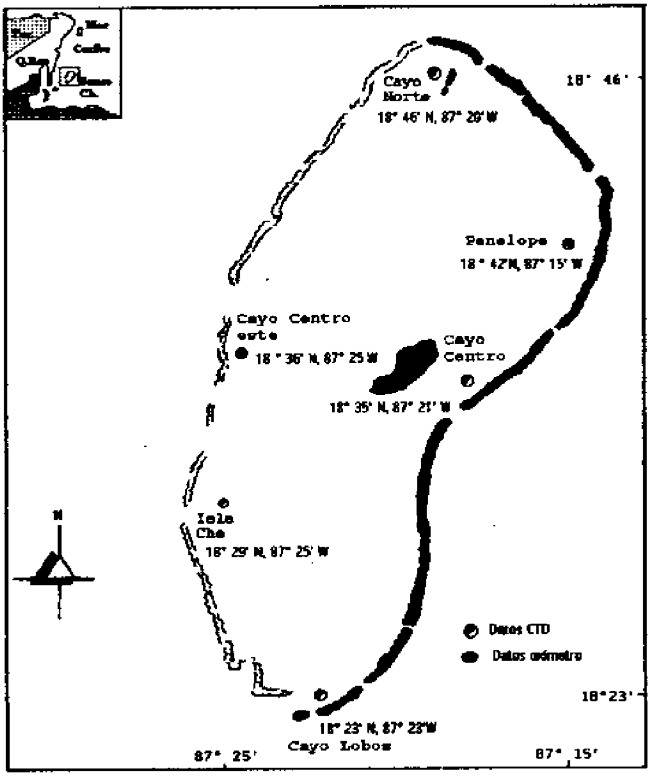


Figura 1. Ubicación del área de estudio.

RESULTADOS

Densidad y distribución de larvas por temporada climática

Se recolectaron 798 larvas de *S. gigas*. La abundancia se distribuyó de la siguiente manera: 58.52% de las velígeras se capturaron en "lluvias" (agosto - julio) con una densidad larval de 8.41 larvas/10 m^3. En "nortes" (octubre - diciembre) se colectó el 35.46% de las larvas, con una densidad de 6.21 larvas/10 m^3. El 6.02% restante se encontró en la temporada de "secas" (marzo - mayo) con una densidad de 1.36 larvas/10 m^3. En lluvias la mayor abundancia se localizó en Penélope (92%, 340 larvas), seguido por Cayo Centro (21.31%, 78 larvas) y finalmente Cayo Norte (9.28%, 34 larvas). La mayor abundancia en "nortes" se ubicó en Cayo Centro (241 larvas) y en Cayo Centro Oeste (23) e Isla Che (17 velígeras). Para "secas" nuevamente Penélope fue el lugar que tuvo la mayor abundancia (36 larvas), mientras que Cayo Centro tuvo (9), Isla Che dos y Cayo Lobos una larva (Figura 2).

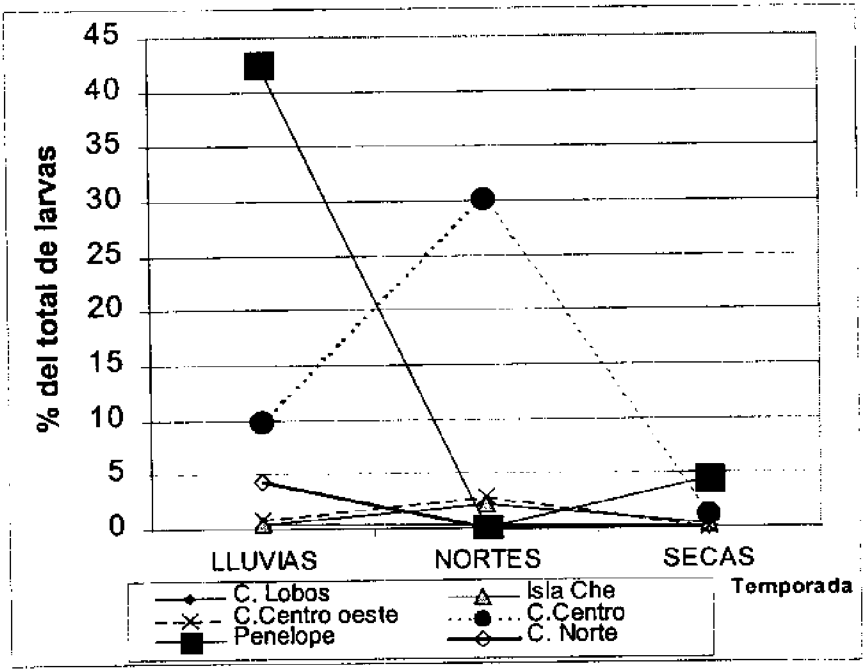


Figura 2. Porcentaje de larvas por temporada climática.

Distribución y abundancia larval por sitio de muestreo

La mayor abundancia se encontró en Penélope con 366 larvas, lo que representó 47.11% del total, Cayo Centro le siguió en importancia con 41.10% de la abundancia, que incluye la abundancia de las muestras diurnas (30.95%) y nocturnas (10.15%). Puede considerarse que éstas dos localidades son las más importantes desde el punto de vista de la abundancia ya que en su conjunto representaron el 88.22% de la abundancia total durante todo el periodo de trabajo.

La distribución de larvas en los otros sitios fue la siguiente: Cayo Norte tuvo el 4.26%, Cayo Centro Oeste el 3.76%, Isla Che tuvo el 2.88%, Cayo Lobos diurno 0.50 y Cayo Lobos noche, sólo el 0.375% (Figura 3).

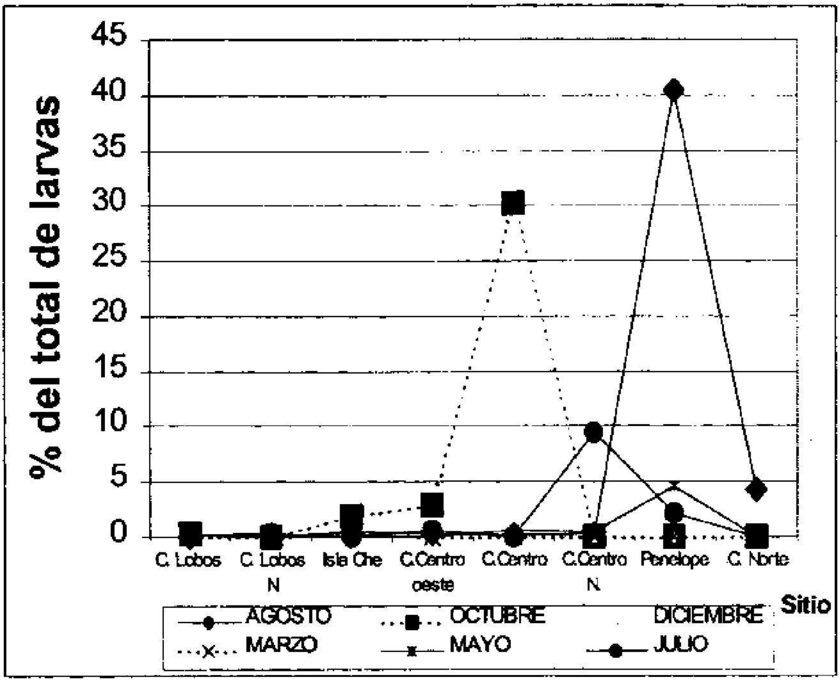


Figura 3. Porcentaje de larvas por sitio de muestreo.

Distribución y abundancia por clases de tamaño

709 larvas (89.07%) correspondieron al estadio I (150 - 450 μm), 30 larvas (3.76%) fueron del estadio II (451 - 700 μm) dos larvas (0.251%) corresponden al estadio III (701 - 950 μm), 52 larvas (6.53%) fueron del estadio IV y tres larvas (0.38%) fueron competentes. La mayor abundancia se localizó en Penélope y Cayo Centro. La distribución se dio en tres picos importantes, con la mayor parte de las larvas ubicadas en los tamaños de 150 a 700 μm (estadios I y II) que se encontraron en prácticamente todos los sitios de muestreo, pero con una mayor abundancia en Penélope y Cayo Centro, las larvas del estadio III se distribuyeron en Cayo Centro oeste y Cayo Centro noche, mientras que las larvas del estadio IV fueron más abundantes en Cayo Centro Noche. Las larvas competentes se encontraron en únicamente en Cayo Lobos (Figura 4).

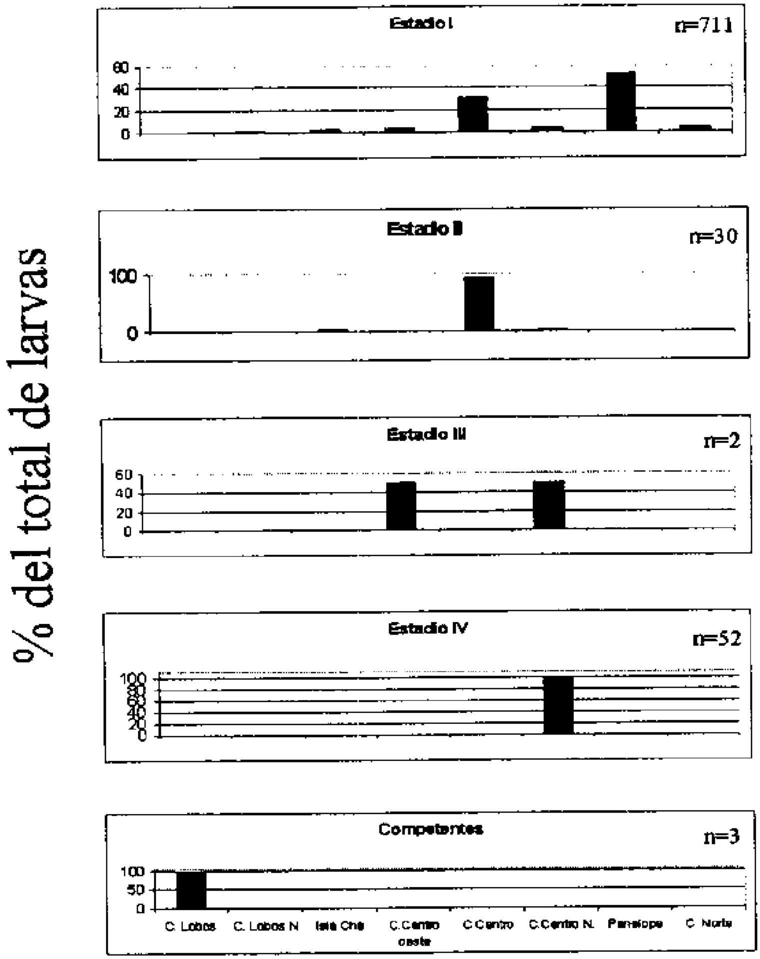


Figura 4. Distribución de las larvas por clases de tamaño.

DISCUSIÓN

Las investigaciones sobre la abundancia larval de *S. gigas* en el Caribe datan de 1989 y han cubierto pocos sitios: Bahamas, Granada y Florida (Stoner et al. 1992, Posada y Appeldoorn 1994, Stoner et al. 1997, Stoner y Davis 1997a, 1997b).

En Bahamas se encontró una densidad máxima de 4.16 larvas/10 m^3 y una mínima de 0.04 larvas/10 m^3. La mayor cantidad de larvas se encontró en Junio y estuvo relacionada con sitios de mayor cantidad de adultos, aunque no se descartó que existiera un arribo de larvas de la plataforma de Exhuma (Stoner et al. 1992). Las abundancias de larvas encontradas en Chinchorro fueron muy similares (7.42 velígeras/10 m^3) y ocurrieron en la temporada con temperaturas más cálidas: agosto, octubre y julio.

Posada y Appeldoorn (1994) cubrieron una zona amplia de Granada y Venezuela. Colectaron muestras dentro y fuera del arrecife y encontraron que la densidad en aguas oceánicas fue baja (0.20 ± 0.251/10m^3, comparada con las zonas protegidas 0.41 ± 0.45 larvas/10m^3. El máximo valor se encontró en aguas de Los Roques, una reserva marina en Venezuela con una densidad de 1.22 larvas/10m^3. Las tallas que dominaron las muestras estuvieron entre 244 a 780 µm, que corresponde con larvas recién eclosionadas y del estadio III, la mayor abundancia ocurrió en julio.

Un estudio comparativo en Florida y Bahamas (Stoner et al. 1996) mostró que en Florida la densidad larval varió de 0.27 ± 0.36 a 0.91 ± 0.69 larvas/10 m^3. La mayor abundancia se presentó en junio y se relacionó con eventos hidrográficos como la temperatura y los vientos. Fue notorio la presencia de una mayor cantidad de larvas de tallas cercanas a la competencia que larvas recién eclosionadas (500 µm) y relacionaron este proceso con el transporte de larvas desde, México, Belice o Cuba, y procesos oceanográficos como giros a mesoescala en la corriente de Florida (Stoner et al. 1996, Kinder 1983). En Chinchorro, se encontraron larvas competentes sólo en Cayo Lobos y esto podía indicar que esas larvas provienen de lugares situados fuera del Banco.

En Bahamas, la densidad larval varió de 0.756 ± 0.57 larvas/10 m^3 a 3.25 ± 2.16 larvas/10m^3. La mayor abundancia (93%) correspondió con larvas mayores de 900 µm de longitud de concha, encontrándose presentes las tallas intermedias lo que sugiere un desarrollo total en el Banco de Bahamas (Stoner et al. 1996), un aspecto similar ocurrió en Chinchorro, en donde el 89.07% de las larvas fueron de los estadios I y II, pero con una presencia de todas las tallas aunque en densidades bajas, lo que sugiere nuevamente un desarrollo total en Chinchorro.

Se ha sugerido una dependencia del flujo larval a nivel del gran Caribe, ya que una velocidad de corriente de 0.8 m/sec y una duración del periodo larval de 2 a 4 semanas en *S. gigas* (Davis et al. 1993) es suficientemente largo para transportar a las larvas desde los sitios de producción en Cuba, México, y posiblemente de Belice y Honduras hasta Florida (Stoner et al. 1997).

Nuevas investigaciones realizadas en Bahamas entre 1992 a 1994, indicaron que la densidad larval promedio varió de 0.26 a 4.46 larvas/10m^3, con un máximo de 7.44 larvas/10 m^3 (Stoner y Davis 1997a). La mayor abundancia se presentó en los arrastres diurnos y tuvieron relación con el comportamiento positivo de las larvas hacia la luz (Barile et al. 1994) en Chinchorro, el 89.48% de las larvas se recolectaron durante el día y sólo el 10.52% fue recolectada en la noche, lo que refuerza la idea de que las larvas tienen una relación directa con la luz (Barile et al. 1994).

Las densidades en Chinchorro variaron de 0.0093 larvas/10m^3, más bajas que las encontradas en Florida en la temporada de máxima reproducción (Stoner et al. 1997), a 7.42 larvas/10m^3 similar a la densidad encontrada en Bahamas (Stoner y Davis 1997a). La mayor abundancia larval se encontró en los meses con temperaturas cálidas (> 28°C) y velocidades de la corriente bajas, o traducida a condiciones ambientales, el mar en calma. Este mismo efecto fue observado para larvas recién eclosionadas en Bahamas y se asociaron con condiciones de calma y máximos valores de temperatura (34°C) (Stoner et al. 1994, Stoner y Davis 1997a).

Los sitios con poblaciones de juveniles y adultos sanos o "naturales" reciben entregas o suministros regulares de larvas en altas densidades. En el caso de Quintana Roo y a pesar de que en Chinchorro ha disminuido la abundancia de juveniles y adultos, presentó una mayor densidad larval (0.0009 a 7.42 larvas.10m^{-3}) que la costa sur (1.4 ± 2.85 larvas/10m^3) en donde las poblaciones de caracol han sido drásticamente disminuidas (de Jesús-Navarrete y Oliva-Rivera 1997). Este mismo efecto se ha observado en sitios desprovistos de juveniles y adultos como Florida (Glazer y Berg 1994, Stoner et al. 1996b).

LITERATURA CITADA

Appeldoorn, R.S. 1994. Spatial variability in the morphology of queen conch and its implications for management regulations Pages 145-157 in: R.S. Appeldoorn y Q. Rodríguez (eds.) *Queen Conch Biology, Fisheries and Mariculture.* Fundación Científica Los Roques, Caracas, Venezuela. 358 pp.

Barile, P.J., A.W. Stoner y C.M. Young. 1994. Phototaxis and vertical migration of the queen conch (*Strombus gigas* Linné) veliger larvae. *J. Exp. Mar. Biol. Ecol.* **183**:147-162.

Creswell, L. 1984. Ingestion, assimilation, and growth of juveniles the queen conch *Strombus gigas* Linné, fed experimental diets. *Jour. Shell. Res.* 4(1):23-24.

D'Asaro, C.N. 1965. Organogenesis, development, and metamorphosis in queen conch *Strombus gigas*, with notes on breeding habits. *Bull. Mar. Sci.*

15(2): 359-416.

Davis, M.C., Bolton, C. y A.W. Stoner. 1993. A comparison of larval development growth, and shell morphology in three Caribbean *Strombus* species. *The Veliger* **36**(3):236-244.

de Jesús-Navarrete, A. y J.J. Oliva Rivera. 1997. Densidad, crecimiento y reclutamiento del caracol rosado *Strombus gigas* L. en Punta Gavilán, Quintana Roo, México. *Rev Biol. Trop.* **45**(2):797-801.

Glazer, R.A. and C.J. Berg. Jr. 1994. Queen conch research in Florida An Overview, Pages 79-96 in: Appeldoorn, R. S. y B. Rodriguez. (eds). 1994. *Biología, Pesqueria, Cultivo y Manejo del caracol Strombus gigas*. Fundación Científica Los Roques, Venezuela. 358 pp.

Jory, D.E. and E.S. Iversen. 1988. Shell strength of queen conch, *Strombus gigas* L.: Aquaculture implications. *Aquaculture and Fisheries Management* **19**:45-51.

Kinder, T.H. 1983. Shallow currents in the Caribbean sea and Gulf of Mexico as observed with satellite tracked drifters. *Bull. Mar. Sci.* **33**(2):239-246.

Posada, J. and R.S. Appeldoorn, 1994. Preliminary observations on the distribution of *Strombus* larvae in the eastern Caribbean. In: R.S. Appeldoorn and B. Rodríguez (eds.) *Queen Conch, biology, fisheries and mariculture*. Fundación Científica, Los Roques, Caracas Venezuela. 356 pp.

Stoner, A.W. and M. Davis. 1997a. Abundance and distribution of queen conch veligers (*Strombus gigas*) in the central Bahamas I Horizontal patterns in relation to reproductive and nursery grownds. *Jour. Shell. Res.* **16**(1):7-18.

Stoner, A.W. and M. Davis. 1997b. Abundance and distribution of Queen Conch veligers (*Strombus gigas* Linné) in Central Bahamas: II Vertical patterns in near shore and depth water habitats. *Jour. Shell. Res.* **16**(1):19-29.

Stoner, A.W., N. Metha, and T.N. Lee. 1997. Recruitment of *Strombus gigas* veligers to the Florida keys Reef tract Relation to hydrographic events. *Jour. Shell. Res.* **16**(1):1-6.

Stoner, A.W., R. Glazer, and P. Barile. 1996. Larval supply to queen conch nurseries: Relationships with recruitment process and population size in Florida and the Bahamas. J *Jour. Shell. Res.* **15**(2):404-420.

Stoner, A.W., M. Ray, R.A. Glazer, and K.J. McCarthy. 1996a. Metamorphic responses to natural substrata in a gastropod larvae: decisions related to postlarval growth and habitat preference. *J. Exp. Mar. Biol. and Ecol.* **205**:229-243.

Stoner, A.W.,R. A. Glazer, and P. J. Barile. 1996b. Larval supply to queen conch nurseries: Relationships with recruitment processes and population size in Florida, and the Bahamas. *Jour. Shell. Res.* **15**(2):407-420.

Stoner, A.W. and M. Ray. 1996. Queen conch, ***Strombus gigas***, in fished an unfished locations of the Bahamas: Effects of a marine fishery reserve on adults, juveniles, and larval production. ***Fish. Bull.*** **94**:551-565.

Stoner, A.W., M.D. Hanisak, N.P. Smith, and R.A. Armstrong. 1994. Large Scale distribution of queen conch nursery habitats: implications for stock enhancement. In. Appeldoorn, R.S. and Q. Rodriguez (eds). ***Queen Conch Biology, Fisheries and Mariculture.*** Fundación Científica los Roques, Caracas Venezuela. 356 pp.

Stoner, A.W., V.J. Sandt and I.F. Boidron-Metairon. 1992. Seasonality in reproductive activity and larval abundance of queen conch ***Strombus gigas. Fish. Bull. U.S.*** **90**:161-170.

Determinación Electroforetica del Patron Proteico de Tres Poblaciones del Caracol Rosado, *Strombus gigas*, en las Costas de Quintana Roo y el Uso de un Software de Imágenes para su Analisis

JORGE ARTURO TELLO CETINA y LUIS ALFONSO RODRÍGUEZ GIL
Instituto Tecnológico de Mérida
División de Estudios de Posgrado e Investigación
Av. Tecnológico S/N. AP 9-11
Mérida, Yucatán. México

RESUMEN

La determinación del proteinograma de muestras de músculo de caracol rosado, *Strombus gigas*, obtenidas en Isla Mujeres, Punta Allen, y Banco Chinchorro en la zona norte, centro y sur del estado de Quintana Roo, se realizó por medio de la técnica de electroforesis en geles de acrilamida (PAGE) al 10 % en medio disociante (SDS) con el propósito de establecer la posible diferenciación y separación geográfica de poblaciones del molusco.

El análisis visual de los proteinogramas presentó similitud en el número de fracciones proteicas para los sitios de Punta Allen e Isla Mujeres y diferencias entre estos sitios y Banco Chinchorro, sin embargo al someter los proteinogramas al análisis digital del programa de PROANA, se estableció la diferencia entre machos y hembras, determinándose 12 fracciones para los machos y 13 fracciones para las hembras en Punta Allen e Isla Mujeres y de 15 y 16 fracciones para machos y hembras en Banco Chinchorro.

Los resultados obtenidos en este trabajo nos permiten establecer lo valioso del software PROANA como herramienta para el análisis digital de las fracciones proteicas reveladas en los proteinogramas obtenidos electroforeticamente que en forma visual no pueden ser detectadas, además de reducir los costos y tiempos de trabajo en el análisis.

La posible separación geográfica de las poblaciones del molusco en el caribe mexicano, con las reservas que el método utilizado presenta, se puede considerar en función de los resultados aquí obtenidos.

PALABRAS CLAVE: *Strombus gigas*, SDS, electroforesis, análisis de imágenes

ABSTRACT

The determination of the proteinogram of samples of muscle of the pink snail, *Strombus gigas*, obtained in Isla Mujeres, Punta Allen and Banco Chinchorro in the north area, center and south of Quintana Roo's state, was carried out by means of the electrophoresis technique in acrylamyde gels (PAGE) to 10% between disociante (SDS) with the purpose of establishing the possible geographical differentiation between populations.

The visual analysis of the proteinograms indicated similarity in the number of proteins fraction for samples taken from Punta Allen and Isla Mujeres and differences between these places and Banco Chinchorro. However, when the proteinograms were subjected to digital analysis (PROANA), the difference settled down among males and female was determined 12 fractions for the males and 13 fractions for the females in Punta Allen and Isla Mujeres and of 15 and 16 fractions for males and females in Banco Chinchorro.

The results of this work provide an evaluation of the efficacy of computer software(PROANA) as a tool for digital analysis of protein fractions revealed in proteinograms that cannot be detected visually, besides reducing the costs and times of work in the analysis.

The possible geographical separation of the populations of *Strombus gigas* in the Mexican Caribbean, with the reservations that the used method presents, should be considered in the interpretation of the results reported in this study.

KEYWORDS: *Strombus gigas*, SDS, electroforesis, image analysis

INTRODUCCION

Estudios poblacionales de especies marinas comercialmente importantes son de interés teórico para los biólogos y de valor practico para los manejadores de pesquerías, sin embargo, la estructura poblacional está influenciada por él concepto mismo de población, ya que en un sentido amplio una población puede ser definida desde una perspectiva biológica la cual implica algún nivel de aislamiento reproductivo, o bien desde la perspectiva pesquera a la cual le concierne una descripción practica de un grupo de organismos explotados en un área especifica, siendo en cualquier caso el nivel y numero de distinciones de la (s) población (es) de interés primario (Kinsey et al. 1994).

El problema se agudiza cuando se tiene como característica de los organismos que componen a la población el desplazamiento de estos en alguna etapa de su vida por medios totalmente independientes, caso de organismos que poseen una fase planctónica, en su dispersión y desplazamiento se rigen por el tiempo de permanencia en la columna de agua y del sistema de corrientes que priven en el área de influencia de su desarrollo (Ayre et al. 1995).

Strombus gigas, caracol marino ampliamente distribuido en el área del caribe, es de importancia capital para los pescadores del área al sustentar estos su economía en la captura de este molusco. A raíz del intenso esfuerzo de pesca a la que esta sometido este recurso, diversas medidas de manejo se han implementado con el propósito de restringir su explotación, así como su eventual desaparición de la zona del caribe mexicano, estas medidas, empíricas la mayoría de las veces, no se sustentan en parámetros validos propiciando con ello ambigüedades en los datos de captura y en la aplicación de las medidas de manejo.

La técnica de electroforesis en geles de poliacrilamida, como una herramienta de separación, nos permite establecer la diferencia entre los patrones electroforeticos de proteínas entre especies y/o poblaciones para establecer una huella de cada una. No obstante, los resultados obtenidos en los proteinogramas la mayoría de las veces suelen tender a propiciar confusión en su interpretación debido al traslape de las fracciones o bien al elevado numero de fracciones presentes. Mediante la digitalización de imágenes estos problemas se han podido minimizar permitiendo al investigador contar con un elemento con el que se obtenga una mejor resolución y definición de las fracciones proteicas reveladas, además de facilitar el trabajo de análisis de las mismas (Gordillo et al. 1992).

El objetivo de este trabajo fue establecer la posible separación geográfica de poblaciones del caracol rosado *Strombus gigas* en tres sitios de captura en las costas de Quintana Roo, utilizando la técnica de electroforesis en medio disociante, SDS, (Corzo et al. 1984) y la utilización de un software de análisis de imágenes para el estudio de las fracciones proteicas reveladas por medios de digitalización de imágenes como un argumento de apoyo para la implementación y aplicación de medidas de manejo de este recurso marino.

METODOLOGIA

La colecta de los organismos se realizó en tres sitios en las costas del estado de Quintana Roo, elegidos por ser considerados zonas tradicionales de captura de *Strombus gigas;* Zona norte, Isla Mujeres; Zona centro, Punta Allen y Banco Chinchorro en la zona sur.

Por medio de buceo se obtuvieron 50 organismos, de todas las tallas, en cada sitio de colecta con ayuda de pescadores de las zonas y previa autorización de la SEMARNAP. El método de captura se hace necesario utilizarlo debido a que los organismos en algunos casos se encuentran a profundidades de 40 metros.

Aproximadamente 1 g de músculo, se utilizo para la extracción de proteínas en un buffer conteniendo Tris-HCl 0.1 M y pH 6.8, SDS al 10 %, Azul de Bromofenol al 1 % y 2- Mercaptoetanol al 5 %.

Geles de poliacrilamida homogéneos al 10 % en medio disociante, SDS, fueron corridos en una cámara de electroforesis vertical (Laemmli, 1970), colocada en un refrigerador a 4 °C, utilizando el buffer Tris – Glicina – SDS 0.5 M con pH 8 y 150 V durante 4 horas. Terminado el corrido el gel fue lavado y fijado en metanol al 30 % y teñido con azul de Coomasie al 0.5 % durante toda la noche. Se utilizó un software de análisis de imágenes denominado "PROANA", para elaborar el histograma de frecuencias de las fracciones reveladas con el objetivo de corroborar si el número de fracciones establecidas y determinadas visualmente coincidían con el número determinado por el software.

RESULTADOS Y DISCUSIÓN

En la naturaleza prácticamente todas las especies de animales y plantas existen como un mayor o menor numero de poblaciones completa o parcialmente aisladas, propiciando con ello la aparición de subespecies geográficas y subgrupos menores.

La identificación morfológica de estas poblaciones muchas veces no satisface los requerimientos establecidos ni muchos menos arrojan resultados confiables, por lo que se convierte esto en un gran problema de sistemática en primera instancia y de manejo en segundo lugar.

Con la implementación de la electroforesis en la época de los cincuentas, su aplicación en la determinación de biomóleculas y la relación de estas con el aspecto genético de los organismos se pudo establecer la identificación de especies cercanas de organismos y mejorar los resultados taxonómicos anteriormente obtenidos (McAndrew and Majumdar 1983).

Los resultados obtenidos al determinar y cuantificar el proteinograma de diferentes organismos se mejoraron cuando se utilizó el agente disociante SDS como un elemento de separación de las diferentes fracciones o subunidades que componen a una proteína (Weber y Osborn 1967) y permitió asimismo la determinación del peso molecular de las mismas (Kukatla 1996) al considerar que el tamaño efectivo de las proteínas era directamente proporcional a la carga proporcionada por la cantidad de SDS utilizado en recubrir a la proteína para su posterior separación (Rybiki and Purves 1998).

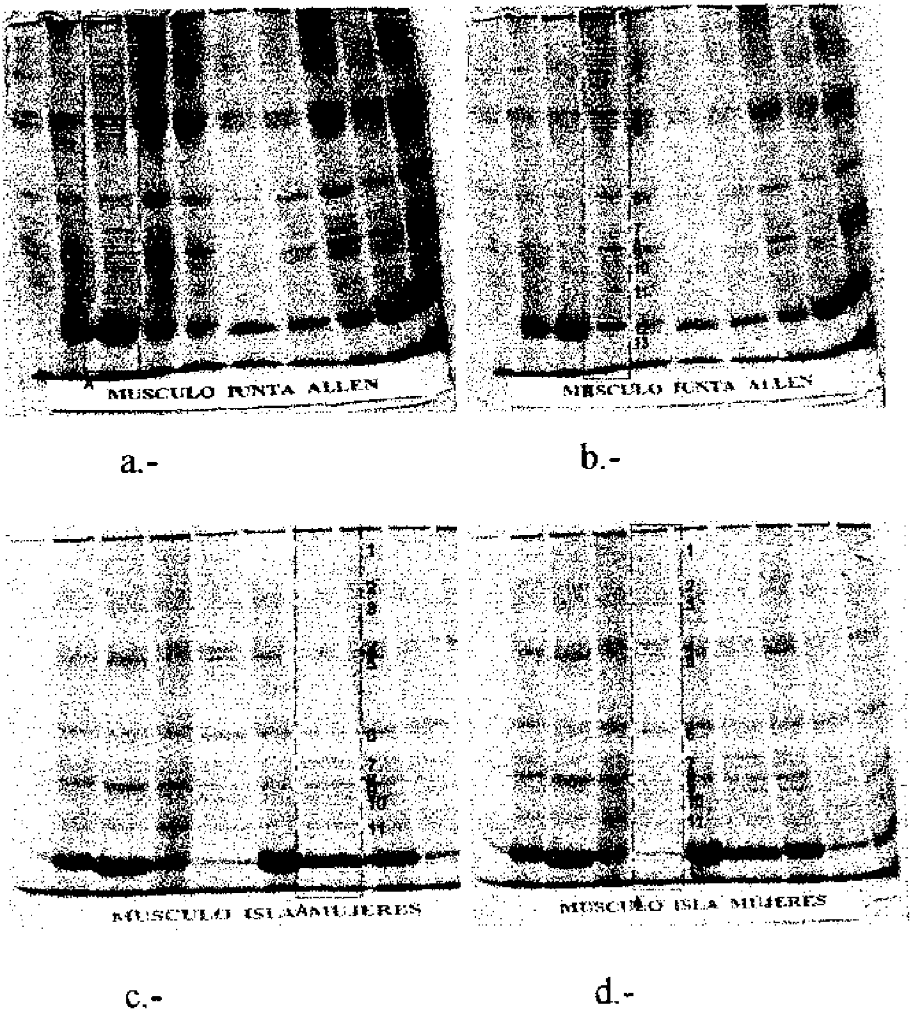
El análisis visual del proteinograma estimo un numero idéntico de fracciones para todas las muestras en cada uno de los sitios analizados, sin embargo al analizar la imagen del proteinograma con el programa PROANA, se pudo establecer que existían diferencias en el número de fracciones en cada sitios y entre los sitios, (Tabla 1) específicamente en lo referente a machos y hembras.

Tabla 1. Número de fracciones proteicas determinadas en músculo de *Strombus gigas* en tres sitios del Caribe Mexicano

Sitio	Numero de Fracciones	
	Machos	Hembras
Isla Mujeres	12	13
Punta Allen	12	13
Banco Chinchorro	15	16

En todos los proteinogramas analizados digitalmente se pudo establecer la presencia de una fracción de más en las columnas correspondientes a las muestras de hembras.

En las imágenes obtenidas de las muestras de Punta Allen e Isla Mujeres se observa la presencia de 12 (Figura 1a y c) y de 13 fracciones (Figura 1b y d), respectivamente.



a.-

b.-

c.-

d.-

Figura 1. Imágenes de proteinogramas. A- Muestras Allen machos 12; B. Muestras Allen hembras 13; C. - Muestras Isla machos 12; D. - Muestras Isla hembras 13.

Al analizar los polígonos de densidad obtenidos del análisis digital de las imágenes (Figura 2, a - d), se determino la presencia de 1 fracción de más en las muestras correspondientes a las hembras.

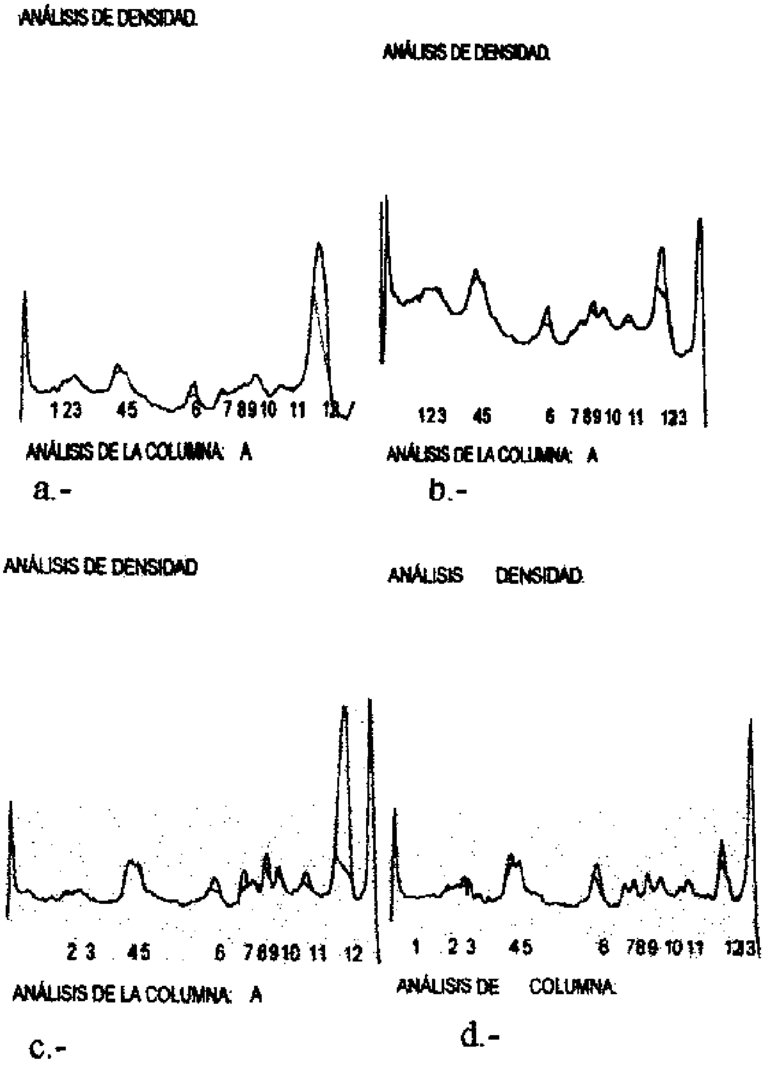
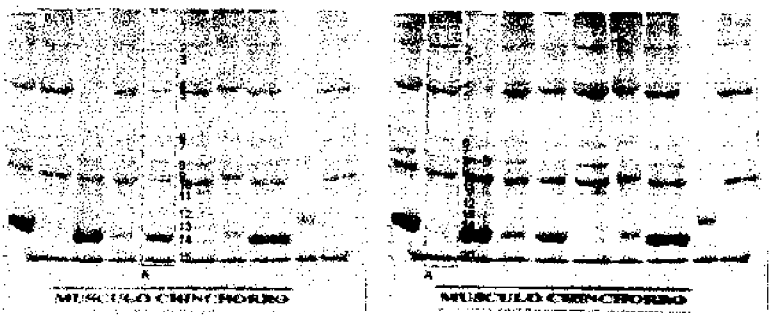


Figura 2. Polígonos de densidad. A. - Muestras Allen machos 12; B. - Muestras Allen hembras 13; C. - Muestras Isla machos 12; D. - Muestras isla hembras 13.

El mismo problema detectado en los sitios de Allen e Isla se presento en Banco Chinchorro, ya que los proteinogramas (Figura 3, a - b) señalaron la presencia de 15 fracciones, pero al editar la imagen del análisis de densidad se pudo establecer la presenciade 16 fracciones en las muestras correspondientes a las embras (Figuraa 4, a - b).

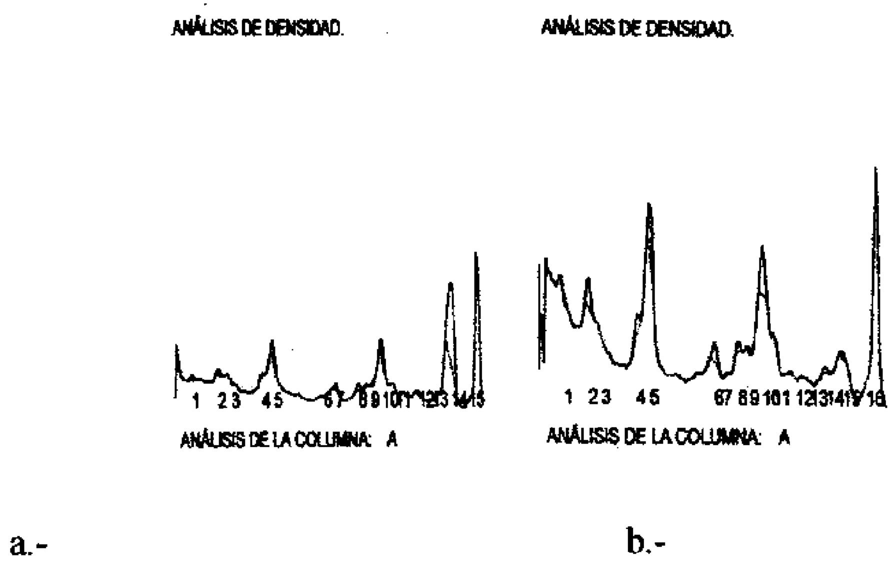


a. -

b.-

Figura 3. Imagen de proteinogramas. A. - Muestras Chinchorro machos 15; b. - Muestras Chinchorro hembras 16.

La determinación de las fracciones reveladas en los sitios muestreados y el análisis digital de los mismos, nos permite establecer que existen diferencias entre los organismos de cada localidad, específicamente entre machos y hembras; similitud en él número de fracciones en las localidades de Punta Allen e Isla Mujeres y diferencias entre estas localidades y la de Banco Chinchorro, lo que nos permite establecer la posible diferenciación geográfica de los caracoles de estos sitios y cuyo resultado debería de ser tomado en cuenta al efectuar las estadísticas de captura de *Strombus gigas* en las costas del caribe mexicano, con las precauciones pertinentes que amerita el obtener más datos que resulten al emitir algún tipo de conclusión referente a esta posible separación y diferenciación geográfica, y que permita con ello tener un adecuado conocimiento del estado actual de la pesquería del caracol rosado y la aplicación de medidas de manejo acordes a su realidad.



a.-

b.-

Figura 4. Polígonos de densidad. a. - Muestras Chinchorro machos 15; b. - Muestras Chinchoro hembras 16

Tradicionalmente el análisis de los proteinogramas revelados electroforeticamente se efectuaba en forma visual y era bastante tedioso y tardado no estando exento de resultados erróneos en el conteo y medición de las movilidades relativas de las fracciones reveladas, para tratar de remediar estos problemas diversos programas de análisis digital fueron desarrollados, como el UN-SCAN_IT, el ONE_DSCAN, el ZERO_DSCAN, etc., (González 1996) sin embargo los costos de ellos son elevados, alrededor de 5,000 DLS, lo cual los hacen difíciles de obtener, además de presentar características que restringen el análisis de los proteinogramas y su poca compatibilidad con otros programas que en forma conjunta redundarían en elevar la resolución y manejo de los resultados.

El software utilizado en este trabajo fue elaborado para el reconocimiento y medición de fracciones proteicas obtenidas electroforeticamente, esta desarrollado en lenguaje C++ en el compilador Visual C++ de Microsoft versión 5.0 y es capaz de proporcionar una interfase de usuario en ambiente Windows con herramientas del sistema operativo que facilitan su manejo y enlace con otros programas del ambiente (Myler and Weeks 1993).

Las imágenes pueden ser abiertas en diversos formatos lo cual permite la manipulación de caracteres de brillo, contraste, ecualización y filtrado que mejoran y aumentan la visualización y resolución de las fracciones reveladas electroforeticamente. Estas características crean un polígono de frecuencias de la composición proteica de la imagen, el cual mediante el manejo y aplicación de los diversos controles contenidos en el programa nos permite discernir, discriminar o considerar la presencia de fracciones que el investigador en forma visual no puede distinguir.

Los resultados obtenidos en este trabajo nos permiten concluir:

Al nivel de presencia de fracciones proteicas existe diferencia entre las poblaciones de Isla Mujeres y Punta Allen con la de Banco Chinchorro.

El programa PROANA, resultó ser una herramienta valiosa para el análisis de las imágenes obtenidas de los proteinogramas obtenidos electroforeticamente, al permitirnos manipular las imágenes nos da la pauta para establecer la presencia de fracciones proteicas que en forma visual no es posible detectar, reduce los tiempos de análisis y nos permite obtener resultados confiables cuya aplicación sería en forma más realista y benéfica para quien así lo requiera.

El considerar en base a los resultados aquí obtenidos el manejar con cuidado el acopio de datos de captura del caracol rosado y de ello la emisión de estadísticas que repercutan en la implementación de medidas de manejo del organismo que tiendan a sustentar su explotación en forma controlada.

LITERATURA CITADA

Ayre, D.J.; J. Read, and J. Wishart. 1991. Genetic subdivision within the eastern australian population of the sea anemone. *Actinia tenebrosa. Marine Biology* **109**:379-390.

Corzo, J., J.M. Cunas, and E. Meléndez 1984. Fish species identification by electrophoresis of muscle protein in SDS containing polyacrylamide slabs gels. *Electrophoresis* **8**:(4).

González, R. R y R. Woods. 1996. *Tratamiento Digital de Imágenes.* Addison Wesley. USA. 189 pp.

Gordillo, M. J. L., J. M. Sosa and P. Wiederhold. 1992. Análisis de Imágenes y Reconocimiento de Patrones. Depto. Ing. Eléctrica. CINVESTAV. 1-23.

Kinsey, S. T., T. Orsoy, T. M. Bert and B. Mahmoud. 1994. Population structure of the spanish sardine, *Sardinella aurita*, natural morphological variation in a genetically homogeneus population. *Marine Biology* **118**:309-317.

Kukatla R. 1996. Electrophoretic separation of proteins one dimensional SDS – PAGE electrophoresis. Department of Biology. Atlanta University. 5 pp.

Laemmli, U.K. 1970. Cleavage of structural proteins during the assembly of the head of bacteriophage T4 nature. London **227**:680-685.

McAndrew, B.J. and C J. Majumdar. 1983. Tilapia stock identification using electrophoretic markers. *Aquaculture* **30**:249-261.

Miller, H.R. and A.R. Weeks 1993. *The pocket handbook of imaging processing algorithms in C ++*. Prentice Hall. New York. USA: 95 pp.

Rybiki, E. and M. Purves 1998. SDS Polyacrylamide gel electrophoresis (SDS PAGE). Dept. Microbiology University of Cape Town. USA. 1(5).

Weber, K. and M. Osborn 1969. The reliability of molecular weight determinations by dodecyl sulfate - polyacrylamide gel elctrophoresis. *The Journal of Biological Chemistry* **244**:16.

Overview of the Spiny Lobster, *Panulirus argus*, Commercial Fishery in Puerto Rico During 1992-1998

DANIEL MATOS-CARABALLO
Fisheries Research Laboratory/PRDNER
P.O. Box 3665
Mayagüez, Puerto Rico 00681

ABSTRACT

Puerto Rico's fishery has shown symptoms of overfishing for the last fifteen years. The spiny lobster, *Panulirus argus*, is the most important shellfish in this fishery. Since 1985, *P. argus* has been protected by fishing regulations. The minimum legal size for this species is 89mm (3.50 inches) carapace length (CL). This measure is related with the minimum size of sexual maturation. Landings data indicated that 1.6 million pounds were caught in Puerto Rico during 1992 - 1998. Traps caught 51% of the *P. argus* landings and divers caught 41% during the same period. To evaluate the resource's status it is necessary among other criteria to obtain size frequency data. From 1992 - 1998, 5,354 *P. argus* caught by commercial fishermen in Puerto Rico were measured by personnel of the Fisheries Research Laboratory of the Puerto Rico Department of Natural and Environmental Resources.

The landings, catch per unit of effort (CPUE), and biostatistical data during 1992 - 1998 have been analyzed to evaluate the status of P. argus.

KEY WORDS: Fishery Management, *Panulirus argus*, Puerto Rico's commercial fishery

INTRODUCTION

Commercial fishery in Puerto Rico (PR) is artisanal, multispecies and multigear. During the last 15 years this fishery has shown overfishing symptoms. Matos-Caraballo (2000) mentioned changes in species catch composition, decreasing size of some species and a decrease in landings reported.

The most important shellfish species in Puerto Rico is the spiny lobster *Panulirus argus*. This species represented 7% of the total catch reported during 1992 - 1998. The average price per pound for this species for the mentioned period of time ranged from of $4.50 - $9.00. Jarvis (1932) mentioned that in the early 1930s fishermen caught *P. argus* but did not have a good market for them. This fact occurred probably because this species spoiled faster than other types of fish and shellfish. Valuable marketing for *P. argus* started approximately in 1947 (Feliciano 1958). Many old commercial fishermen interviewed by the author mentioned that in early 1950s, this species was used mostly as bait for traps. Overfishing indications in *P. argus* had been observed since the early 1980s. This fact resulted in the Lobster Fishery Management Plan (LFMP) implemented in 1985 for the waters in the jurisdiction of the Commonwealth of PR (shore to 9 nautical miles) and USA Exclusive Economic Zone (from 9-200 nautical miles). The LFMP includes a minimum legal size (MLS) of capture of 89 mm (3.50 in.) of carapace length (CL). The minimum size of sexual maturity in this species is 76 mm (3.0 in.) CL. The rationalization of the MLS of 89 mm, is to give the chance of any lobster to reproduce at least once before being harvested. Another important regulation is the prohibition to kill, possess or disturb females with attached eggs.

Matos-Caraballo (1999) reported that during 1989 - 1991, more than 50% of the *P. argus* were caught before reaching the MLS. He mentioned that LFMP was not enforced, indicating the importance to educate the enforcement officers, commercial fishermen and general public in the need to protect this resource. Since 1995, it was observed that the PR Department of Natural and Environmental Resources (DNER) enforcement officers started to work to force the LFMP. This paper presents an overview of the fishery of the *P. argus* using landings, catch per unit effort (CPUE) and biostastistical data from 1992 - 1998. Also, it examines the impact of recent enforcement efforts of the DNER on the *P. argus* resource.

PROCEDURES

Commercial Landings Data

Commercial fishery landings data were collected from Puerto Rico's fishermen, fish buyers and fishing associations, whom voluntarily cooperate with the Fisheries Statistics Program (FSP). Five port agents visited the 42 coastal municipalities including the islands of Vieques and Culebra, and the 88 identified fishing centers. Port agents worked in the data collection from January 1992 - December 1998. Efforts were made to collect the following data: fishing date, name of fish buyer, fisher and/or helper (to avoid data duplication), municipality; fishing center (municipality landing area), number of trips, gear type, fishing effort (hours spent fishing), weight in pounds by species, market value to the fisherman (price in U.S. dollars/pound), maximum and minimum fishing depth and fishing area.

Landings data were entered MS-DOS computers, using Microsoft FoxPro for Windows, checked against the original landing trip tickets to validate the information. The data was analyzed using Microsoft Excel. All data presented in this paper are reported data (no correction factor to estimate non-report data was used). Traps category includes fish pot and lobster pot. Diver category includes skin and SCUBA divers.

Commercial Biostatistical Data

Biostatistical data of spiny lobster were collected by port agents. Spiny lobsters were measured in CL, in millimeters (mm), weighed in grams and identify the sex. Data were recorded on data sheets form. The form was designed to facilitate entry and processing of effort data. Biostatistical data were entered in Trip Interview Program (TIP) developed by NMFS Southeast Fishery Science Center. Later, the data stored in TIP format was converted to FoxPro and analyzed using Microsoft Excel.

RESULTS

Commercial Landings Data

Figure 1 shows the total landings reported in PR of *P. argus* during 1992 - 1998. The observed increase of landings for the years 1995 - 1998 occurred mostly because more fishermen participated in the FSP, due to an incentive granted by the DNER. This fact indicates that at least the *P. argus* fishery resource seems to be stable during this period of time. The price per pound averaged $4.50 in 1992, and increased to $5.50 in 1998. On the north coast, port agent Héctor Y. López-Pelet, reported that in 1998, some restaurants were paid $9.00/pound. This is the highest price reported in Puerto Rico for *P. argus* and occurred due to the scarcity and high demand for this species in this coast.

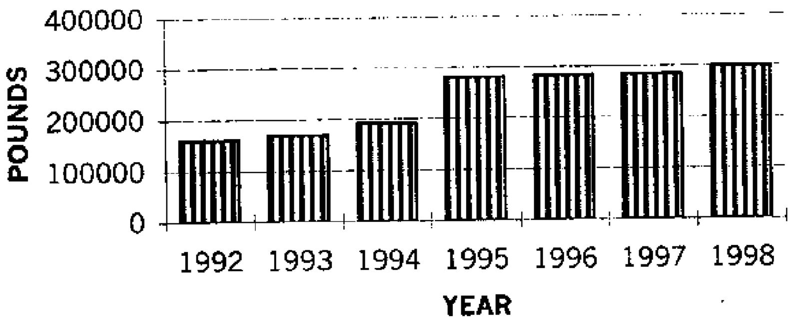


Figure 1. Landings reported of *Panulirus argus* in Puerto Rico during 1992 - 1998

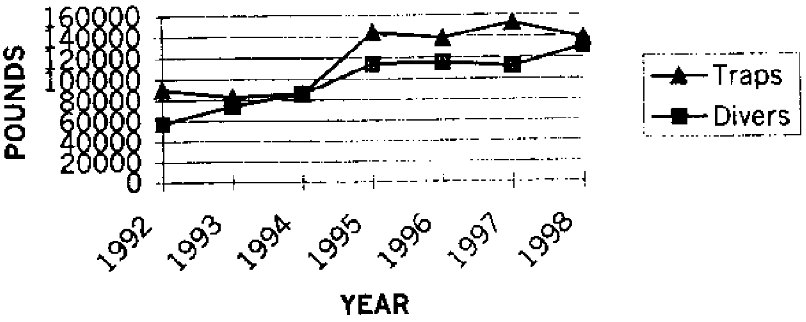


Figure 2. Landings reported by gear of *Panulirus argus* in Puerto Rico during 1992 - 1998

Traps reported 51% of the *P. argus* total landings reported during 1992 - 1998. Divers reported 41% of the *P. argus* for the same period of time. Figure 2 shows the landings reported by gear of *P. argus* in Puerto Rico during 1992 - 1998. During the 1970s and early 1980s in Puerto Rico's fishery the traps were the most productive gear (Matos-Caraballo, A). In the mentioned figure it is observed how the divers increased their landings and in 1998, being very close to the traps, which showed a decrease in landings reported.

A total of 36,139 trip tickets was reported with reliable pounds/trip information during 1992 - 1998. This information was used to obtain the CPUE (pounds reported/trip). Figure 3 shows the average of pounds per trip by gear for *P. argus* in PR during 1992 - 1998. This figure indicates that divers and traps were very similar in their CPUE from 1992 - 1997 (18-21 pounds/trip). In 1998, divers had an average of 33 pounds/ trip and traps had 28 pounds/trip.

Commercial Biostatistical Data

A total of 5,354 *P. argus* was measured by FSP port agents during 1992-98. From the mentioned total, 2,798 were males and 2,392 were females. Figure 4 shows the mean carapace length for *P. argus* in PR during 1992-98. *P. argus* males were larger than females. This figure shows during 1993-95, a trend to catch bigger *P. argus*. For years 1996 - 1997 the *P. argus* size shows a decreasing trend and finally increased for 1998.

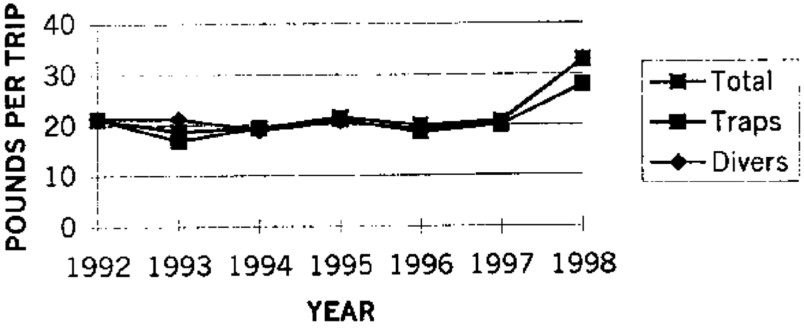


Figure 3. Average weight (in pounds) reported per trip by gear for *Panulirus argus* in Puerto Rico during 1992 - 1998

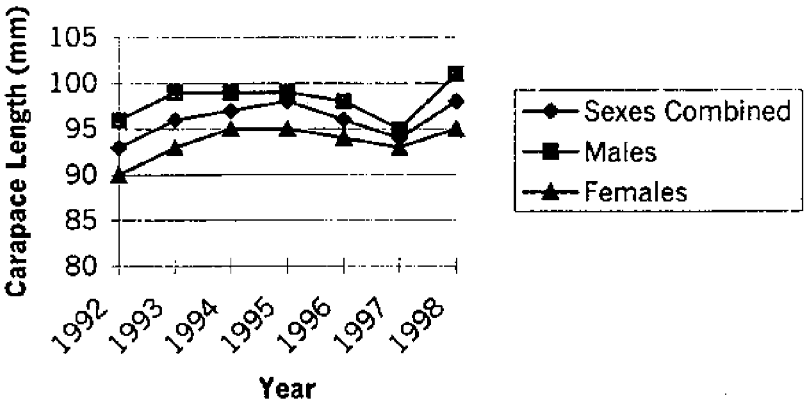


Figure 4. Mean carapace length of *Panulirus Aagus* in Puerto Rico during 1992 - 1998

Figure 5 shows the percentage of *P. argus* caught before reaching the legal size (89 mm) in PR during 1992 - 1998. The percentage of *P. argus* caught before MLS shows a decreasing trend for males, females and total (both male and female individuals combined). The percentage of *P. argus* males caught below MLS is lower than females.

During 1992 - 1998, port agents of the FSP measured 540 males and 294 females caught by traps. For the same period, they measured a total of 2,059 males and 1,955 females caught by divers. The mean CL of males and females of *P. argus* caught by traps and divers did not show any trend (Figures 6 and 7). The traps caught larger males for years 1992, 94 and 97 (Figure 6). The traps caught larger females for years 1992, 93, 94 and 97 (Figure 7).

The percentage of *P. argus* males and females caught by traps and divers before reaching the MLS since 1992 - 1998, showed a decreasing trend (Figure 8). Higher percentage of females were caught before reaching MLS than males. Figures 5 and 8 indicate that the work of DNER's enforcement officers organized in 1995, probably had impacted positively the *P. argus* fishery.

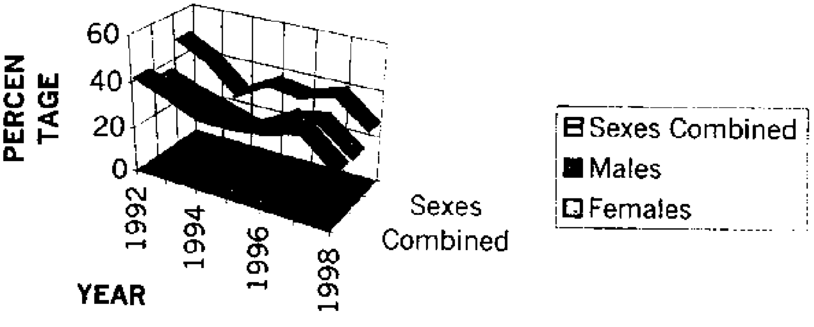


Figure 5. Percentage of *Panulirus argus* caught before reaching minimum legal size (89 mm CL) in Puerto Rico during 1992 - 1998

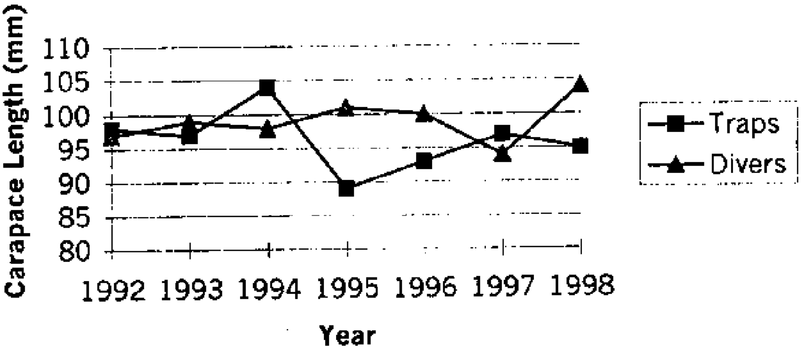


Figure 6. Mean of carapace length for *Panulirus argus* males caught by gear in Puerto Rico during 1992 - 1998.

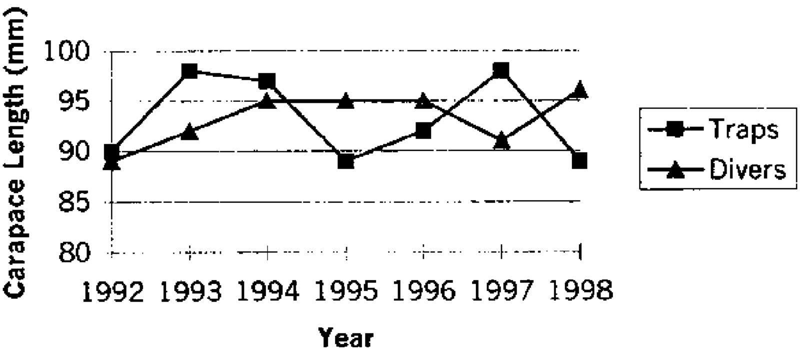


Figure 7. Mean of carapace length for *Panulirus argus* females caught by gear in Puerto Rico during 1992 - 1998.

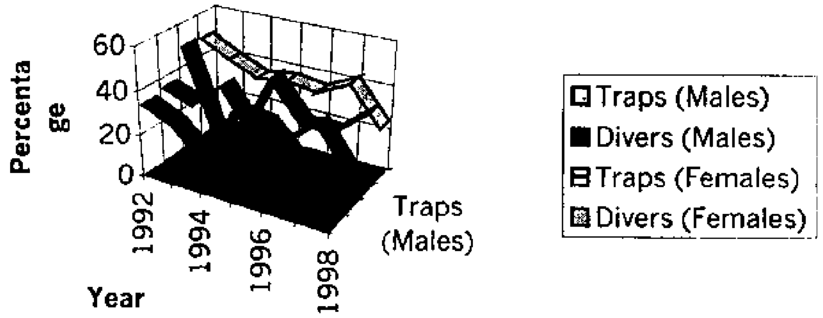


Figure 8. Percentage of *Panulirus argus* caught before reaching the legal size (8 mm CL) by gear and by sex inPuerto Rico during 1992 - 1998.

DISCUSSION

Commercial Landings Data

Mattox (1952) estimated that for 1951 a total of 466,000 pounds of *P. argus* was fished by 466 fishers. Matos-Caraballo (1999) mentioned that for 1991, 211,941 pounds of lobster were fished by 576 fishers. The landings data reported in Puerto Rico from 1971 - 1998 indicate that the fishery resource is overfished. For years 1979 - 1982, the peak years in landing reports for Puerto Rico, the average pounds reported for *P. argus* by year was 456,000 (Collazo and Calderón 1988). For 1995 - 1998, this average was 276,000 pounds per year by approximately 650 fishers. However, the data for 1992 - 1998, show that the decreasing tendency observed in the late 1980s and early 1990's is discontinued. The mean cost per pound of *P. argus* increased from $4.50 - $5.50 (in the north coast some restaurants pay $9.00/pound), indicating that this resource has a great demand and also continues to be overfished.

Traps fished more than 70% of the total landings reported in Puerto Rico species during the 1970s (Collazo and Calderón 1988). Matos-Caraballo (2000), mentioned that due to the overfishing situation the fishermen that used traps showed a tendency to shift to other fishing gears, such as lines, nets and diving. The data of *P. argus* also suggest that tendency.

The CPUE observed in this research was very similar for traps and divers. Although most trap fishermen are lifting their gear once per week, this gear fished 51% of the resource during 1992 - 1998. On the other hand, the divers fished 4-5 days per week. This fact suggests that the *P. argus* trap fishery is efficient.

Commercial Biostatistical Data

P. argus mean CL decreased from 117 mm in 1951 (Mattox 1952) to 107 mm in 1958 (Feliciano 1958). The decrease continued during 1989 - 1991, when it was observed that CL mean for *P. argus* was approximately 91mm (Matos-Caraballo, 1999). During the present study the mean of CL for the *all P. argus* was 96 mm, 98 mm for males and 93 mm for females. Larger individuals were caught during this period of time than for 1989 - 1991. The tendency to caught larger *P. argus* during 1992 - 1998 rather than 1989 - 1991,was also observed by gear. During 1989 - 1991, *P. argus* males caught by traps had a mean CL of 92 mm and females 87 mm. On the other hand, for 1992 - 1998, for the same gear males mean CL was 96 mm and females was 93 mm. For 1989 - 1991 was reported that *P. argus* males caught by divers had a mean CL of 94 mm and females was 89 mm. On the other hand, for 1992 - 1998, divers caught males with a mean CL of 99 mm and females were 93 mm. Mattox (1952), Rosario [1987] and this research found that male *P. argus* were caught at a larger size than females.

During 1989 - 1991, approximately 59% and 50% of *P. argus* fished by traps and divers respectively, were caught before reaching the MLS (Matos-Caraballo 1999). These results suggest that LFMP was ineffective, because no enforcement effort was observed at that time. The implementation of the enforcement effort of the LFMP starting in 1995 was observed immediately, because in 1996, only 31% of the total *P. argus* was caught before reach MLS, in 1997 it was 35% and in 1998 it was 24%.

The results of this research strongly suggest that the enforcement efforts of the DNER's enforcement officers helped significantly the LFMP to attain its objectives. Evidently this effort helps to educate more fishermen in the conservation of this resource and they supported the LFMP. If this tendency continues the *P. argus* fishery resource of Puerto Rico will be available for future generations.

ACKNOWLEDGMENTS

I wish to express my gratitude to all the people who contribute to the completion of this paper. Port agents Walter Irizarry, Jesús León, Héctor Y. López and Luis A. Rivera, who collected the data. Statistic clerk Lucía T. Vargas handled, entered and corrected the data. To Miguel Figuerola for reviewing this manuscript. To NMFS Cooperative Fisheries Statistics Program and PRDNER for providing the funds that financed this research. In particular, I wish to acknowledge the cooperation of the commercial fishermen for assisting

the Fisheries Statistics Project: without their help this report would not have been possible.

LITERATURE CITED

Collazo, J. and J.A. Calderón. 1988. Status of the Fisheries in Puerto Rico 1979-1982. Technical Report. CODREMAR. 3(2):1-30.

Feliciano, C. 1958. The Lobster Fishery in Puerto Rico. *Proc. Gulf and Carib. Fish. Inst.* **10**:147-156

Jarvis, N.D. 1932. The Fisheries Of Puerto Rico. U.S. Department of Commerce, Bureau of Fisheries, Investigational Report No. 13:1-41.

Matos-Caraballo, D. 2000 Overview of Puerto Rico Small-Scale Fisheries Statistics 1994 - 1997. *Proc. Gulf and Carib. Fish. Inst.* **51**:215-231.

Matos-Caraballo, D. 1999. Comparison of Size of Capture by Gear and by Sex of Spiny Lobster (*Panulirus argus*) during 1989 - 1991. *Proc. Gulf and Carib. Fish. Inst.* **45**:809-820.

Mattox, N.T. 1952. A Preliminary Report on the Biology and Economics of the spiny lobster in Puerto Rico. *Proc. Gulf and Carib. Fish. Inst.* **4**:69-70.

Rosario A. [1987] Population Dynamics of *Panulirus argus* in the east and south coast of Puerto Rico. Fisheries Research Laboratory, Mayaguez, Puerto Rico Unpubl. MS. 61 pp.

Alarming Overexploitation of the Caribbean Spiny Lobster Stock in Martinique: Ecological Point of View and Recommendations

EMMANUEL RICLET
Department of Biological Sciences
Old Dominion University,
Norfolk, VA 23529, USA

ABSTRACT

The Caribbean spiny lobster (*Panulirus argus*) is the most valuable marine species along the entire intertropical area of the western Atlantic. *Panulirus argus* represents 54 % of the world catch of Palinuridae, about 39,700 tons. From 1996 to 1998, I managed an ecological monitoring program and conducted grow-out experiments on spiny lobsters in Martinique, to evaluate the state of the stock and the potential enhancement in the West French Indies. Postlarval supply, as measured on surface collectors, is moderate and continuous throughout the year, and throughout the shallow water habitats. However, juveniles are rare in the wild and can not be concentrated or protected by artificial habitats. Three non-exclusive marine reserves exist in Guadeloupe, but there are none in Martinique, nor is there any biological monitoring of neritic populations, or of shallow water nurseries. In fact, the decline of *P. argus* stock seems to be inexorable. In absence of any enforcement of fishery regulations (established since 1970), the artisanal fishery is largely unregulated and now impacts juveniles as much as adult spiny lobsters. All significant local markets are provided by imports of live (St-Vincent, Grenadines) and frozen (Cuba) *P. argus*. Given the dire status of the wild population near Martinique, the potential for future juvenile culture is theoretically good but still limited by the availability of an adequate and inexpensive lobster food. There is much that can be accomplished in further applied research studies, but I strongly recommend that local elected representatives work for respect of pragmatic regulations and the education of younger generations on the environmental problems facing Martinique and Guadeloupe.

KEY WORDS: Spiny lobster, fishery, Martinique

INTRODUCTION

Although their valuable spiny lobster resources are already overfished, necessity of a strong fishery management plan is still under-estimated in the French overseas departments and territories (DOM-TOM). The main reef areas concerned are Martinique and Guadeloupe in the Caribbean, Reunion in the

Indian Ocean, New Caledonia and French Polynesia in the Pacific (the exclusive economical area of the latter is as large as Europe). Thousands of tons of the European (*Palinurus elephas*) and Mauritanian (*Palunirus mauritanicus*) spiny lobsters were annually caught at the beginning of the century, before the forseeable collapse of those stocks (Ceccaldi and Latrouite 1994). Similarly, the current situation in the French West Indies is alarming. The increase of interest for *Panulirus argus* led to a dramatic situation described here for Martinique. Although Guadeloupe benefits from a larger reef area and a lower human population, it is subject to similar ecological, social and economical influences (Le Bail 1994).

SPINY LOBSTER FISHERY

Locally called *langouste royale*, *Panulirus. argus* is now considered a luxury product that brings $ 35 - 40 per kilogram at dockside. However, it was previously neglected in the French West Indies and until the 1950s was considered a harmful beast that interfered with fishes. As a sub-product of that fishing, spiny lobsters were thrown back or crushed to bait wire fish traps. Morice (1958) reported that *P. argus* was omnipresent and abundant, even in the shallowest seagrass meadows. Fishers caught them by hand, sometimes using fires to attract them closer to beaches (Morice 1958). Interest for fishes decreased soon in favor of spiny lobster. As *P. argus* mainly lives in shallow water habitats (i.e. accessible for both professional and recreational fishers), fishing with snare, harpoon or octopus (*chatrou*) were common, until those devices became inefficient.

Indeed, most fishers, because of a lack of law enforcement, do not follow local regulations. Juvenile and adult spiny lobsters populations are now reduced all around the island. Fishers gradually work further and further from the Atlantic coast, where the continental plate extends 15 km from shore. On the west coast of Martinique, where the continental shelf is very narrow, wire fish traps are now set at depths of 100-150 m but are inefficient. Our SCUBA dives to 70 meters confirmed that only a few *P. argus* are found in such deep habitats. However, along with the lack of controls of catches and fishing devices, there is no long-term monitoring of catches. Gobert (1990) stated that lobster size structure reversed since Clairouin's study (1980): in 1988, 60% of the sampling *P. argus* population was represented by animals under 22 cm carapace length (legal harvesting size).

As we regularly worked with fishers, we noted that many of them neglect regulations, yet all of them deplore the decrease in abundance of adult *P. argus*. Mesh dimension of wire fish traps is not respected, use of night lights and SCUBA diving are common, and animals under legal size and egg-berring females are commonly caught and sold on the beach or on the sidewalk. In fact, there are no organized markets. Places and times of marketing are numerous and variable.

ECOLOGICAL APPROACH

Although some benthic processes have been described for spiny lobsters (Booth and Phillips 1994, Herrnkind et al. 1994), the local conditions (Durand 1996, Lazure et al. 1996) and the lack of any previous biological monitoring (Coton and Nijean 1987, Gobert 1994) in Martinique justified our ecological investigations. The studies we managed in Martinique from 1996 to 1998 showed that ecological processes are not favorable to an improvement of *P. argus* stock situation. After trying several methods for postlarval collection (static plankton nets equipped with a thermal or electric engine for water pumping, algal and/or cryptic artificial habitats on horizontal and vertical "networks") (Riclet 1998), we deployed surface collectors derived from Hunt's (Phillips and Booth 1994). Although one thousand postlarvae were collected, postlarval supply – which occurs during new moon nights (Acosta and Butler 1999) – was moderate. Collection was continuous throughout the year, occuring throughout the shallow water habitats, and presumably originates from upstream populations (Riclet 1998, Riclet submitted).

Although Farrugio (1976) noticed juvenile spiny lobsters were sometimes abundant in mangroves of Martinique, detection of any *P. argus* is now rare. Thus, we managed a visual census on five stations around Martinique: only two juveniles were detected on the south-eastern station. Then we set benthic artificial habitats on the same stations to determine their influence on juvenile *P. argus* survival. However, those habitats which proved efficient in the Florida Keys (Herrnkind et al. 1997, Butler and Herrnkind 1997), did not collect or concentrate juvenile *P. argus* during 14 months deployed in Martinique waters (Riclet submitted).

Other factors have to be considered. First, the development of the favorite settlement habitat of *P. argus* (Marx and Herrnkind 1986) - *Laurencia* sp. (Rhodophycae) – is negligible in Martinique. Thus, in spite of their potential suitability as nursery grounds, mangrove and seagrass beds (*Thalassia testudinum* and *Syringodium filiforme*) represent alternative nurseries for *P.argus* in the French West Indies (Acosta and Butler 1997). Their extent is limited around Martinique and will be estimated with the use of remote sensing data (IFREMER unpublish. data). Moreover, many neritic habitats are submitted to increasing pollution, due to both industrial and urban activities (Bouchon and Laborel 1986, Pommepuy et al. 1996, Riclet 1998).

RECOMMENDATIONS

The gravity of the ecological status of *P. argus* in Martinique suggests new management considerations. In spite of its artisanal character, local fishing practices are lethal for the exploitable stock by inexorably reducing its reproductive potential.

All significant markets are provided by importation of frozen (Cuba) or live spiny lobsters (Grenadines, St-Vincent), but regulations are not respected in those islands either. The official import (130 tons) is certainly underestimated because official controls are imprecise, and "ghost" imports using fishing boats are commonplace. So, which Caribbean countries will be able to provide live *P. argus* in the future? Even if Martinique and Guadeloupe's spiny lobster stocks are replenished by a postlarval supply originating from upstream populations (probably Grenadines, St-Vincent and Venezuela), such dependence is increased by natural variations of recruitment, alteration of settlement habitats, and recruitment overfishing. Ironically, the French West Indies, after having heavily exploited their own stocks, seem to contribute in upstream islands to the reduction of the larval production, from whom they are dependant. Although 23 % of the *P. argus* catches come from Brazil - about 9,300 tons per year (FAO 1998) – the recent distinction of a Brazilian subspecies from the Caribbean one – respectively called *P. argus westonii* and *P. argus argus* by Sarver et al. (1998) - destabilizes the hypothesis of the main participation of Brazilian native lobsters to the larval flux, which go from the Lesser Antilles to the Greater Antilles and the Gulf of Mexico (Ingle et al. 1963, Menzies 1981, Lyons 1981). In comparison with other cases (Australia, New Zealand, Japan), the fishery for *P. argus* concerns many small countries. However, the ecological connectivity of that system is obvious (Lessios et al. 1984, Cubit Roberts 1997), even between distant areas like the Lesser Antilles and the Florida Keys (Silberman et al. 1994, Shulman and Bermingham 1995).

Therefore, a long-term investment should be locally initiated to avoid a drastic increase of biological resources exploitation and the degradation of neritic environments. As the ecological monitoring of some main populations allows efficient management (Cuba, Florida Keys, Mexico) (Phillips et al. 1994), a pragmatic ecological survey and a minimal management, controlled by law enforcement, should be the first step for a potential replenishment of the overfished populations of *P. argus* in Martinique. Considering our dependence to upstream production of larvae and the natural variations of postlarval supply due to oceanographic processes, surviving recruits have to be protected. After an estimation of the ecological impact, only a fraction of those recruits should be collected for aquaculture projects. Such a regulation recently begun in New Zealand, where the government now allows the collection of 40,000 postlarvae for each ton of adult spiny lobsters fished (Booth et al. 1999). Without pragmatic management of the spiny lobster population and its environment, scientists and investors have also considered juvenile culture (Ryther et al. 1988, Phillips 1988, Phillips and Evans 1997). But such applied projects remain dependent on a suitable food for grow-out (Riclet 1998).

While there are three non-exclusive marine reserves in Guadeloupe, there are none in Martinique. The potential efficiency of such reserves for spiny lobster populations dynamics has already been described (Acosta 1999, Childress 1997, Roberts and Polunin 1991). Marine reserves are not specific and would benefit all exploited species in Martinique: spiny lobsters, fishes, molluscs, sea urchins. If the potential effect of a reserve on the postlarval supply for *P. argus* by local recruitment is not clear, it might still benefit downstream stocks (Lipcius et al. 1997). Creation of exclusive reserves in Martinique could benefit to Guadeloupe's population, for example.

While sublegal size animals have to be protected, most recent studies showed that the proportion of bigger animals has a strong influence on potential recruitment (McDiarmid and Butler 1999). In fact, fecundity sharply increases with age and would justify a maximal size limit in the next few years. Such information should be disseminated now to permit a gradual change in fishing behavior.

Fishing with beach seines should be prohibited because it is an inefficient practice and destroys shallow settlement habitats, essential for *P. argus* and other benthic populations. Mesh size for wire fish traps should be respected, while their effective use should be improved. In fact, many of them are not regularly inspected and are lost. Made with iron wire, those fishing devices remain lethal for several months for spiny lobsters and fishes because no wooden escape boards are used.

Panulirus argus are also impacted by another practice: the neglect of traditional neritic fishing resources for the exploitation of oceanic populations. As in French Polynesia, the difficulty in managing stocks and in protecting them from pollution led to the promotion and development of "open sea" fishing. But one forgets that change of localization necessitates improved fishing equipment and only represents an alternative to nearshore ecosystem depletion, not a solution. The productivity of the vulnerable productive nurseries and coastal fishing grounds justifies their protection. Complementing ecological studies, information and education of the youngest human generations would be beneficial for the future exploitation of marine resources in the French overseas departments and territories.

ACKNOWLEDGEMENTS

The author is grateful to the Ministère de l'Education Nationale, de la Recherche et de la Technologie (Paris, France) and the Association pour le Developpement de l'Aquaculture à la Martinique (ADAM) for the financial support of his doctorate. Sincere thanks to Dr. Mark J. Butler for the review of this paper.

REFERENCES

Acosta, C.A. and M.J. Butler IV 1997. Role of mangrove habitat as a nursery for juvenile spiny lobster, *Panulirus argus*, in Belize. *Mar. Freshwat. Res.* **48**(8):721-728.

Acosta, C.A. and M.J. Butler IV 1999. Adaptative strategies that reduce predation on Caribbean spiny lobster postlarvae during onshore transport. *Limnol. Oceanogr.* **44**(3):494-501.

Booth, J.D., P. Davies and C. Zame 1999. Commercial-scale collections of young rock lobster aquaculture. *The Lobster Newsletter* **12**(1):13-14.

Bouchon, C. and J. Laborel 1986. Les peuplements coralliens des côtes de la Martinique. *Ann. Inst. Océano., Monaco* **62**(2):199-237.

Butler, M.J. IV and W.F. Herrnkind 1997. A test of recruitment limitation and the potential for artificial enhancement of spiny lobster populations in Florida. *Can. J. Fish. Aquat. Sci.* **54**:452-463.

Ceccaldi, H.J. and D. Latrouite 1994. The French fisheries for the European spiny lobster *Palinurus elephas*. Pages 169 – 178 in: B.F. Phillips, J.S. Cobb and J. Kittaka (eds), *Spiny lobster management*, Blackwell Sci. Publ., Oxford.

Childress, M.J. 1997. Marine reserves and their effects on lobster populations : report from a workshop. *Mar. Freshwat. Res.* **48**(8):1111-1114.

Clairouin, N. 1980. Contribution à l'étude du stock de la langouste *Panulirus argus* en Martinique. *Science et Pêche, Bull. Inst. Pêches Marit.* **300**:7-18.

Coton, P. and C. Nijean 1987. Les post-larves de langoustes *Panulirus argus*. *Proc. Gulf Carib. Fish. Inst.* **38**: 591-599.

Durand, F. 1996. *Hydrodynamique sédimentaire sur le plateau insulaire de la Martinique*. Thèse Doctorat, Univ. Bordeaux I. 200 pp.

Farrugio, H. 1976. Contribution à la connaissance de la sexualité des langoustes *Panulirus guttatus* et *Panulirus argus* dans les eaux martiniquaises. *Science et Pêche, Bull. Inst. Pêches Marit.* **254**:1-11.

Gobert, B. 1994. Status of spiny lobster stocks in Martinique (1990). *Proc. Gulf Carib. Fish. Inst.* **43**:669-682.

Herrnkind, W.F., P. Jernakoff, and M.J. Butler IV 1994. Puerulus and post-puerulus ecology. Pages 213-229 in: B.F. Phillips, J.S. Cobb and J. Kittaka (eds), *Spiny lobster management*, Blackwell Sci. Publ., Oxford.

Ingle, R.M., B. Eldred, H.W. Sims and E.A. Eldred 1963. On the possible Caribbean origin of Florida's spiny lobster populations. *Fla. Bd. Conserv. Tech. Ser.* **40**:1-12.

Le Bail, J. 1994. Quel avenir pour la pêche française dans les Caraïbes ? *J. Rech. Océanogr.* **19**(3-4):202-206.

Lessios, H.A., D.R. Robertson and J.D. Cubit 1984. Spread of *Diadema* mass mortality through the Caribbean. *Science* **226**:335-337.

Lipcius, R.N., W.T. Stockhausen, D.B. Eggleston, L.S. Marshall Jr. and B. Hickey 1997. Hydrodynamic decoupling of recruitment, habitat quality and adult abundance in the Caribbean spiny lobster : source-sink dynamics ? *Mar. Freshwat. Res.* **48**(8): 807-815.

Lyons, W.G. 1981. Possible sources of Florida's spiny lobster population. *Proc. Gulf Carib. Fish. Inst.* **33**:253-266.

McDiarmid, A.B. and M.J. Butler IV 1999. Sperm economy and limitation in spiny lobsters. *Behav. Ecol. Sociobiol.* **46**:14-24.

Morice, J. 1958. Langoustes et scyllares des Petites Antilles. *Rev. Trav. Inst. Pêches Marit.* **22**(1):105-114.

Phillips, B.F. 1988. The potential for rock lobster mariculture in Australia. Pages 294 – 300 in: L.H. Evans and D.O'Sullivan (eds.), *Proc. First Australian Shellfish Aquaculture Conference*, Curtin Univ. of Tech.

Phillips, B.F. and L.H. Evans 1997. Aquaculture and stock enhancement of lobsters : report from a workshop. *Mar. Freshwat. Res.* **48**(8):899-902.

Phillips, B.F., J.S. Cobb and J. Kittaka 1994. *Spiny lobster management*. Blackwell Sci. Publ., Oxford. 550 pp.

Riclet, E. 1998. ***Recrutement et essais d'elevage de la langouste Panulirus argus (Latreille, 1804) en Martinique***. These Doctorat, Universite Paris VI – Ecole Pratique des Hautes Etudes. 167 pp.

Riclet, submitted. Recruitment of the spiny lobster ***Panulirus argus*** (Latreille, 1804) in Martinique: an assessment for future regional management. *C.R. Acad. Sci., Life Sciences, Sér. III*, Paris.

Roberts, C.M. 1997. Connectivity and management of Caribbean coral reefs. *Science* **278**:1454-1457.

Roberts, C.M. and N.V.C. Polunin 1991. Are marine reserves effective in management of reef fisheries ? *Rev. Fish Biol. Fish.* **1**(1):65-91.

Ryther, J.H., W.A. Lellis, S.P. Bannerot and J.A. Chaiton [1988] Crab and spiny lobster mariculture - Part II: Spiny lobster mariculture. Final Report USAID Grant N° 538-0140.03(1), HBOI, Fort Pierce, Florida. 42 pp. Unpubl. MS.

Sarver, S.K., J.D. Silberman and Walsh P.J. 1998. Mitochondrial DNA sequence supporting the recognition of two subspecies or species of the Florida spiny lobster *Panulirus argus*. *J. Crust. Biol.* **18**(1):177-186.

Silberman, J.D., S.K. Sarver and P.J. Walsh 1994. Mitochondrial DNA variation and population structure in the spiny lobster *Panulirus argus*. *Mar. Biol.* **120**(4):601-608.

Shulman, M.J. and E. Bermingham 1995. Early life histories, ocean currents, and the population genetics of Caribbean reef fishes. *Evolution* **49**(5):897-910.

Assessment of the Functional Effects of a Harvest Refuge on Spiny Lobster and Queen Conch Populations at Glover's Reef, Belize

CHARLES A. ACOSTA
SFNRC, Everglades National Park
40001 State Rd 9336
Homestead, Florida 33034 USA

ABSTRACT

Marine harvest refuges have great potential for protecting, sustaining, and possibly, increasing fisheries resources. Since 1996, I have monitored populations of Caribbean spiny lobsters *Panulirus argus* and queen conch *Strombus gigas* in a new marine reserve at the Glover's Reef atoll, Belize. Habitat use accounted for the size-class distribution of lobsters and conch. Density initially fluctuated with fishing season in both the General Use zone (fishing by permit) and the Conservation zone (fishing prohibited), indicating that compliance in the no-fishing area was initially low. Trends show a recent increase in the density of adult conch in the no-fishing zone coinciding with increased enforcement, but the density of adult lobsters has not increased. The mean size of conch has also increased in the no-fishing zone, but there has been no significant change in the mean size of lobsters in the protected area. The spiny lobster population in the reserve has not responded as expected, most likely due to insufficient amounts of critical habitats under protection and fragmentation of habitats by zonal boundaries.

KEY WORDS: Spiny lobster, queen conch, marine reserves

INTRODUCTION

Marine reserves that function as harvest refuges are increasingly being considered as a necessary management tool for sustaining, and even increasing, stocks of overexploited species (Bohnsack 1994, Davis and Dodrill 1980, Dugan and Davis 1993, Polacheck 1990). The potential consequences inside the areas permanently closed to fishing include increases in the density of animals and increases in the average size of individuals. Increased densities will subsequently result in net dispersal of animals from the reserve to adjacent fishing grounds ("spillover" effect). An increase in reproductive output of eggs from more and larger adults may lead to greater larval recruitment to the regional population (Carr and Reed 1993). Therefore, marine reserves that are fully functional are economically prudent investments, particularly in the Caribbean where most major fisheries are currently being overexploited (Chakallall and Cochrane 1997).

The two most economically valuable and heavily exploited species in the Caribbean are spiny lobsters, *Panulirus argus*, and queen conch, *Strombus gigas*.

Fishery trends for both species show consistent declines over the past two decades, indicating that most coastal stocks are being overexploited (e.g., Berg and Olsen 1989, Ehrhardt 1994). I have been monitoring spiny lobster and queen conch populations in the Glover's Reef Marine Reserve in Belize to assess population responses to protection from an artisinal fishery. I have also conducted experiments on dispersal dynamics and habitat requirements to evaluate the functional effects of the reserve on these populations. Here I describe changes in lobster and conch populations using the trends in density and average sizes in fished and unfished areas to assess the functional effects of this isolated reserve.

STUDY SITE

Glover's Reef (16°50'N, 87°47'W) is an isolated coral atoll with an approximate area 12 x 28 km, located 45 km off the coast of Belize. The atoll is surrounded by deep water (400 m leeward and 1000 m windward) with the forereef sloping sharply to vertical walls that begin less than 2 km from the reef crest in most areas. The atoll is ringed by an emergent reef crest with only three major cuts and several small breaks. The lagoon of the atoll contains extensive shallow water habitats used by all life history stages of spiny lobsters and queen conch, including sand-algal flats, seagrass, and more than 700 coral patch reefs. These major habitats in the lagoon are shallow (2 - 3 m) and easily accessible to artisinal fishermen who are limited only by the logistics of fishing trips to the atoll (e.g., ice for storing the catch). Currently, no commercial fishermen are resident on the five islands at Glover's Reef.

The Glover's Reef Marine Reserve was designated in 1993. The Conservation Zone (with an internal Wilderness area) is permanently closed to fishing and encompasses 72.3 km^2 or about 20.5 % of the lagoon of the atoll (Gibson 1999). A seasonal closure area is also closed to fishing and mainly covers forereef habitat that is known to be a spawning site for groupers. The General Use Zone encompasses the rest of the atoll and is open to fishing by permit. Fishing by SCUBA, traps, or spearfishing is prohibited, although fishermen capture lobsters by gaffing the animals. Fishing is concentrated in the lagoon habitats that are easily accessible by snorkeling.

METHODS

Quarterly surveys were conducted in representative habitats in the Conservation and General Use Zones from August 1996 to the present. Observations were recorded on the number of fishing boats near sampling sites and on the approximate amount of time rangers were present. Spiny lobsters were surveyed on patch reefs in the lagoon and on the forereef. Eight patch reefs in the Conservation Zone and eight in the General Use Zone (total patch reef area = 2.4 ha) were located using Global Positioning Satellite (GPS) and re-surveyed on a quarterly basis. Divers also surveyed 100 x 10 m transects at 10 sites on the forereef at depths of 5 and 20 m (total forereef area = 1.0 ha); due to logistical difficulties, only 5 forereef surveys were conducted. All lobsters were measured (carapace length, CL), sexed using external characters, and females were inspected for the presence of egg masses under the abdomen.

Queen conch were surveyed on the sand-algal flats of the back reef and around the patch reefs that had distinct sand-algal margins. Quarterly surveys were conducted at three Conservation and three General Use sites on sets of four 50 x 4 m transects (total sand-algal habitat = 0.48 ha). Conch were also surveyed around the margins of the 16 patch reefs in the lagoon (estimated reef margin = 1.2 ha). All conch were measured (total shell length SL and width/thickness of shell lip if present) and the presence of egg masses were noted.

To analyze population changes in space and time, a doubly multivariate repeated measures analysis of variance (RMANOVA) was conducted on density and mean size (dependent variables) and reserve zone and sample month (factors). Multivariate assumptions of normality, homoscedasticity, multicollinearity, and equality of variance-covariance matrices were tested, and all assumptions were met after the density data were log-transformed. Trend analysis was conducted on the abundance of adult lobsters and conch in and out of the no-fishing area over time. For analysis, adult status was considered to be the legal fishery-size animal (spiny lobster: 76 mm CL; queen conch: 180 mm SL).

RESULTS AND DISCUSSION

The known habitat use patterns of spiny lobsters accounted for the observed distribution of different size classes. Most size classes ranging from two months old juveniles to adults occupied the shallow patch reefs of the lagoon. Juveniles were most abundant on patch reefs surrounded by seagrass (Acosta 1999, Acosta and Butler 1997). In contrast, only large subadults and adults occupied the forereef and deep reefs where they were twice as abundant (mean ± SE: 20.4 ± 2.1 lobsters/ha) as adults in the lagoon (mean ± SE: 9.4 ± 1.4 lobsters/ha). The average sizes of adult lobsters on the forereef were also larger (mean ± SE: 101 ± 6.8 mm CL) than the adults in the lagoon (mean ± SE: 78.1 ± 2.9 mm CL). Data from tracking and mark-recapture studies indicate that large adults regularly move between lagoon reefs and the deep reefs (Acosta, unpubl.). Most reproductive activity (90%) occurred January to November, substantially longer than the closed fishing season designated to protect breeding adults. On the lagoonal patch reefs, a small proportion of mature females (8%) carried egg masses on these shallow (< 3 m) reefs during the reproductive season.

Lobster population fluctuations were most pronounced in the Glover's Reef lagoon. Lobster density fluctuated significantly between sample months but not across the fishing zones (Table 1A). From 1996 to early 1998, the density of lobsters fluctuated with the fishing season, increasing significantly only during the closed season (about March through June) in Belize (Figure 1A). In the past year, density of lobsters increased in the no-fishing area, but this increase is not greater than during the closed fishing season. Over the three years, the trends in densities of adult lobsters in the fishing and no-fishing zones indicate that no statistically significant changes have occurred (Figure 1A). The abundance of small juvenile lobsters (< 40 mm CL) throughout the study area did not fluctuate over the three year period which suggests that larval recruitment remained relatively constant. Therefore, fluctuations in the density of adult, as well as large subadult, lobsters mostly accounted for these observed patterns. The spiny lobster population is reduced by 60% during the fishing season, and the data show that a substantial number of subadults (50-76 mm CL) are also taken in the fishery.

Changes in the mean sizes of spiny lobsters were similar to density patterns. Again, mean sizes increased only during the closed fishing season in both the fishing and no-fishing zones (Table 1A, Figure 2A). Although mean lobster size was generally larger in the no-fishing zone, this did not vary in a statistically significant way over time. In contrast to these results, MacDiarmid and Breen (1992) showed that a population of the lobster *Jasus edwardsii* in a New Zealand marine reserve had a five-fold increase in five years. Additionally, the number of legal-sized adults increased by 90% in the reserve. At the Glover's Reef Marine Reserve in Belize, inconsistent enforcement and violation of the no-fishing zone by fishermen may have initially kept spiny lobsters *P. argus* density and mean sizes low. However, with increased enforcement, the population has still not responded as expected.

Queen conch were distributed throughout the lagoon of the atoll but not on the forereef which lacked suitable habitat. Both juveniles and adults were abundant on the shallow sand-algal flats of the backreef with abundant macroalgae *Laurencia* spp. Mostly large adults foraged around the sand-algal margins of patch reefs in the lagoon. The distribution of juveniles on the sand-algal flats coincides with the nearby distribution of dense turtlegrass *T. testudinum* that is a primary settlement habitat for conch larvae. In preliminary surveys, few adults were found in the dense seagrass beds and deeper rubble fields in the lagoon. Most reproductive activity (90%) occurred on the sand-algal flats during April to October.

Table 1. Results of the doubly multivariate repeated measures analysis of variance with sample month and reserve zone as factors and density and size distribution as dependent variables for A. spiny lobsters and B. queen conch at the Glover's Reef Marine Reserve, Belize. The test statistic used for the factors is Pillai's trace and the mean square is used for the dependent variables.

A. Spiny lobsters

Source	Value	Error (df)	Hypothesis (df)	F	P
Month	0.473	252	18	4.34	<0.001
Density	0.316	9	-	2.725	0.006
Size	1165	9	-	7.67	<0.001
Zone	0.034	13	2	0.228	0.80
Interaction	0.154	252	18	1.168	0.29
Error (density)	0.116	126	-	-	-
Error (size	151	126	-	-	-

B. Queen conch

Source	Value	Error (df)	Hypothesis (df)	F	P
Month	0.614	140	10	6.197	<0.001
Density	468	5	-	1.796	0.13
Size	16060	5	-	20.58	<0.001
Zone	0.202	13	2	0.012	0.05
Density	0.011	1	-	0.98	0.10
Size	0220	1	-	5.87	0.05
Interaction	0.155	140	10	1.174	0.31
Error (density)	260	70	-	-	-
Error (size	780	70	-	-	-

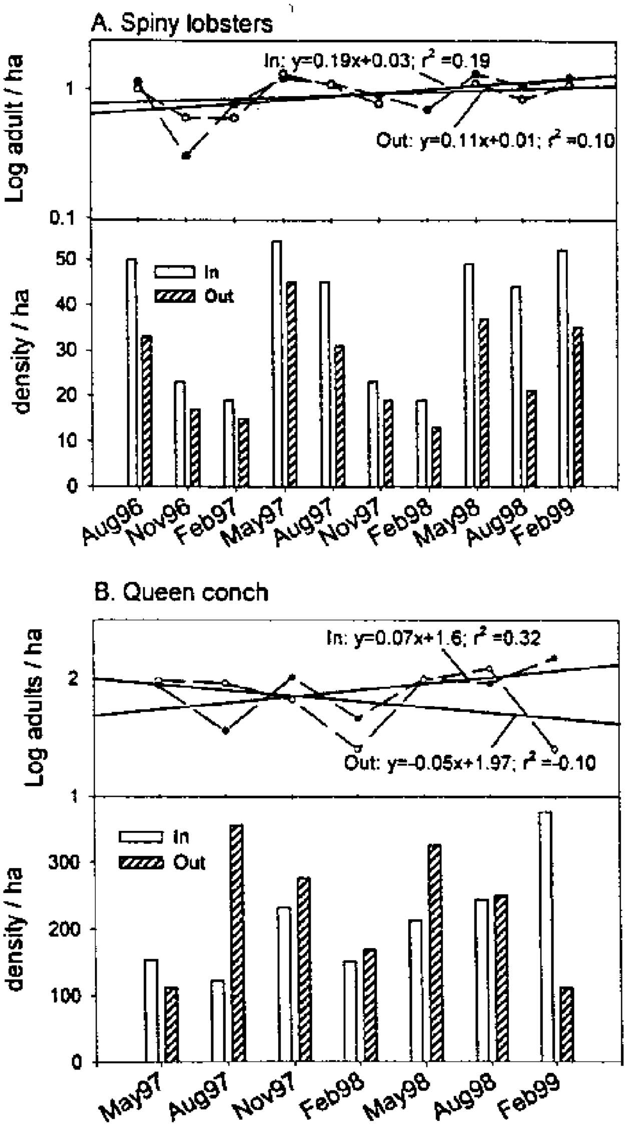


Figure 1. Abundance of A. spiny lobsters and B. queen conch in the no-fishing (In) and fishing (Out) zones of the Glover's Reef Marine Reserve, Belize. Bar graphs show total densities per hectare, and upper graph shows change in the densities of fishery-sized adults.

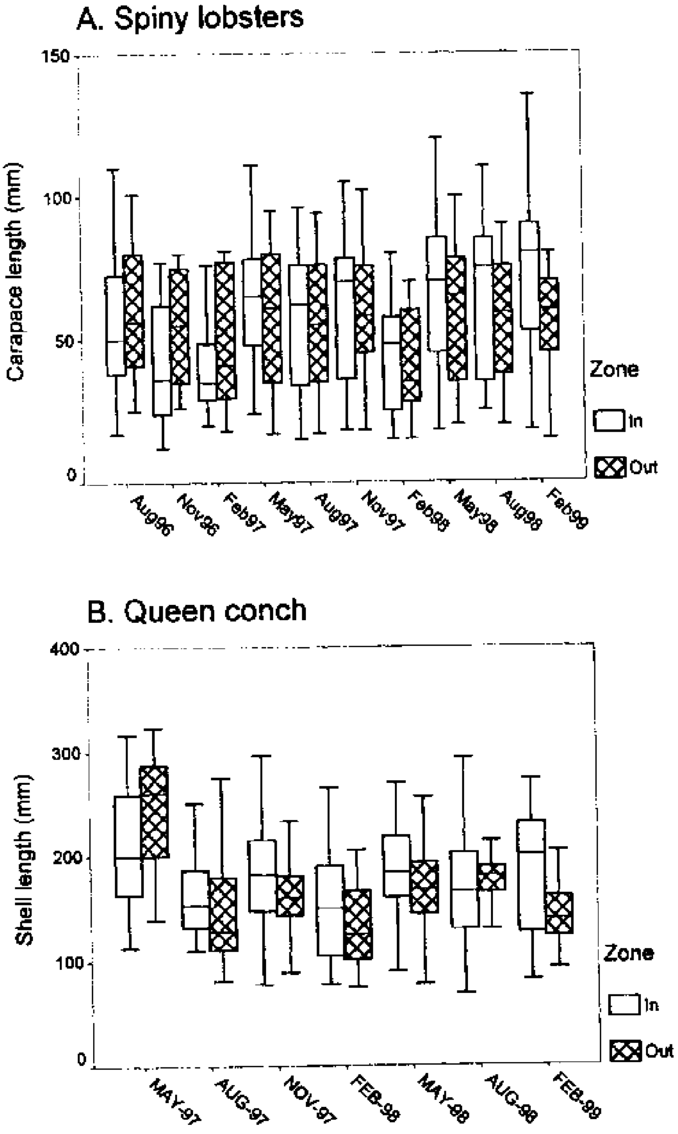


Figure 2. Mean sizes of A. Spiny lobsters and B. Queen conch in the no-fishing (In) and fishing (out) zones of the Glover's Reef Marine Reserve, Belize. Box plots show means, quartiles, and 95% confidence limits.

The highest concentration of juveniles consistently occurred on the southwest backreef that is open to fishing. This unusually high density of young conch (1,127 conch/ha) does not appear to be a result of benthic aggregative movement, but instead, may be linked to larval supply by water currents that flow northeast to southwest in the lagoon (Acosta, unpubl.). Due to juvenile densities at this single southwestern site, the relative density of conch in the fishing zone was generally higher than in the no-fishing zone, but density in the unfished area is slowly increasing (Figure 1B, Table 1B). Total conch density was generally higher in both zones during the closed fishing seasons. The trend for adult conch in the no-fishing zone accounted for most of the increase in this area, whereas adult abundance in the fishing zone has actually declined over time (Figure 1B).

Mean sizes of queen conch were larger on average in the no-fishing zone than in the fishing zone, but there were no significant changes over time in the no-fishing zone of the reserve (Figure 2B, Table 1B). In the Belize conch fishery, there is no provision for limiting fishing to sexually mature conch with shell lip formation. Gibson et al. (1983) reported that up to 70% of the legal catch in Belize may consist of immature conch, which is similar to the estimate (60%) at Glover's Reef.

Conch densities in both the no-fishing and fishing zones of the Glover's Reef Marine Reserve are considerably higher than recent estimates from other Caribbean areas (U.S. Virgin Islands: Friedlander et al. 1994, Bermuda and Florida Keys: Berg et al. 1992). Nevertheless, adult densities (89.3 adults/ha) in the no-fishing zone at Glover's Reef are considerably lower than reported for marine reserves at Las Roques, Venezuela (460/ha; Weil and Laughlin 1984) and the Bahamas (270/ha; Stoner and Ray 1996). Stoner and Ray (1996) reported that adult conch in the Bahamas reserve are at least five times as abundant as in the fished area.

The conceptual benefits of increased densities and mean sizes in marine reserves closed to fishing have been shown for populations of spiny lobsters (Davis and Dodrill 1980, MacDiarmid and Breen 1992) and queen conch (Stoner and Ray 1996, Weil and Laughlin 1984). At the Glover's Reef Marine Reserve, the queen conch population in the no-fishing area is slowly increasing, but the spiny lobster population has not responded as expected. This may be due to initially inconsistent enforcement or because three years is inadequate to detect significant changes. However, MacDiarmid and Breen (1992) observed dramatic changes in lobster density and sizes in the reserve in New Zealand in a five-year period. A number of factors may influence the magnitude of change in protected areas including the size, shape, and number of reserves necessary to protect a target species (e.g., Schonewald-Cox and Bayless 1986). For example, only those fish species whose home ranges are contained within the no-fishing area will be adequately protected in a marine reserve (Kramer and Chapman 1999). My data from studies in progress suggest that spiny lobsters and queen conch in the Glover's Reef Marine Reserve are differentially affected by the size of the no-fishing area and fragmentation of important habitats, like seagrass beds and reef cuts, by the zonal boundaries (Acosta, unpubl.). Current size regulations and closed seasons may be inadequate to protect regional lobster and conch populations. Marine reserves may be the best alternative tool for sustaining these fisheries, but protected areas must have certain characteristics, such as adequate area, to be effective.

LITERATURE CITED

Acosta, C.A. 1999. Benthic dispersal of Caribbean spiny lobsters among insular habitats: Implications for the conservation of exploited marine species. *Conserv. Biol.* **13**:603-612.

Acosta, C.A. and M.J. Butler IV. 1997. Role of mangrove habitat as a nursery for juvenile spiny lobsters, *Panulirus argus*, in Belize. *Mar. FreshwaterRes.* **48**:721-727.

Berg, C.J. Jr., R. Glazer, J. Carr, J. Krieger, and S. Acton. 1992. Status of the queen conch in Florida waters: a progress report. *Proc. Gulf Carib. Fish. Inst.* **41**:439-443.

Berg, C.J. Jr. and D.A. Olsen. 1989. Conservation and management of queen conch (*Strombus gigas*) fisheries in the Caribbean. Pages 421-442 in: J. F. Caddy (ed.) *Marine Invertebrate Fisheries: Their Assessment and Management.* Wiley, NY.

Bohnsack, J.A. 1994. How marine fishery reserves can improve reef fisheries. *Proc. Gulf Carib. Fish. Inst.* **43**:217-241.

Carr, M.H. and D.C. Reed. 1993. Conceptual issues relevant to marine harvest refuges: examples from temperate reef fishes. *Can. J. Fish. Aquat. Sci.* **50**:2019-2028.

Chakalall, B. and K.L. Cochrane. 1997. The queen conch fishery in the Caribbean – an approach to responsible fisheries management. *Proc. Gulf Carib. Fish. Inst.* **49**:531-554.

Davis, G.E. and J.W. Dodrill. 1980. Marine parks and sanctuaries for spiny lobster fisheries management. *Proc. Gulf Carib. Fish. Inst.* **32**:194-207.

Dugan, J.E. and G.E. Davis. 1993. Applications of marine refugia to coastal fisheries management. *Can. J. Fish. Aquat. Sci.* **50**:2029-2042.

Ehrhardt, N.M. 1994. The lobster fisheries off the Caribbean coast of Central America. Pages 133-143 in: B. F. Phillips, J. S. Cobb, and J. Kittaka (eds.) *Spiny Lobster Management.* Blackwell Scientific, U.K.

Friedlander, A., R.S. Appeldoorn, and J. Beets. 1994. Spatial and temporal variations in stock abundance of queen conch, *Strombus gigas*, in the U.S. Virgin Islands. Pages 51-60 in: R. S. Appeldoorn and B. Rodriguez (eds.) *Queen Conch Biology, Fisheries and Management*. Fundación Científica Los Roques, Caracas, Venezuela.

Gibson, J. 1999. Report on Glover's Reef Marine Reserve. Coastal Zone Management Institute, Belize City, Belize.

Gibson, J., S. Strasdine, and K. Gonzales. 1983. The status of the conch industry of Belize. *Proc. Gulf Carib. Fish. Inst.* **35**:99-107.

Kramer, D.L. and M.R. Chapman. 1999. Implications of fish home range size and relocation for marine reserve function. *Env. Biol. Fish.* **55**:65-79.

MacDiarmid, A.B. and P.A. Breen. 1992. Spiny lobster population change in a marine reserve. Pages 47-56 in: C. N. Battershill, D. R. Schiel, G. P. Jones, R. G. Creese, and A. B. MacDiarmid (eds.) *Proceedings of the Second International Temperate Reef Symposium*. NIWA, New Zealand.

Polacheck, T. 1990. Year-round closed areas as a management tool. *Nat. Res. Modeling* **4**:327-354.

Schonewald-Cox, C.M. and J.W. Bayless. 1986. The boundary model: a geographical analysis of design and conservation of nature reserves. *Biol. Conserv.* **38**:305-322.

Stoner, A.W. and M. Ray. 1996. Queen conch, *Strombus gigas*, in fished and unfished locations of the Bahamas: effects of a marine fishery reserve on adults, juveniles, and larval production. *Fish. Bull.* **94**:551-565.

Weil, E.M. and R.G. Laughlin. 1984. Biology, population dynamics, and reproduction of the queen conch *Strombus gigas* Linné in the Archipelago de Los Roques National Park. *J. Shellfish Res.* 4:45-62.

Preliminary Results from an Experimental Trap Fishery for the Spotted Spiny Lobster, *Panulirus guttatus*, in Bermuda

BRIAN E. LUCKHURST, TAMMY TROTT and SARAH MANUEL
Division of Fisheries
P.O. Box CR52
Crawl CRBX
Bermuda

ABSTRACT

A two year experimental trap fishery commenced in April 1998 to assess the feasibility of re-establishing a commercial fishery for the spotted spiny lobster *Panulirus guttatus* following the fish pot ban of 1990. Historically, this species was caught using fish pots and reported annual landings ranged from 13,000 to almost 42,000 lobsters for the period 1975 - 1989. As no measures of directed effort for *P. guttatus* are available in the database, it is not possible to generate catch per unit effort (CPUE) data to assess changes in the fishery during this period.

The current experimental fishery tested two basic trap types, a commercially manufactured plastic crustacean trap (Fathoms Plus) and a wire mesh trap (A1) produced by the Division of Fisheries. In the first year, four commercial fishers were provided with approximately equal numbers of the two trap types to fish commercially with the provision that detailed information on trap catches was provided. In the second year, the same number of traps were used but were divided equally between three pairs of fishers. In addition, biological sampling was conducted by the authors and fishers to collect data on size structure, sex ratio and reproductive condition.

A total of 10,592 spotted spiny lobsters were caught in the first year with 64.8% taken in the Fathoms Plus traps. The CPUE values (lobsters per haul) were 1.87 (± 0.06 SE) and 1.43 (± 0.06 SE) for the Fathoms Plus and wire traps respectively. In the second year, the total catch was 9,206 lobsters with CPUE values higher in wire traps than in the Fathoms Plus, although statistical analyses showed no significant difference between the means. In addition, the fish by-catch in the Fathoms Plus traps was much less than the level in the wire traps in both years.

The highest catch rates of spotted spiny lobster were recorded in the summer months (June – August). The biological sampling revealed a male-biased sex ratio in the trap samples of approximately 10:1 (M:F) in the first year and 12.8:1 in the second year. The mean size of males (66.7 mm carapace length (CL) (± 0.06 SE)) was larger than females (60.0 mm CL (± 0.39 SE)). The presence of ovigerous females in the catches indicated that the reproductive period was from May to September with a probable peak in June-July.

KEY WORDS: Spotted spiny lobster, *Panulirus guttatus* , trap fishery, Bermuda

INTRODUCTION

The spotted spiny lobster *Panulirus guttatus*, is known only from the western Atlantic Ocean from Bermuda, the Bahamas, the Caribbean Sea and Brazil (Williams 1984). There are few established fisheries for this species in the region although there is some small-scale harvesting of *P. guttatus* in Barbados, Dominica and Trinidad (Luckhurst and Marshalleck in press). There is also some recreational harvesting in southeast Florida (Moe 1991).

The biology of the spotted spiny lobster, known locally as the "guinea chick", was first studied by Sutcliffe (1953). Records indicate that this species has been fished commercially in Bermuda since the early 1960s but reported landings are only available from 1975 - 1989. Historically, a smaller scale version of the Antillian fish trap, used to harvest reef fishes and spiny lobster (*P. argus*), was employed to capture spotted spiny lobsters. Although a limited number of fishers specialised in harvesting *P. guttatus* over the years, the local market for this species became well-established. Field studies on the biology and the fishery for this species were conducted in Bermuda by Evans and Lockwood (1994) and Evans et al. (1996).

Following the Fish Pot Ban in Bermuda in 1990, which made the use of Antillian fish traps illegal, the Bermuda Division of Fisheries embarked on an experimental program to design and evaluate a lobster-specific trap to allow the continued harvest of spiny lobsters *P. argus* (Ward and Luckhurst 1996). The result was the establishment of a limited entry spiny lobster fishery utilising standard government-owned traps (Luckhurst 1999) which are leased for the duration of the spiny lobster season (September 1st to March 31st) each year. Following the successful implementation of this program, the Division of Fisheries addressed the issue of the more limited fishery for *P. guttatus*. After an evaluation of historical participation by fishers, the Division of Fisheries decided to work with experienced "guinea chick" fishers in order to evaluate the performance of two trap types in the experimental fishery. Standardised traps were provided to the selected fishers in about equal numbers and they were permitted to sell their catch in return for providing detailed statistics on their fishing activities.

MATERIALS AND METHODS

The experimental "guinea chick" fishery opened in April 1998 with four participants each with an allotment of 15 traps (total 60 traps). In the second year of the experimental fishery, three partnerships were formed consisting of two fishermen per partnership. The total number of traps remained the same, but was divided into three allotments of 20 traps.

The main objective of this experimental fishery was to evaluate the efficiency of two trap types, the Fathoms Plus trap and the A1 trap. The Fathoms Plus traps are designed specifically for crustaceans (crabs and lobsters) and are used extensively in commercial fisheries, mainly in the United States. They are lightweight, oval-shaped traps made of moulded, black plastic mesh. The A1 traps are rectangular (91.4 x 121.9 x 45.7 cm) and constructed of galvanised, green vinyl-coated 3.8 cm square mesh. Both trap types have one functional funnel on the side of the trap and all funnels are fitted with an identical 10.1 cm diameter white PVC ring at the inner end. The number of *P. guttatus* and the levels of fish by-catch in each trap type were compared.

In the second year of the experimental fishery, the funnel placement on half of the A1 traps was changed to a ramp-like entrance and these traps were designated as A2 traps. This modification was made because some fishers felt that the funnel placement on the A1 trap was the cause of the lower catches experienced in the first year.

Whenever possible samples of *P. guttatus* were measured (carapace length) and the sex and reproductive condition of each individual was determined.

RESULTS AND DISCUSSION

The Fathoms Plus trap outperformed the A1 trap (Table 1) in the first year (ANOVA, $F = 28.27$, $p < 0.001$, $\alpha = 0.05$). Although the mean CPUE for the Fathoms Plus trap in the second year was slightly lower than those of the A1 and A2 wire traps (Table 1), statistically there were no significant differences (ANOVA, $F = 2.38$, $p = 0.09$, $\alpha = 0.05$). This result suggests that the modification of the funnel entrance with a ramp-like structure did not significantly change the lobster catch rate.

Due to the variability in the levels of commitment to the experimental fishery by different participants, it was decided to further evaluate the performance of the trap types by selecting the most productive fisher, determined by total number of lobsters landed, in each of the two years. CPUE values for each trap type within each year were compared (Table 2). In 1998-1999 the mean CPUE of the Fathoms Plus trap was significantly higher than that of the A1 trap, but in 1999-2000 (Table 2) there were no significant differences between the three trap types which is consistent with the previous results (Table 1).

Table 1. Spotted spiny lobster experimental fishery data summary for 1998-2000.

Season	Trap Type	Mean CPUE	SE	No. of Trap Hauls	No. of lobsters landed	Fish By-Catch (CPUE)
1998-1999	A1	1.43	0.06	2,844	3,834	2,782 (1.04)
	Fathoms Plus	1.87	0.06	3,456	6,758	519 (0.15)
1999-2000	A1	3.34	0.17	759	2,554	1,420 (1.87)
	A2	3.30	0.20	733	2,382	1,348 (1.84)
	Fathoms Plus	2.87	0.12	1,450	4,270	252 (0.17)

Table 2. Summary of data from the most productive fisher in each of the two years of the experimental fishery. In 1998 - 1999 the fishing area was off the east end; in 1999 - 2000 the fishing area was off the south shore.

Season	Trap Type	Mean CPUE	SE	No. of Hauls	No. of lobsters landed	Fish By-Catch (CPUE)
1998-1999	A1	1.67	0.12	906	1,278	1,414 (1.56)
	Fathoms Plus	2.47	0.10	1,459	3,655	171 (0.12)
1999-2000	A1	3.16	0.19	537	1711	1,310 (2.44)
	A2	3.21	0.21	488	1603	1,259 (2.58)
	Fathoms Plus	2.93	0.15	940	2832	180 (0.19)

The data from the two most productive fishermen were also used to test for differences in CPUE values between the two years for a particular trap type. The mean CPUE for the Fathoms Plus traps in the second year was significantly greater to that in the first year ($F = 6.96$, $p = < 0.0088$, $\alpha = 0.05$) (Table 2). However, this difference was not as great as that between the two years for the A1 traps ($F = 50.00$, $p = < 0.001$, $\alpha = 0.05$) where the mean CPUE value for the A1 traps in the second year was almost twice that of the first year (Table 2). There are a number of factors which could have been responsible for these differences including trapping in different areas of the reef platform and different levels of fishers' skill and experience. The fishers believe that the "ageing" of the trap, as the fouling community develops, is an important factor affecting catching power. As the same traps were used for the two year period, both trap types developed substantial fouling communities but the effect was perhaps greater with the wire mesh traps. The fishers suggest that the black plastic mesh of the Fathoms Plus initially provided more attractive habitat for lobsters, but as the wire mesh traps became fouled by invertebrates and algae, they were frequented more by lobsters and catch rates became very similar to the Fathoms Plus.

Figure 1 shows the mean CPUE by trap type by month for the most productive fisher in each year. This figure indicates that the Fathoms Plus trap caught more lobsters in the first months of the experimental fishery but that CPUE values for both trap types converged by August 1998. Thereafter trap performance by type was similar until January 1999 when the CPUE dropped in the A1 traps. In the second year, in all trap types there was a summer peak in CPUE (Figure 1b) similar to that of the Fathoms Plus in the first year (Figure 1a). In general, monthly mean catch rates were very similar between trap types during the second year. In both years, catch rates were generally lower during the winter months.

This experimental fishery was essentially harvesting a virgin stock as no directed commercial harvest of this species had taken place since 1990. If fishing pressure is maintained at a modest level from the time that the fishery recommences, CPUE levels similar to those of the most productive fishers might be expected. If these figures are used as maximum estimates of catchability, the Division of Fisheries will be able to formulate management measures which are conservative in approach by scaling back from these maximum estimates to fishing effort levels which should be sustainable over time.

The by-catch rates were consistently lower for the Fathoms Plus traps (Tables 1 and 2). In contrast, the fish by-catch in the wire traps (A1 and A2) was markedly higher. Given these results, the Fathoms Plus trap is clearly the preferred trap type as minimising reef fish by-catch is an important fisheries management objective for the local lobster fishery.

The above results, suggest that the exclusive use of Fathoms Plus traps should be recommended when the commercial fishery for spotted spiny lobsters in Bermuda commences. The Fathoms Plus trap performs about equally as well as the wire traps (A1 and A2), which were the designs originally favoured by fishers prior to the experimental fishery, and has the lowest fish by-catch. An added benefit of selecting this trap type is the fact that the theft of Fathoms Plus traps was almost negligible during the two year experimental fishery in comparison with wire traps. Most of the participants have now come to accept that the Fathoms Plus trap is suitable for spotted spiny lobsters and the majority support its continued use.

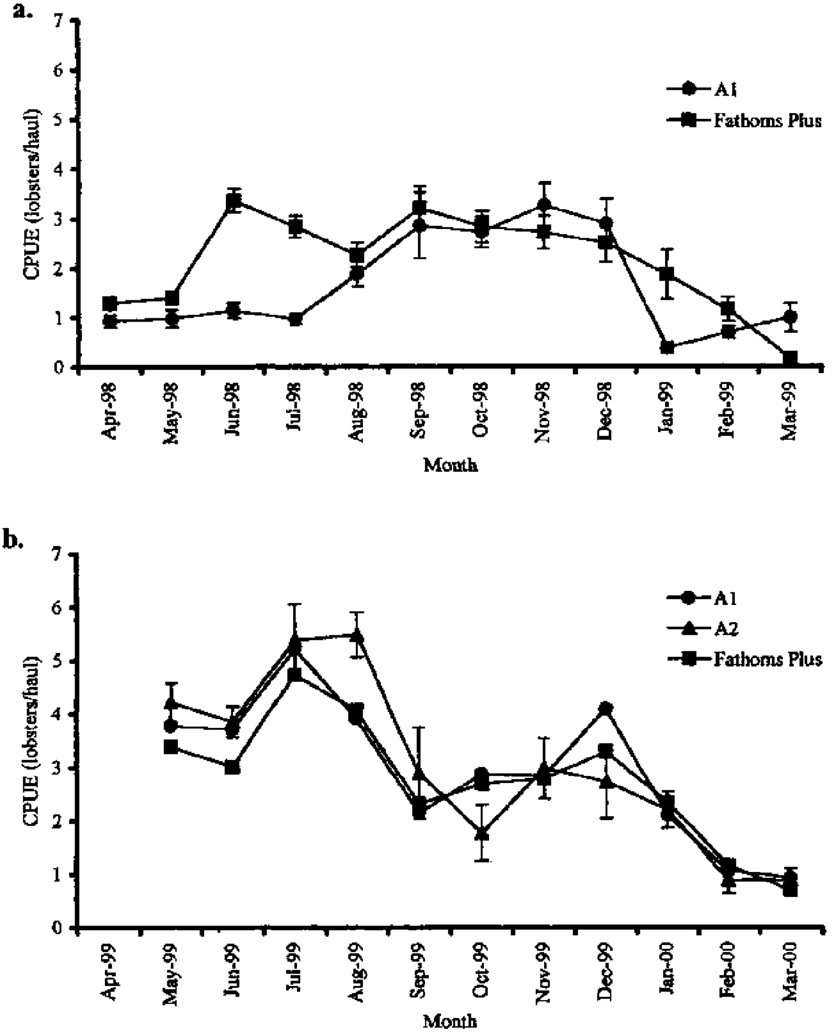


Figure 1. Mean CPUE (± SE) by month for: **a**. the most productive fisher in 1998 - 1999 and **b**. the most productive fisher in 1999 - 2000

Due to the variability in the levels of commitment to the experimental fishery by different participants, it was decided to further evaluate the performance of the trap types by selecting the most productive fisher, determined by total number of lobsters landed, in each of the two years. CPUE values for each trap type within each year were compared (Table 2). In 1998 - 1999 the mean CPUE of the Fathoms Plus trap was significantly higher than that of the A1 trap, but in 1999 - 2000 (Table 2) there were no significant differences between the three trap types which is consistent with the previous results (Table 1).

The biological sampling in both years indicated a strongly male-biased sex ratio in the trap catches. In the first year, the sample size was 1,238 (1,124 males, 114 females) for a sex ratio of approximately 10 Male: 1 Female. This finding is similar to that reported by Evans et al. (1995). The sex ratio in the second year of sampling (N = 6,112) was 12.8 Male:1 Female. If this is a true indication of the population sex ratio, then it is very important to protect females to maintain an adequate spawning stock. As a consequence, we will recommend that a commercial fishery commence with a prohibition on the possession of females in an attempt to ensure maximum reproductive output from the stock. Only eight female "guinea chick" lobsters carrying eggs were actually measured. These ranged in size from 52 to 70 mm carapace length (CL). However, the presence of ovigerous females in the catches, as recorded by the fishers' statistical forms, suggests that the reproductive period was from May to September with a probable peak in June-July.

A total of six night dives were made during the summer months to capture and sex spotted spiny lobsters to determine if the sex ratio differed from that observed in the trap catches. Unfortunately, only a small number of specimens were observed and captured (N = 16, 9 males, 7 females) and thus the sample size is inadequate to address this important issue.

The size range of lobsters sampled (N=1238) in the first year was 46-85 mm CL with the largest female measuring 75 mm CL. Mean sizes were 67.3 mm CL (± 0.16 SE) for males and 60.6 mm CL (± 0.44 SE) for females. In the second year, with a much larger sample size (N = 6,112), the size range was almost identical (47 - 85 mm CL) and the largest female was 71 mm CL. Mean sizes changed only marginally (Males = 66.5 mm CL (± 0.06 SE); Females = 58.4 mm CL (± 0.75 SE). Figure 2 shows the size frequency distribution of the lobsters sampled over the two years of the experimental fishery.

This experimental program has provided a scientific database which will be used in the formulation of a fisheries management program to allow the sustainable harvest of this small spiny lobster which will compliment the existing limited entry fishery for the spiny lobster *P. argus* in Bermuda.

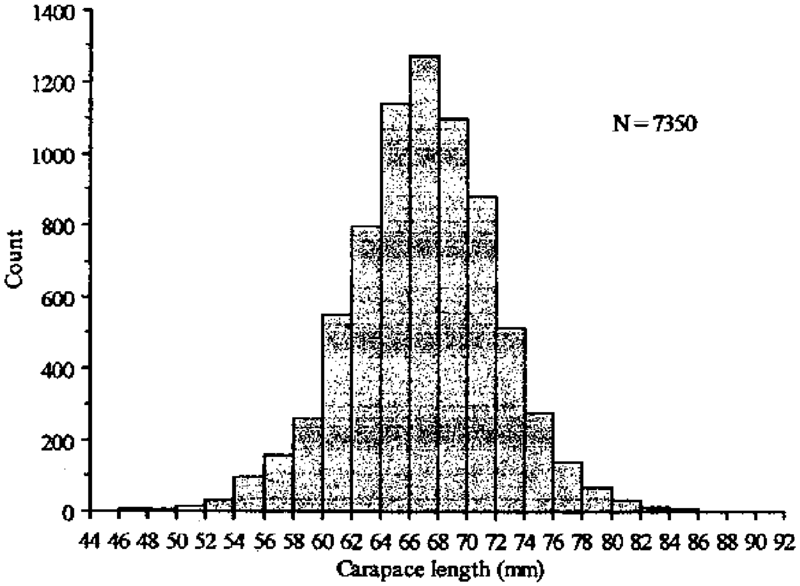


Figure 2. Size frequency distribution of the spotted spiny lobsters sampled over the two years of the experimental fishery

ACKNOWLEDGEMENTS

We wish to thank the following fishers for their cooperation and assistance in collecting the data presented in this paper: Charles Desilva, Arthur Ray, Llew Hollis, Peter Fox, Gene Dublin, Peter Bridges, Andrew Marshall, Nolan Crane and Terry Gibbons.

LITERATURE CITED

Evans, C.R. and A.P.M. Lockwood. 1994. Population field studies of the guinea chick lobster *Panulirus guttatus* (Latreille) at Bermuda: abundance, catchability and behaviour. *J. Shellfish Res.* **1** 3:393-415.

Evans, C.R., A.P.M. Lockwood, A.J. Evans and E. Free. 1995. Field studies of the reproductive biology of the spiny lobsters *Panulirus argus* (Latreille) and *P. guttatus* (Latreille) at Bermuda. *J. Shellfish Res.* **1** 4:371-381.

Evans, C.R., A.P.M. Lockwood, A.J. Evans. 1996. Field studies of the population dynamics of the spotted spiny lobster *Panulirus guttatus* (Latreille) at Bermuda. *Gulf of Mexico Science* **2**: 55-65.

Luckhurst, B.E. and S. Marshalleck. Current status and assessments of the fisheries for spiny lobster and conch in the CARICOM region. *Gulf Carib. Fish. Inst.* **48**: In press.

Luckhurst, B.E. 1999. National Report of Bermuda. P. 180 -182. in: FAO / Western Central Atlantic Fishery Commission. Report on the FAO/ DANIDA/ CFRAMP/ WECAFC Regional Workshop on the Assessment of the Caribbean Spiny Lobster (*Panulirus argus*). FAO Fisheries Report No. 620. Rome, FAO.

Moe, M.A. Jr. 1991. *Lobsters - Florida, Bahamas, The Caribbean.* Green Turtle Publications, Plantation, FL. 510 pp.

Sharp, W.C., J.H. Hunt and W.G. Lyons. 1997. Life history of the spotted spiny lobster *Panulirus guttatus* , an obligate reef-dweller. *Mar. Freshwater Res.* **48**:687-698.

Sutcliffe, W.H. 1953. Notes on the biology of a spiny lobster, *Panulirus guttatus* , in Bermuda. *Ecology* **34**:794-796.

Ward, J.A. and B.E. Luckhurst. 1996. Development of a lobster- specific trap in Bermuda and fisheries management considerations for the re-establishment of a Commercial lobster fishery. *Gulf Carib. Fish. Inst.* **44**:566-578.

Williams, A.B. 1984. *Shrimps, lobsters, and crabs of the Atlantic coast of the eastern United States, Maine to Florida.* Smithsonian Institution Press, 550 pp.

Regression Analysis of the Relationships Among Life-stage Abundances of Brown Shrimp (*Penaeus aztecus*) and Environmental Variables in Southern Louisiana, USA

HEATHER L. HAAS[1], RICHARD F. SHAW[1,2], KENNETH A. ROSE[1,2],
MARK C. BENFIELD[1,2], and WALTER R. KEITHLY[2]
[1]*Department of Oceanography and Coastal Sciences*
[2]*Coastal Fisheries Institute*
Louisiana State University
Baton Rouge, LA 70803 USA

ABSTRACT

Brown shrimp (*Penaeus aztecus* Ives) landings in the Gulf of Mexico display substantial interannual variability. We used regression techniques to analyze relationships among 27 years of postlarval, juvenile, and adult abundance estimates and a suite of environmental variables. Environmental variables included water temperature, salinity, turbidity, river flow rate, acres of suitable habitat, and precipitation. We used a combination of manual and stepwise model building procedures to develop annual models with offshore catch, late-juvenile, early-juvenile, and postlarval abundances as dependent variables. Environmental variables and preceding life history stages were exploratory variables. Commercial catch was described by late-juvenile abundance, water temperature, previous commercial catch, and river flow. The biological variables in the model explained 55% of the variability in offshore catch, whereas the environmental variables explained 24% of the variability. Environmental variables explained variation between each life history stage and were the only significant predictors of postlarval and juvenile abundance. The lack of biological links among early life history stages may result from environmentally-driven, density-independent relationships or biologically-driven, nonlinear relationships. We are designing an individual-based simulation model of shrimp to further explore the relationships among their early life history stages. Both the regression and simulation models are ongoing efforts. We hope the combination of statistical and individual-based simulation modeling will provide further insight into the factors that affect variability in brown shrimp recruitment.

KEY WORDS: Penaeid, brown shrimp, annual recruitment

INTRODUCTION

Like many penaeid fisheries, brown shrimp (*Penaeus aztecus* Ives, *Farfantepenaeus aztecus* Ives) landings in the Gulf of Mexico display substantial interannual variability. Because most of the shrimp caught in the Gulf fishery are less than one year old (Caillouet and Koi 1981), brown shrimp have an essentially annual life cycle. Adults spawn offshore, larvae move toward the coast, juveniles grow within the estuaries, and adults move away from the shallow coastal zone and enter the offshore fishery. The short life cycle and intense fishing pressure should make brown shrimp one of the easiest fisheries to forecast (Matthews et al. 1994).

With the world-wide commercial importance of penaeids, there are many models which attempt to predict shrimp abundance. Although the details of these models vary geographically and by species, they exhibit several common features. Commercial catch is related to several environmental variables including temperature (Barrett and Gillespie 1973, Barrett and Ralph 1977, Hettler 1992), rainfall (Staples 1985, Staples et al. 1995), intertidal vegetation (Turner 1977), and water level in the marsh (Zimmerman and Minello 1984). Many studies that examine biological variables use juvenile abundance to predict subsequent catch (Berry and Baxter 1969, Caillouet and Baxter 1973, Barrett and Gillespie 1973). Some models use postlarval abundance to forecast catch (Delauncy et al. 1994, Matthews 1995). Fewer studies (Vance et al 1998) look for links among the earlier life history stages.

Our objective in this paper is to use regression analysis to better understand the sources of interannual variation in Louisiana's brown shrimp recruitment. Recruitment in this paper is defined as the abundance of shrimp that reach the next life stage. We are interested in the prediction of annual catch and in what regression analysis reveals about penaeid ecology. We used 28 years (1970 – 1997) of biological and environmental data related to brown shrimp recruitment in order to examine the linear relationships among successive life history stages and environmental variables.

METHODS

We assembled a dataset that included annual values of brown shrimp abundance estimates by life stage and monthly or annual values of a suite of environmental variables. The four life stages examined were postlarval (number of shrimp/m^3), early juvenile (number of shrimp/10 minute trawl with six foot net), late juvenile (number of shrimp/10 minute trawl with sixteen foot net), and adult (pounds of offshore catch). Environmental variables examined water temperature, salinity, turbidity, river flow rate, precipitation, and acres of habitat > 10 ‰. Because shrimp may be affected by the environment at critical periods in the spring, annual monthly averages (for February through July) were calculated for salinity, temperature, turbidity, precipitation, and river flow. We dropped February salinity and turbidity because we had incomplete data at the beginning of the season.

The Louisiana Department of Wildlife and Fisheries (LDWF) provided postlarval and juvenile abundance estimates. A detailed description of the data collection procedures can be found in Marine Fisheries Division Field Procedures Manual (LDWF 1996). Table 1 summaries the LDWF sampling protocol for each life stage. In general, LDWF sampled weekly when the brown shrimp densities were known to be historically high. Postlarval data were collected from four major tidal passes in Barataria Bay, which functions as an index for brown shrimp recruitment. We calculated postlarval densities based on tow catch and flow meter readings. Estimates of juvenile abundance in Louisiana were obtained from six and sixteen foot otter-trawl surveys. Because of sample location and mesh size, the six foot trawl selected for smaller shrimp than the sixteen foot trawl. Hence, we used the six foot trawl data to estimate early-juvenile abundance and the sixteen foot trawl data to estimate late-juvenile abundance. The mean total length of the early juveniles was 65 mm, and the mean total length of the late juveniles was 75 mm.

Table 1. LDWF sampling protocol used to monitor the brown shrimp fishery in southern Louisiana.

Stage	Gear	Mesh Size	Sampling Location
Postlarvae	0.5 m plankton net	500 micron	Tidal passes
Early juvenile	6' trawl	3/8 inch	Shallow marshes
Late juvenile	16' trawl	3/4 inch	Bays, sounds, or lakes

We estimated adult abundance by using commercial catch data (in pounds) provided by the National Marine Fisheries Service. We also converted the annual pounds of commercial catch to annual numbers of shrimp caught. The use of numbers reduces effects of differential growth of individual shrimp during the juvenile stage when they are in the estuaries. Very similar results were obtained with numbers and pounds. We used total annual offshore heads-off catch as a dependent variable because offshore catch accounted for over 50% of the total annual recorded catch, and offshore catch was well correlated with total catch ($R^2 = 0.84$).

The LDWF provided two sets of environmental data. The first set of data was acquired in conjunction with the biological sampling of the postlarvae and juvenile abundances, and it included surface and bottom water temperature, surface and bottom salinity, and turbidity. The second set of environmental data was collected independently of the biological sampling, and included an annual estimate of the total number of acres with a salinity greater than 10 ‰, mean monthly precipitation in southern Louisiana (Source: NOAA Climatological Data), and mean monthly Mississippi River (at Tarbert Landing, Miss) and Atchafalaya River (at Simmesport, LA) flow rates (Source: US Corps of Engineers, New Orleans District). Examples of water temperature and river flow variables are shown in Figure 1a.

We did not discriminate between surface and bottom measurements of the environmental variables because the water column in areas of data collection were generally shallow and well-mixed. In cases where the LDWF sampled salinity and temperature at the top and bottom of the water column, we deleted outliers and used simple linear regression models to compare the surface and bottom values for each life history stage (postlarval, early juvenile, and late juvenile). The salinity near the bottom explained > 90% of the variation in surface salinity, and the bottom temperature explained > 97% of the variation in surface temperature. Because brown shrimp are primarily benthic, we used the bottom values of temperature and salinity. If bottom values were not available, we substituted surface values.

We used regression analysis to explore the relationships among life history stages and environmental variables. Specifically, we created regression models to describe the interannual variation in total annual pounds of offshore catch, mean annual late-juvenile abundance, mean annual early-juvenile abundance, and mean annual postlarval abundance. In order to describe each life history stage, we used the entire suite of environmental variables plus all preceding life history stage abundances. Because commercial catch may be an indicator of spawning potential, we used last year's catch (in pounds and numbers) as possible indicators of this year's abundances.

We used a combination of manual and stepwise (automated) regression to build our models. Automated stepwise model building presented potential problems due to the loss of data because of missing values. When multiple regressions were performed using SAS stepwise regression (SAS Institute 1989), the data were limited to years that had a value for all exploratory variables. Thus, initial regressions were run on few data points ($n < 10$) and had inappropriately large correlation coefficients ($R^2 > 0.95$). We therefore used a manual model-building procedure to identify a reduced set of exploratory variables, which were then used in the stepwise procedure. First the dependent variable was correlated to each exploratory variable. A simple linear regression was run using the dependent variable and the exploratory variable with the highest single correlation. The residuals from this regression were then correlated with all of the remaining exploratory variables. A second model was built using the residuals as the dependent values and the exploratory variable with the highest correlation as the new predictor variable. We repeated this procedure until no additional predictor variables were significantly correlated to the residual from the preceding model.

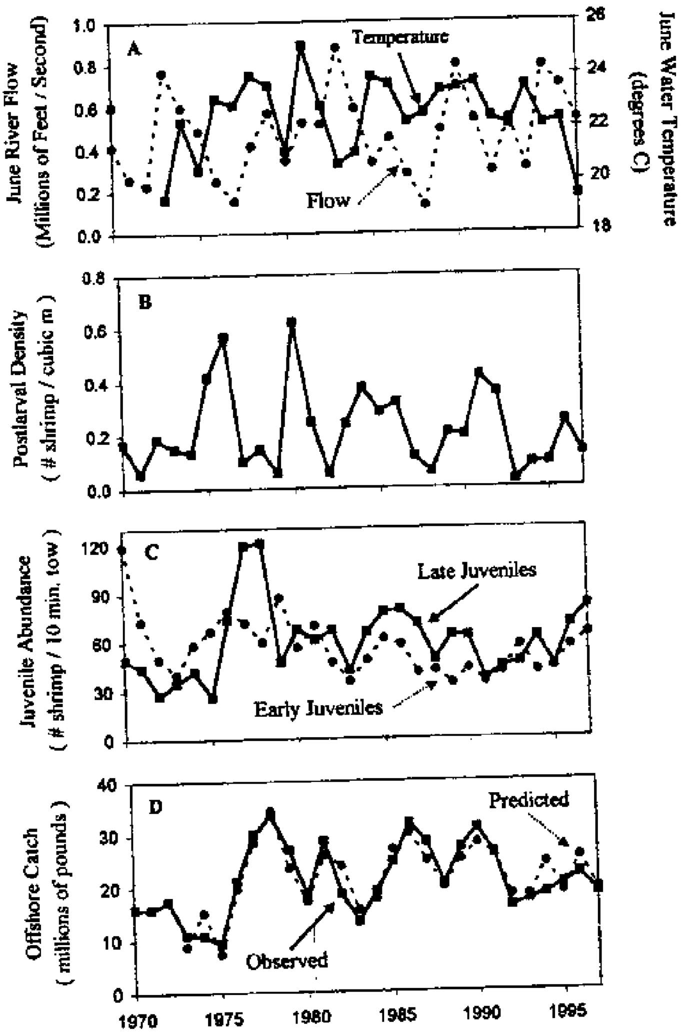


Figure 1. Annual values of selected environmental variables and brown shrimp life stage abundances. **1a.** June estuarine water temperature (°C) and June Atchafalaya and Mississippi River flow rates (10^4 cubic feet/second). **1b.** Postlarval Abundance (# of shrimp/m^3) **1c.** Early and Late-juvenile abundance (# of shrimp/10 minute trawl) **1d.** Offshore commercial catch (in millions of pounds).

We then preformed an automated stepwise multiple regression using the variables that were selected with the manual procedure. The stepwise process allowed the removal of predictor variables in later models. All variables that were not significant at the 0.10 level were removed one at a time from the regression equation. Removing variables changed the regression relationship because it allowed more years to be included in the analysis. We used the resulting regression equation to predict catch for each year in the study period. We report the final models which were identified by the combination of manual and stepwise model building. In an attempt to explore the contribution of environmental and biological variables, the model predicting commercial catch was subsetted into two reduced models. The first reduced model contained only biological variables, and the second reduced model contained the environmental variables.

RESULTS

Commercial catch of brown shrimp in Louisiana was described by late-juvenile abundance, April water temperature, total estimated number of shrimp caught in Louisiana in the previous year, and June Mississippi and Atchafalaya River flow (Table 2, Figure 1d). The reduced biological model explained 55% of the variability in offshore catch, whereas the reduced environmental model explained 24% of the variability in catch.

The early life history stages were correlated with environmental variables but not with preceding life-stage abundances. Late juvenile abundance was correlated with salinity variables and with May water temperature (Table 2). All of the salinity variables, except March salinity, suggested that abundance of late juveniles was higher under saltier conditions. Early juvenile abundance was correlated with turbidity, river flow, precipitation, and salinity (Table 2). Post larval abundance was correlated with temperature in the tidal passes, salinity, precipitation, and river flow (Table 2).

DISCUSSION

Environmental variables helped explain the variation between each life history stage and were the only significant predictors of the postlarval and juvenile abundances. In general, environmental conditions in the later months (May and June) were correlated more closely with the later life stages (late-juveniles and adults) and earlier months (February and March) were correlated more closely with the earlier life stages (postlarvae and early-juveniles). Although there were exceptions, higher shrimp abundance was generally associated with higher salinity and warmer water temperature.

Table 2. Final regression models from the analysis of brown shrimp life stage abundances and environmental variables. Final models are those determined from the combination of the manual and stepwise model building procedure.

Dependent Variable	Predictor Variables (Positive or Negative Relationship)	n	R^2	P
Commercial Catch	Full Model: + Late juvenile abundance + April estuarine water temperature + Number of shrimp caught last year + June river flow	25	0.85	p < 0.0001
	Reduced Biological Model: + Late juvenile abundance + Number of Shrimp caught last year	27	0.55	p < 0.0001
	Reduced Environmental Model: + April estuarine water temperature + June river flow	26	0.24	p < 0.0454
Late Juveniles	+ March estuarine salinity + June estuarine salinity - Annual precipitation (in SE LA) + May estuarine water temperature + May precipitation (in SE LA) - March salinity (in lower estuary)	21	0.87	p < 0.0001
Early Juveniles	- June estuarine turbidity - February river flow - Annual precipitation (in SE LA) - March salinity (in lower estuary)	23	0.66	p < 0.0005
Postlarvae	+ February temperature (in passes) - May temperature (in passes) + March salinity (in lower estuary) - February precipitation (in SE LA) + April river flow	20	0.77	p < 0.0004

As is the case with most regression analysis, correlation does not necessarily mean causation. Variables selected by the regression procedure may represent a controlling mechanism or may simply be correlated with a controlling mechanism. For example, June turbidity was selected as a significant predictor of early juvenile abundance. Turbidity might directly affect juvenile abundance by affecting predator prey relationships (Minello et al. 1987). Alternatively, turbidity may simply be correlated with a controlling mechanism, such as wind-driven events that affect postlarval transport and salinity regimes. The lagged catch variable is a second example of an uncertain controlling mechanism. Whereas last year's catch may reflect this year's spawning potential, it may also be a reflection of trends in expended effort and gear efficiency.

Caution in the interpretation of the final regression models is appropriate due to the multicollinearity among predictor variables. Collinearity can arise from covariance among related variables (e.g. salinity and precipitation) and from using several monthly values of a single environmental variable (e.g. April temperature and May temperature). Because each iteration of the selection procedure chose only one variable for inclusion, several well-correlated variables were not included in the prediction equation. For example, the selection procedure included late juvenile abundance ($r = 0.62$) but did not include annual mean turbidity ($r = 0.56$). Correlated environmental variables might help explain the inclusion of suspicious variables. For example, May temperature predicted postlarval abundance even though the postlarval abundance peak is in March and April.

Whereas environmental variables were significant predictors at all life history stages, the biological link among abundances at successive life history stages was only detected between the late-juvenile and adult abundance. Failure to document the biological links among the early life history stages could be explained in several ways. One explanation is that we have inappropriate or insufficient data to detect relationships in abundance. The juvenile abundance estimates may be inappropriate because trawl data can be qualitative rather than quantitative measures of abundance (Rozas and Minello 1997). The postlarval data stage is especially susceptible to insufficient sampling due to the difficulties in capturing pulses of immigrants. Nevertheless, we presume that because the LDWF sampling protocol captured the pattern in the late-juvenile to adult abundance, it could also capture an abundance pattern in the early to late-juvenile stage.

Assuming we have appropriate and sufficient data to detect relationships among successive life stages, the lack of biological links among early life history stages may result from environmentally-driven, density-independent relationships and density-dependent, nonlinear relationships. To explore these possibilities, we are designing an individual based model to simulate early life history stages. The simulation model begins at post-larval immigration into the estuaries and tracks individual shrimp in a spatial grid of land and water cells until the shrimp emigrate back to coastal waters. Habitat-dependent growth and mortality are recorded at each tidal cycle. Initial simulations indicate the important, but complex, role of edge habitat in affecting shrimp survival within the estuary.

Both the regression and simulation models are ongoing efforts that are still being refined. We plan to expand the regression model by adding variables associated with estuarine water levels, acres of vegetated habitat, shrimping effort, and timing of the inshore fishing season. Using a biological year (Klima et al. 1982) rather than a calendar year might also strengthen the regression relationships. After we expand the regression model, we will examine regression diagnostics and then use the same regression procedures to examine inshore shrimp catch. The pounds caught inshore and offshore are not well correlated ($r^2 = 0.22$) and may be related to a different suite of variables. We are also considering alternative analytical methods to regression, such as neural network analysis and general additive models, to explore any relationships among the early life stages. We hope the combination of statistical and individual-based simulation modeling will provide further insight into the factors that affect variability in brown shrimp recruitment.

ACKNOWEDGEMENTS

We would like to thank Louisiana Department of Wilflife and Fisheries and National Marine Fisheries Service for providing the data and the Louisiana Board of Regents and the Louisiana Sea Grant College Program for providing research funds.

LITERATURE CITED

Barrett, B.B. and E.J. Ralph. 1977. 1977 environmental conditions relative to shrimp production in coastal Louisiana along with shrimp catch data for the Gulf of Mexico. La. Dept. Wildl. Fish. Tech. Bull. **26**: 1-16.

Barrett, B.B. and M.C. Gillespie. 1973. Primary factors which influence commercial shrimp production in coastal Louisiana. La. Wdlf. Fish. Comm. Tech. Bull. **15**:1.

Berry, R.J. and K.N. Baxter. 1969. Predicting brown shrimp abundance in the northwestern Gulf of Mexico. Pages 775 - 798 in: M.N. Mistikadis, (ed.) Proceedings of the World Scientific Conference on the Biology and Culture of Shrimps and Prawns, Mexico City, Mexico 12-21 June

1967, Volume III - Experience Papers, FAO Fish. Rep. No. 57, Vol. 2.

Caillouet, C.W. and D.B. Koi. 1981. Trends in ex-vessel value and size composition of reported May-August catches of brown shrimp and white shrimp from the Texas, Louisiana, Mississippi, and Alabama coasts, 1960-1978. *Gulf Res. Rep.* **7**:59.

Caillouet, C.W. and K.N. Baxter. 1973. Gulf of Mexico shrimp resource research. *Mar. Fish. Rev.* **35**:21-24.

Delauncy, L.B., J.E. Jenkins and J.D. Whitaker. 1994. Results of long-term, seasonal sampling for Penaeus postlarvae at Breach Inlet, South Carolina. *Fishery Bulletin* **92**: 633-640.

Hettler, W.F. 1992. Correlation of winter temperature and landings of pink shrimp *Penaeus duorarum* in North Carolina. *Fish. Bull.* **90**:405-406.

Klima, E.F., K.N. Baxter and F.J. Patella, Jr. 1982. A review of the offshore shrimp fishery and the 1981 Texas Closure. *Mar. Fish. Rev.* **44**:16-30.

Louisiana Department of Wildlife and Fisheries. 1996. Marine fisheries division field procedures manual. Louisiana Department of Wildlife and Fisheries Office of Fisheries, Marine Fisheries Division. Version No. 96-1. Unpubl. MS. 25 pp.

Matthews, G.A. 1995. Developing a new forecasting model for the annual catch of brown shrimp, *Penaeus aztecus*, in the Gulf of Mexico off Texas. National Marine Fisheries Service, Galveston, TX. Unpubl. MS. 45 p.

Matthews, G.A., K.N. Baxter and F.J. Patella. 1994. Two models tat forecast the annual catch of brown shrimp, *Penaeus aztecus* Ives, in the Gulf of Mexico off Texas. National Marine Fisheries Service, Galveston, TX. Unpubl. MS. 19 p.

Minello, T.J., R.J. Zimmerman, and E.X. Martinez. 1987. Fish predation on juvenile brown shrimp, *Penaeus aztecus* Ives: Effects of turbidity and substratum on predation rates. *Fish. Bull.* **85**(1):59-70.

Rozas L.P. and T. J. Minello. 1997. Estimating densities of small fishes and decapod crustaceans in shallow estuarine habitats: a review of sampling design with focus on gear selection. *Estuaries.* **20**(1):199-213.

SAS Institute Inc., SAS/STAT User's Guide, Version 6, Fourth Edition, Volume 2, Cary, NC: SAS Institute Inc., 1989. 846 pp.

Staples, D.J. 1985. Modelling the recruitment processes of the banana prawn, *Penaeus merguinsis*, in the southeastern Gulf of Carpentaria, Australia. Pages 175-184 in: P.C. Rothlisberg, B.J. Hill, and D.J. Staples, (eds.) *Second Australian National Prawn Seminar*. NPS2 Cleveland, Australia.

Staples, D.J., D.J. Vance, N.R. Loneragan. 1995. Penaeid prawn recruitment variability: effect of the environment. Pages 41 - 50 in: A.F. Courtney and M.G. Cosgrove (eds) *Proceedings of the Workshop on Spawning Stock-Recruitment Relationships (SRRs) in Australian Crustacean Fisheries*. Department of Primary Industries, Bisbane, Queensland.

Turner, R.E. 1977. Intertidal vegetation and commercial yields of penaeid shrimp. *Trans. Am. Fish. Soc.* **106**:411-416.

Vance, D.J., M.D.E. Haywood, D.S. Heales, R.A. Kenyon and N.R. Loneragan. 1998. Seasonal and annual variation in abundance of postlarval and juvenile banana prawn *Penaeus merguiensis* and environmental variation in two estuaries in tropical northeastern Australia: a six year study. *Mar. Ecol. Prog. Ser.* **163**:21-36.

Zimmerman, R.J. and T.J. Minello. 1984. Densities of *Penaeus aztecus, Penaeus setiferus*, and other natant macrofauna in a Texas salt marsh. *Estuaries*. **7**(4A):421-433.

Variables Ambientales como Factores Importantes en la Distribución y Abundancia de Juveniles de Camarón Blanco (*Litopenaeus setiferus*) en la Laguna de Términos, Sur del Golfo de México

J. RAMOS-MIRANDA[1], A. SOSA-LÓPEZ[1],
D. FLORES-HERNÁNDEZ[1] y T. DO-CHI[2]

[1]*Centro EPOMEX, Universidad Autónoma de Campeche*
CP. 24030, Campeche, Mexico
[2]*Ecosistèmes Lagunaires, Université Montpellier*
II.UMR CNRS 5556
Montpellier Cedex 5, France

RESUMEN

El camarón blanco es uno de los principales recursos pesqueros de las costas de Campeche, su captura no sólo se realiza en altamar sino en la costa y aguas interiores como la Laguna de Términos. Debido a la alta productividad de esta laguna y al ciclo de vida de la especie, en esta laguna se encuentran los juveniles que permanecen hasta alcanzar la edad suficiente y migrar al mar para crecer y reproducirse. Los factores ambientales que influyen en la abundancia y distribución de la especie son abordados en el presente trabajo durante el periodo de octubre de 1997 a marzo de 1999. Se observa un patrón estacional marcado en el que la mayor abundancia (número de individuos) se presenta durante la temporada de lluvias (junio-julio) y nortes (noviembre). La distribución espacial del camarón blanco muestra una mayor concentración en la región suroeste de la laguna, como lo reportado en los estudios de los años 1970s y 1980s. Este patrón está asociado al aporte de agua dulce, sedimentos limosos y ausencia de vegetación sumergida. Ninguna variable físico-química parece estar relacionada directamente con la abundancia del camarón blanco. Es probable que el efecto conjunto de dichas variables tenga una mayor influencia sobre la abundancia. Existe una disminución entre los valores máximos densidad (0.25 ind/m^2) obtenidos en este estudio, con respecto a los reportados por Gracia (1989), densidad = 1.10 ind./m^2. La duración de este trabajo no permite explicar la disminución de la densidad reportada, sería conveniente mantener un monitoreo permanente de variables ambientales e impactos antropogénicos.

PALABRAS CLAVE: Factores ambientales, Golfo de México Laguna de Términos *Litopenaeus setiferus*

ABSTRACT

The white shrimp is one of the most important fishery resources of the coasts of Campeche, its catches are carried out offshore sea but also in the coastal lagoons as Terminos Lagoon. Due to high productivity and type of life cycle of this specie, in this lagoon are found the juveniles that stay until to have necessary age to migrate to high sea and reproduced. The environmental factors that influence shrimp abundance and distribution are analyzed in this work during October of 1997 to March of 1999. There are observed a strong seasonal pattern, where the largest abundance (number of individuals) is presented during rains season (June to July) and storms season (November). The spatial distribution of the white shrimp showed the most concentration at southwest region of the lagoon, as was reported in the studies of the years 70's and 80's. This distribution was associated with freshwater contribution, slimy sediments and absence of submerged vegetation. Not relationships between physico-chemistry parameters were found directly with the abundance of the white shrimp. It is probably that the effect joint of such variables has a larger influence on the abundance. There are observed a decrease between the maximum density values (0.25ind/m^2) obtained in this study, with respect in comparison with reported by Gracia (1989), density = 1.10 ind./m^2. Duration of this work does not permit to explain the decrease of the reported density, would be convenient to support a permanent monitoring of environmental variables and human impacts.

KEY WORDS: Environmental factors, Gulf of Mexico, Terminos Lagoon, *Litopenaeus setiferus*

INTRODUCCIÓN

La región de la Laguna de Términos ecosistema asociado a la Sonda de Campeche, es particularmente importante debido a su complejidad biológica. Por ejemplo 43 especies entre peces, crustáceos y moluscos explotados en la Sonda de Campeche, utilizan la laguna como zona de crianza, alimentación y/o reproducción en algún momento de su vida (Sánchez-Gil et al. 1994). Entre estas especies, el recurso camarón sobresale por su gran importancia; comercialmente aportó 17,713 toneladas anuales en 1997 (SEMARNAP 1997) y ecológicamente sirve como fuente de alimento a un numeroso grupo de peces (Rivera Arriaga et al. 1993). Cuatro especies de camarón de importancia comercial se distribuyen en la Sonda de Campeche y Laguna de Términos: café (*Farfantepenaeus aztecus*) rosado (*Farfantepenaeus duorarum*), siete barbas (*Xiphopenaeus kroyeri*) y blanco (*Litopenaeus setiferus*). Este trabajo se enfoca sobre la última especie mencionada. Gracia (1989), describe el ciclo de vida del camarón blanco para la Sonda de Campeche- Laguna de Términos señalando que una vez efectuado el desove en el mar, las postlarvas penetran durante la primavera y en el otoño, y permanecen apróximadamnete tres meses hasta alcanzar la etapa de preadultos (tres a quatro meses de edad), momento en que migran hacia el mar para madurar y reproducirse en aguas marinas.

Las relaciones entre la abundancia y distribución del camarón con factores abióticos han sido estudiados por diversos autores desde la etapa larval hasta la adulta (Zein-Eldin y Renaud 1986). García y Le Reste (1981) sugieren que las especies dependientes del estuario están sujetas a gran variabilidad de mortalidad natural, señalando que la sobrevivencia depende de la disponibilidad del hábitat, disponibilidad de alimento, depredación, influencia antropogénica y mortalidad por pesca. Rothlisberg et al. (1985a) señalan que la disminución de la salinidad en el estuario provoca la emigración de juveniles hacia aguas mas salinas de mar abierto. Zimmerman y Minello (1984a), Zimmerman y Minello. (1984b) demuestran que el camarón blanco es más abundante en áreas sin vegetación sumergida y fondos blandos.

Particularmente en la Laguna de Términos, Signoret (1974), determinó la distribución y abundancia de las especies de camarón en esta laguna señalando las relaciones entre cada especie con el aspecto ambiental lagunar, el autor señala además que existe una relación entre la abundancia con la precipitación anual. Gracia (1989 a y b) y Gracia et al. (1997), señalan que existe una relación entre la abundancia de juveniles y la descarga de ríos, indican también que este actúa como estimulante para la emigración hacia el mar. El mismo aspecto fue señalado por Rothilsberg et al. (1985b) y Staples y Vance (1986). Actualmente se han reportado cambios importantes en las condiciones ambientales del hemisferio norte, asociados con el cambio climático global, estos cambios han sido detectados en la dinámica ambiental de la laguna de Términos, observándose para algunas especies de peces, cambios de hábitat, así como de abundancia (Ramos-Miranda et al. 1999); por lo que es importante determinar el impacto de este cambio en la distribución y abundancia de camarón blanco en la región. Con este marco de referencia, el presente trabajo pretende identificar los factores ambientales asociados a la distribución y abundancia de juveniles de camarón blanco presentes en la Laguna de Términos.

AREA DE ESTUDIO

La Laguna de Términos se localiza en el Sur del Golfo de México (Fig.1), en la Península de Yucatán (91° 00' y 92° 00' W; 18° 20' y 19° 00' N). Presenta una longitud de 70 km y 26 km de ancho. Su superficie total es de 1 661.50 km2 sin contar los sistemas fluvio-lagunares asociados. Esta laguna tiene una profundidad media de 3.5 m. Al norte se encuentra una isla de barrera que divide a la laguna con el mar, se comunica por dos bocas ubicadas al oeste y este de la isla; las bocas del Carmen y Puerto Real, respectivamente. Tres ríos principales descargan sus aguas a la Laguna, el río Palizada (233.74 m^3/seg) al Suroeste de la laguna, perteneciente a la red hidrológica del Grijalva-Usumacinta cuyos aportes son los más importantes en todo el año, el río Chumpan (34.55 m^3/seg) al sur y el río Candelaría (63.60 m^3/seg) al este. Tres épocas climáticas caracterizan la región, la época de nortes octubre a enero, la época de secas de febrero a mayo y la época de lluvias de junio a septiembre (Yáñez-Arancibia y Day 1982). Durante todo el año, se presentan vientos provenientes del este y Sureste principalmente; a partir del mes de octubre los vientos provienen del norte (velocidades de 40 - 90 km./hora) juegan un papel importante en la circulación de la laguna, presentándose un flujo de agua con dirección este oeste. En la época de norte, el agua puede penetrar también por la Boca del Carmen, afectando el tercio oeste de la laguna. Los sedimentos al interior de la laguna de acuerdo a Yáñez Correa (1971) se distribuyen de la siguiente manera: al suroeste de la laguna y la Boca del Carmen, se encuentran sedimentos limo arcillosos; en la boca de Puerto Real y el interior de la Isla del Carmen se encuentran arenas; parte del este, noroeste sur y centro de la laguna predominan arena y limo arcilla. Asociado a los fondos de arena y alta transparencia se encuentran bancos de pastos marinos, otras áreas con esta vegetación, pero menos abundancias, están asociados a fondos de limo y arena arcillosa hacia el este de la laguna (Vargas Maldonado et al. 1981).

MATERIALES Y MÉTODOS

Muestreo

Se realizaron muestreos mensuales desde Octubre 1997 a Marzo 1999 en 23 estaciones distribuidas en todo el cuerpo lagunar (Figura 1), en cada estación se realizó un arrastre experimental con una red de prueba camaronera "chango" de 5 metros de largo, 5 metros de ancho, 2.5 m de abertura de trabajo, con una luz de malla de 3/4". La red esta provista de dos puertas laterales de 50 cm de largo X 40 cm de ancho. Los arrastres experimentales se realizaron a bordo de una lancha de 27" de eslora, con motor fuera de borda de 65 HP, cada arrastre tuvo una duración de 12 min., a una velocidad de 2.5 nudos, cubriendo un área de 2,000 m^2 de acuerdo a la técnica de Loesh et al. (1976), para especies similares y con el fin de realizar comparaciones con estudios realizados en la década de los 70's y 80's (Signoret 1974, Smith, 1984 y Gracia 1989). Además se midieron variables físico-químicas del agua (temperatura, salinidad, oxígeno disuelto y pH de superficie y fondo) utilizando un Hydrolab HL 2011; asimismo se registró la transparencia (Secchi) y profundidad (Sondaleza). La estación de muestreo fue georeferenciada (GPS Garmin con precisión de 80 m). Los individuos capturados fueron metidos en bolsas plásticas previamente etiquetadas y permanecieron en hielo hasta su llegada al laboratorio. En laboratorio los ejemplares fueron lavados, separados, identificados, medidos en longitud total (desde la punta del rostro hasta la punta del telson), pesados y el sexo fue determinado. Para la identificación se utilizaron las obras de Fisher (1978), publicadas por FAO; así como Pérez Farfante (1970) y Pérez Farfante y Kensley (1997).

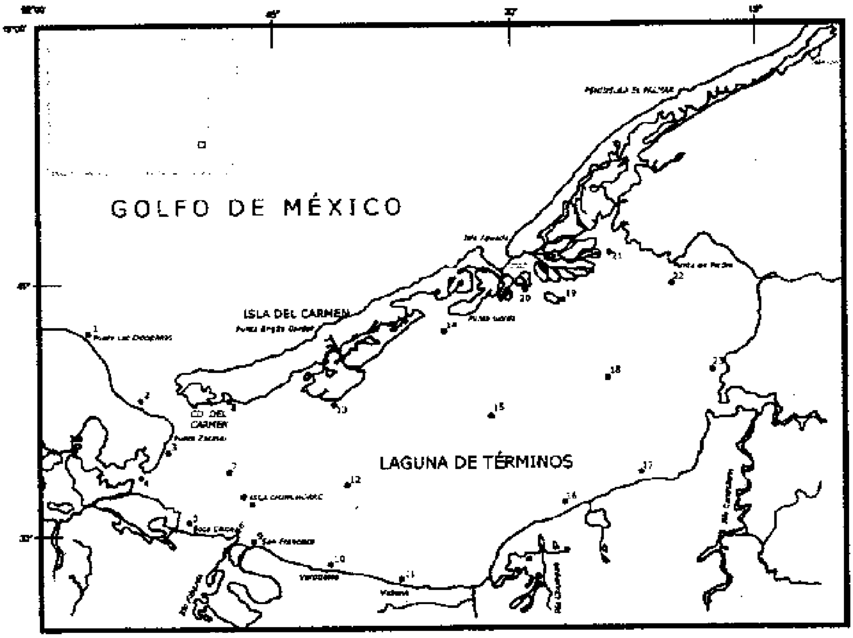


Figura 1. Area de estudio y estaciones muestreo.

Análisis de Datos

Tomando en cuenta la estacionalidad (Nortes, Secas, y Lluvias) definida en estudios previos para la Laguna de Términos (Yáñez-Arancibia y Day 1982, Yánez Arancibia et al. 1988) y asumiendo la variabilidad climatica temporal se realizó un análisis basado en la elaboración de diagramas TS (temperatura-salinidad), a partir de los datos generados en cada una de estaciones durante el período de muestreo.

La distribución espacial de la abundancia en número de individuos se modeló estacionalmente por medio del software SURFER (Smith Lead et al. 1985), aplicando el kriging (basado en el modelo esférico) como técnica de interpolación espacial, bajo el supuesto de isotropía y estacionalidad de segundo orden.

Con el objeto de diferenciar zonas al interior de la Laguna, de acuerdo con la heterogeneidad de la distribución espacial de las variables ambientales relacionadas con la abundancia y la talla de los individuos de camarón, se realizó un análisis factorial (Componentes Principales), para lo cual se utilizó el software STATISTICA.

La densidad media mensual (individuos/m^2) para la zona de mayor concentración de la especie (380 km^2) situada en la región suroeste de la laguna, reportada por Gracia et al. (1989), fue calculada y comparada con los resultados del autor mencionado.

RESULTADOS

Estacionalidad

Los diagramas de temperatura-salinidad (TS) por estación de muestreo mensual durante el periodo analizado (Figura 2) permitieron identificar meses con características de temperatura y salinidad similares en cada estación de muestreo, lo que permitió confirmar la estacionalidad reportada, por Yáñez-Arancibia y Day (1982), y Yáñez Arancibia et al. (1988).

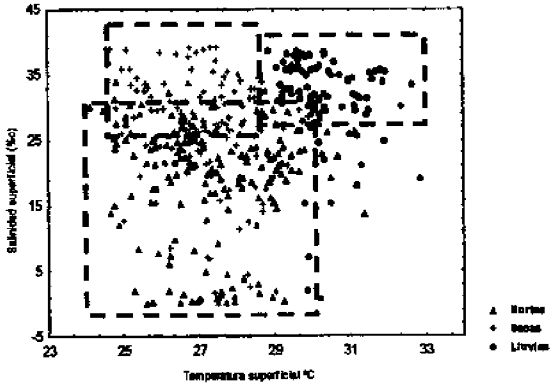


Figura 2. Diagrama TS basado en el total de estaciones de muestreo, indicándose la época climática.

Un total de 4,785 individuos de camarón blanco fueron contabilizados durante el periodo analizado; la Figura 3, presenta el porcentaje de importancia en número de individuos para las diferentes épocas climáticas. Se observa que las mayores abundancias se presentan durante la temporada de lluvias 98 y nortes 98-99, que corresponden al 44.06% (n = 2,076) y 29% (n = 1,437) del total.

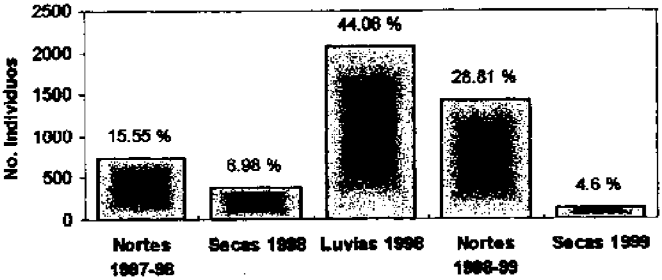


Figura 3. Variación de la abundancia en número de individuos de camarón blanco por época climática.

Variabilidad Ambiental

La abundancia en número de individuos durante el periodo estudiado fue analizada tomando en cuenta la variabilidad ambiental de las características físico-químicas del agua. La Figura 4, muestra las fluctuaciones mensuales de la temperatura, la salinidad, el oxígeno disuelto y el pH de la zona estudiada. Se observa que las mayores abundancias (n = 2,076) se presentan en los meses del año en que las temperaturas son altas (> a 30°C, junio-julio) con una talla modal de 6.85 cm, talla mínima de 3 cm y máxima de 13.5 cm. El mismo patrón de valores altos se observa con la salinidad (30 o/oo) y el pH (> 9). El oxígeno disuelto muestra un comportamiento inverso presentándose altas concentraciones de oxígeno (> 7 mg/L) y menores abundancias (n = 132 y n = 393) durante enero-marzo. Sin embargo, durante el mes de noviembre de 1998, se observan las mayores abundancias (n = 831) al momento en que el oxígeno disuelto se incrementa nuevamente (> 6.5 mg/L).

Distribución Espacial

La Figura 5, muestra la distribución espacial de los ejemplares analizados tomando en cuenta la estacionalidad. Los 4,785 ejemplares se encontraron distribuidos en mayor proporción en las estaciones de muestreo ubicadas al oeste de la laguna: estaciones, 2, 3, 4, 5, 6, 9, 10 y 11 (Figura 1); aunque en la parte sur y este también se encuentra presente en ciertos meses del año (noviembre): estaciones 16 y 23. La mayor cantidad de individuos (n = 1,310) se registró en la estación 4 que corresponde al 27.4 % del total. Se observa que durante la época de nortes 1997 - 1998, la mayor cantidad de individuos (n = 751) se concentran en la parte suroeste de la laguna, con talla media de 8.25 cm, mínimas de 3.00 y máximas de 13.5 cm. En la temporada de secas 1998, la abundancia diminuyó a la mitad de su valor de nortes (n = 393); la talla mínima observada se presentó a los 3 cm, media de 8.77 y máxima de 15 cm. Durante lluvias 1998, época en que se presenta la mayor abundancia (n = 2,076), los individuos se concentraron en la parte oeste de la laguna, particularmente en la boca del sistema Pom-Atasta; estos individuos presentaron tallas medias de 8.54 cm, con un mínimo de 2.50 y máximo de 15.00 cm. Durante la estación secas 1999, se presenta un patrón similar al de secas 1998 en cuanto a distribución, abundancia y tallas; finalmente durante la temporada de nortes 1998 - 1999, se observa una diferencia comparada con la temporada de nortes 1997 - 1998, esta diferencia es marcada en cuanto al número de individuos totales capturados (751 y 1,435 respectivamente), como en su distribución. En nortes 1997 - 1998 los individuos se concentran en la parte oeste y Suroeste de la laguna, mientras que en nortes 1998 - 1999, la distribución abarca además la parte sur.

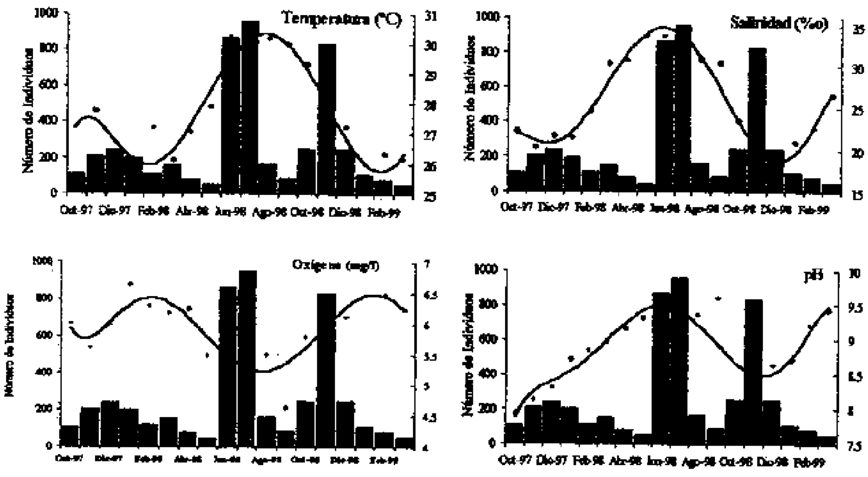


Figura 4. Comparación mensual entre parámetros ambientales y la abundancia en número de individuos de camarón blanco.

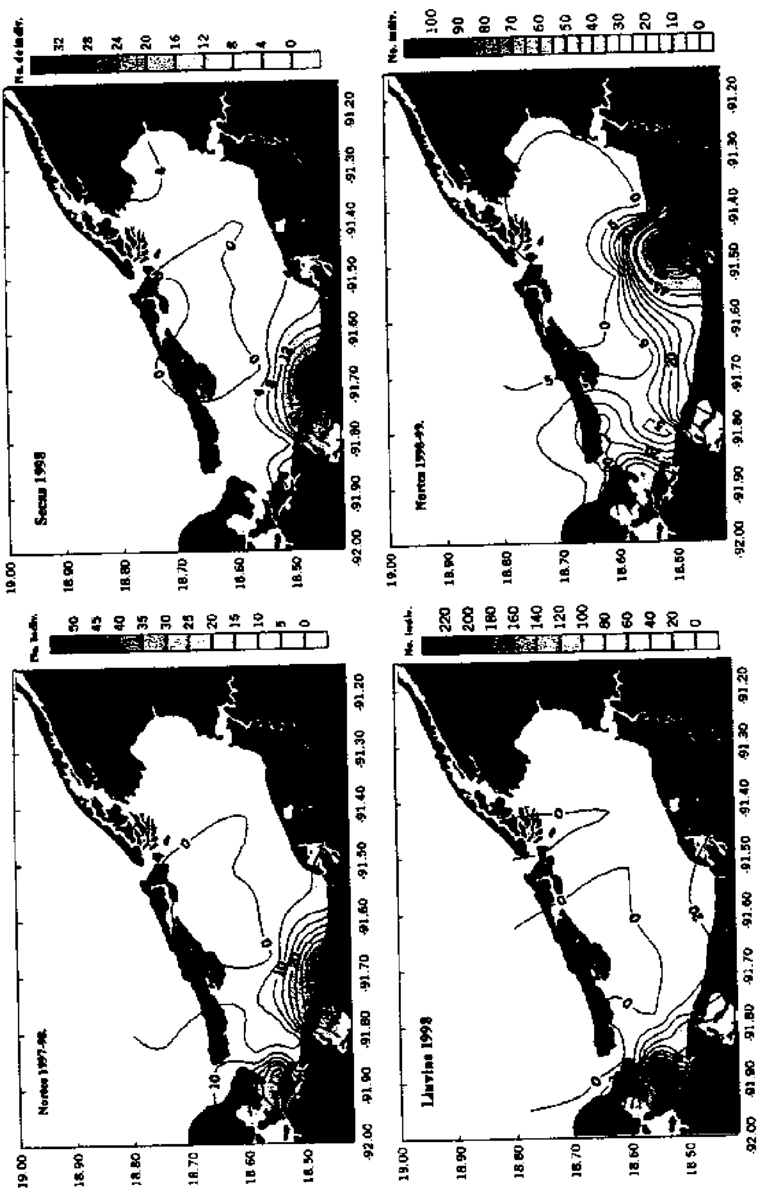


Figura 5. Distribución en número de individuos de camarón blanco en cada una de los períodos climáticos.

El análisis factorial por época climática agrupó estaciones de muestreo definidas por su propia variabilidad (Figura 6A y B) . En el presente trabajo solo se muestran los resultados para las temporadas de lluvias 98 y nortes 98-99 ya que son los meses asociados a los momentos de mayor abundancia de la especie. La Figura 6A, muestra las estaciones agrupadas por factor para la temporada de lluvias; las estaciones agrupadas al factor 1, aportan 67% de la variabilidad, sin embargo corresponden a zonas de poca o nula abundancia de camarón, el grupo que separa al factor 2 (23% de la varianza observada), corresponde a estaciones asociadas a la parte oeste de la laguna, particularmente se localizan en las bocas de la región de Atasta y del río Palizada (estaciones 4, 5 ,6 y 9). En la Tabla 1 se presentan las variables medidas para cada factor en las temporadas de lluvias y nortes. Durante lluvias, el factor 1 muestra que de las variables ambientales analizadas se agruparon las estaciones cuya salinidad elevada (valor mediano 34.34 o/oo), amplio rango de transparencia (93.7 a 14.5 %) y oxígeno disuelto bajo (5.0 mg/l) presentan poca abundancia (n = 1.5). Por el contrario, el factor 2 asoció salinidades bajas (26o/oo), poca transparencia (mínima de 19.5% y máxima de 43.9%), elevadas temperaturas (30.52°C), mayores abundancias (mediana de 71.25) y tallas medianas de 6.84 cm.

La Figura 6B, muestra las estaciones agrupadas por factor para la temporada de nortes 1998 - 1999; la mayor variabilidad en el factor 1 (52%) fue inducida por la baja salinidad (mediana de 24.5 o/oo), aumento en el oxígeno disuelto (6 mg/L), disminución en la temperatura (27.18°C), rangos de transparencia elevados (80.6 a 19%) y disminución en el valor de pH (8.56), a la que se asociaron pocos individuos (n = 24) y tallas elevadas (mediana de 9.48 cm). El factor 2, asoció las estaciones localizadas en la región oeste y sur de la laguna (1-5, 10, 11 y 16). Es importante señalar que es justamente durante esta temporada cuando se presenta el mayor aporte de agua dulce por los ríos, provocando con ello bajas salinidades media (16.57o/oo), con rangos de 20.3-3.1o/oo. Por otra parte los vientos del norte y sureste, provocan que el movimiento del agua, lo que se manifiesta en un aumento en la concentración de oxigeno (mediana de 6.74 mg/L), que se asocia a mayores abundancias (n = 109.75 máxima promedio), pero de individuos con tallas menores (7.98 cm).

Densidad

Los valores de densidad durante las diferentes épocas del año fluctuaron entre 0.008 (ind./m^2) en secas '98 y 0.043 (ind./m^2) lluvias '99. Las densidades más altas se presentaron en junio y julio con 0.228 y 0.253 (ind./m^2) respectivamente, en la región de la boca de Atasta (estación 4) y 0.199 (ind./m^2) en la región sur (estación 16) durante la temporada de lluvias y nortes respectivamente.

Tabla 1. Valores de respuesta producto del análisis factorial.

	Factor I		Factor II	
	Máximo – Mínimo	Mediana	Máximo - Mínimo	Mediana
Lluvias 1998				
Número de individuos	19.00 - 0.00	1.50	241.00 - 31.50	71.25
Talla (cm)	10.71 - 6.80	7.95	7.85 - 6.43	6.84
Temperatura (ºC)	31.91 - 28.75	29.77	31.33 - 30.00	30.52
Salinidad (‰)	37.10 - 28.09	34.34	32.99 - 20.45	26.05
Oxígeno (mg/l)	6.66 - 3.78	5.04	6.35 - 5.12	5.59
pH	9.52 - 9.20	9.47	9.55 - 9.16	9.47
Transparencia (%)	93.75 - 14.58	42.51	43.91 - 19.52	39.42
Nortes 1998-99				
Número de individuos	24.00 - 0.00	0.00	109.75 - 19.75	32.5
Talla (cm)	12.13 - 9.40	9.48	8.93 - 6.48	7.98
Temperatura (ºC)	28.28 - 26.44	27.18	28.74 - 26.93	27.40
Salinidad (‰)	28.98 - 19.30	24.52	20.25 - 3.1	16.57
Oxígeno (mg/l)	7.31 - 4.99	6.08	7.4 - 5.80	6.74
pH	8.72 - 7.61	8.56	8.86 - 7.79	8.42
Transparencia (%)	80.63 - 18.19	47.98	59.02 - 13.82	29.11

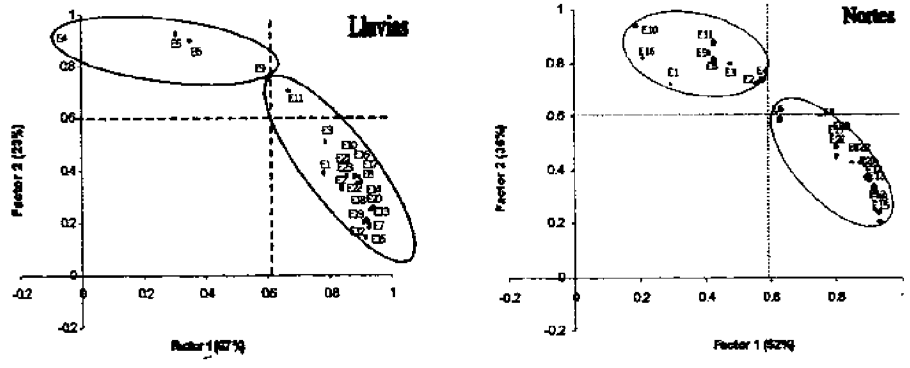


Figura 6. Análisis de factores basado en los distintos parámetros ambientales, así como la abundancia en número de individuos y tallas de camarón por estación

DISCUSIÓN

Estacionalidad

Las mayores abundancias observadas durante lluvias 98 y nortes 98-99 coinciden con lo reportado por Smith (1984), Gracia (1989) y Gracia et al. (1997) quienes señalan la mayor abundancia de juveniles en la laguna durante la temporada de lluvias y nortes (junio a diciembre) y las menores de febrero a mayo; pero difieren con lo señalado por Signoret (1974) quien observa la mayor abundancia de la especie en la laguna durante la primavera, y las menores en otoño. Estas diferencias en la abundancia pudieran ser causadas por una variabilidad temporal en la reproducción de la especie en la plataforma adyacente la cual según Gracia (1989) y Ré Regis (1996) se presentan dos periodos principales, uno a fines de primavera-inicio de verano y otro en otoño.

Variabilidad Ambiental

El patrón de variación del número de individuos de camarón fluctuando positivamente con la temperatura, salinidad y pH, mostrado por los resultados contrasta con lo reportado por Signoret (1974), quien indica que la abundancia no está relacionada directamente con las variables ambientales analizadas; no obstante señala que los individuos de mayor talla se encuentran asociados a aguas de alta transparencia. Por otra parte, Smith (1984) concluye que la abundancia estuvo más relacionada con cambios de temperatura que con las diferencias de salinidad entre las épocas de lluvia en Atasta (zona donde coinciden los reportes de mayor abundancia de camarón blanco en los estudios mencionados). Gracia et al. (1989) manifiestan que la lluvia y la descarga de ríos tiene un efecto en la abundancia de los juveniles, relacionando el aporte de nutrientes al sistema, más que al cambio en la salinidad.

Variabilidad Espacial

Los resultados del análisis factorial indican que la abundancia del camarón blanco se concentra en el suroeste de la laguna en la época de lluvias, bajo la influencia de aportes de agua dulce a través de los ríos Palizada y Chumpán, aporte de sedimentos terrígenos, nutrientes y ausencia de vegetación sumergida. Durante la época de nortes la abundancia se concentró en la región sur, cercana a la desembocadura del río Chumpán, zona con sedimentos limo y arenas arcillosas. Cabe señalar que nunca se presentaron ejemplares en las zonas de distribución de pastos marinos, como es el caso del camarón rosado y café, cuyas distribuciones se asocian a estas comunidades (Zimmerman et al. 1984, Zimmerman y Minello, 1984 y Benfield et al. 1996), sin embargo Zimmerman y Minello, 1984, señalan que *L. setiferus* puede habitar indistintamente en regiones con o sin vegetación sumergida. Por otra parte, el patrón de circulación de la laguna induce tanto la variación de los factores fisico-químicos del cuerpo de agua, como la distribución de sedimentos e indirectamente de organismos.

Densidad

Los valores de densidad encontrados en el presente estudio; 0.22 y 0.25 ind/m^2 en junio y julio y 0.19 ind/m^2 en noviembre, son similares a los reportados por Gracia (1989) y (Gracia y Soto (1986a y b), para la época de lluvias, en la región cercana a la boca del Carmen (0.3 ind/m^2 en agosto) sin embargo son bajos a los reportados en noviembre (1.1 ind/m^2). Los mismos autores señalan también valores de densidades de 0.12 y 0.62 ind/m^2 en la misma temporada pero en la laguna de Chacahito en el sur de la Laguna de Términos. Smith (1984) reporta valores de densidad de (0.03 a 0.05 ind/m^2) para la región de Atasta. Las regiones señaladas se localizan muy cerca a las encontradas en este estudio. Ramos Miranda *et al.*, 1999, señala en un estudio comparativo de la comunidad nectónica de la laguna entre los años 1980s y 1990s; cambios en la abundancia y distribución de algunas especies de peces; evidentemente los cambios ambientales naturales y antropogénicos (agricultura de tierras bajas, pesca y explotación petrolera) que han impactado la región afectan directa o indirectamente la persistencia de las comunidades en el ecosistema por lo que sería adecuado continuar con estudios específicos sobre estos tópicos que integralmente permitan definir la magnitud de estos cambios.

AGRADECIMIENTOS

Los autores desean expresar su agradecimiento a CONACYT-SISIERRA, El Area Protegida de la Región de la Laguna de Términos, Campeche y la Sociedad de Pescadores del Golfo de Cd. del Carmen por su apoyo económico para el proyecto cuya parte de los resultados son vertidos en el presente trabajo. A CONACYT, México, las facilidades otorgadas, Al Centro EPOMEX, de la Universidad Autónoma de Campeche, su apoyo a lo largo del proyecto. Al Instituto Europeo del Mar (IUEM), Universidad de Bretaña Occidental, Francia su ayuda en el aspecto metodológico del presente trabajo

LITERATURA CITADA

Fisher, W. 1978. Consultation on shrimp stocks in the Caribbean and adjacent regions. FAO, Fisheries Reports, FIRD/R124. Vol. VI.

García, S. y L. Le Reste. 1981. Life cycles, dynamics, exploitation and management of coastal penaeid shrimp stocks. *FAO Fish. Tech. Pap.*, Vol. 203: 215 pp.

Gracia, G.A. y E. Soto. 1986a. Estimación del tamaño de la población, crecimiento y mortalidad de los juveniles de *Penaeus setiferus* (Linnaeus. 1767) mediante marcado-recaptura en la laguna de Chacahito, Campeche. México. *An. Inst. Cienc. Del Mar y Limnol. Univ. Nal. Autón. México* 13(3):217-230.

Gracia, G.A y E. Soto. 1986b. Condiciones de reclutamiento de las poblaciones de camarones peneidos en un sistema lagunar-marino tropical: Laguna de Términos-Banco de Campeche. Paginas 257-265 en: A. Yáñez-Arancibia y D. Pauly (eds.) IOC/FAO Workshop on Recruitment in Tropical Coastal Demersal Communities. *IOC Workshop Report* No. 44.

Gracia, A. 1989a. Ecología y pesquería del camarón blanco Penaeus setiferus (Linnaeus 1767) en la Laguna de Términos-Sonda de Campeche, *Tesis Doctoral, Fac. de Coencias, Univ. Nal. Autón. México.* 127 pp.

Gracia, A. 1989b. Relationships between environmental factors and white shrimp abundance in the southern Gulf of México, *An. Inst. Cienc. Del Mar y Limnol. Univ. Nal. Autón. México* 16(1):171-182.

Gracia, A., A. Vázquez-Bader, F. Arreguín-Sánchez, L. Schultz-Ruíz, y Sánchez, J. 1997. Ecología de camarones peneidos, Paginas 107-124 en. D. Flores-Hernández, P. Sánchez-Gil, J. C. Seijo y F. Arreguín-Sánchez (eds.) *Análisis y Diagnóstico de los recursos Pesqueros Críticos del*

Golfo de México. Universidad Autónoma de Cmapeche. *EPOMEX Serie Científica, 7.*

Pérez-Frafante, I. 1970. Claves ilustradas para la identificación de los camarones comerciales de América Latina. *Secretaría y Comercio, Investigación biológica pesquera, México*. 48 pp.

Pérez-Farfante I. and B. Kensley. 1997. Penaeoid and Sergestoid shrimps and prawns of the world. Keys and diagnoses for the families and genera. *Mémoires du Muséum National d'Histoire Naturelle*. Tome 175. Zoologie. 233 pp.

Ramos-Miranda, J., L. Quiniou, D. Flores-Hernández, L. Ayala-Pérez, y A. Sosa-López. 1999. Evaluación de las comunidades nectónicas de la Laguna de Términos, Sur del Gulfo de México: Comparcion de Comunidades a 12 años de los últimos estudios. Paginas 592-593 en. Tresierra-Aguilar A.E. y Culquichicon Malpica Z.G. (eds).Res. VIII COLACMAR, Tomo II.

Rivera-Arriaga, E., A.L. Lara-Dominguez, J. Ramos-Miranda, P. Sánchez-Gil and Yañez-Arancibia. 1996. Ecology and population dynamics of Lutjanus synagris on ampeche Bank (Ecologia y dinámica poblacional de Lutjanus synagris en la Sonda de Campeche). Pages 11-18 in F. Arreguín-Sánchez, J.L.Munro, M.C. Balgos and D. Pauly (eds.) *Biology Fisheries and Culture of Tropical Groupers and Snappers.* ICLAM, Conf. Proc. 48. 449 pp.

Rothlisberg, P. C., B.J. Hill, and D.J. Staples. (1985*) Second Australian National Prawn Seminar *CSIRO Div. Of Fish Res., Cleveland, Australis*. 368 pp.

Sánchez-Gil, P., D. Flores-Hernández, J. Ramos-Miranda, F. Arreguín-Sánchez, y J. Sánchez-Chávez. 1994. Diagnóstico integrado de las pesquerías del Estado de Campeche. *Informe final de investigación. Proyecto (910104001336), Convenio C910400104 de la Secretaría de Educación Pública/DGICSA-UAC, Programa de Ecología, Pesquerías y Oceanografía del Golfo de México (EPOMEX)*, 54 p., III anexos.

Staples, D. J. and D.J. Vance. 1986. Short-term and long-term influences on the immigration of postlarval banana prawns *Penaeus merguiensis*, into a mangrove estuary of the Gulf of Carpentaria, Australia. *Mar. Ecol. Prog. Ser.* **23**:15-29.

SEMARNAP. 1997. Anuario estadístico de pesca 1997. *Secretaría del Medio Ambiente Recursos Naturales* y *Pesca*. 241 pp.

Signoret, M. 1974 Abundancia, tamaño y distribución de camarones (crustacea, Penaeidae) de la Laguna de Términos, Campeche y su relación con algunos factores hidrológicos. *An Inst. Biol. Univ. Nal. Autón. México 45, Ser. Zoología* 1:119-140.

Smith, K.M. 1984. *Some Ecological Determinants of the growth and survival of juvenile Penaeid shrimp Penaeus setiferus (Linnaeus) in Terminos Lagoon, Campeche, México, with special attention to the role of population density.* Ph.D. Dissertaion, University of California, Berkeley.

Vagas-Maldondo, Y., A. Yañez-Arancibia, y F. Amezcua Linares. 1981. Ecologia y estructu ra de la comunidades de peces en áreas de Rizophora mangle y Thalasia testudinum de la Isla del Carmen Laguna de Términos Sur del Golfo de México. An Inst. Cienc del Mar y Limnol. Univ. Nal. Auton. México **8**(1):241-266

Yáñez-Arancibia A. and J.W. Day, Jr. 1982. Ecological characterization of Terminos Lagoon, a tropical lagoon-estuarine system in the Southern Gulf of México. *Oceanologica Acta. No SP. Coastal lagoons Estuary System ecology.* Pages 413-440.

Yáñez-Arancibia, A., Lara-Domínguez, A.L., Chavance, P. y Flores-Hernández, D. 1988. Comportamiento ambiental de la Laguna de Términos, Cap. 2. Paginas 27-40 en: Yáñez-Arancibia, A. y Day. J. W. Jr. (eds.). *Eología de los Ecosistemas Costeros en el Sur del Golfo de México: La Región de la Laguna de Términos.* Inst. Cienc. Del Mar y Limnol. UNAM, Coast. Ecol. Inst. LSU. Edi torial Universitaria, México, D. F.

Zein-Eldin Z.P. and M.L. Renaud. 1986. Inshore environmental effects on Brown shrimp, *Penaeus aztecus* and *P. setiferus*, populations in coastal waters, particulary of Texas. *Marine Fisheries Review.* **48**(3):9-19.

Zimmerman, R.J., and T.J. Minello. [1984a] Fishery Habitat requerimients: utilization of nursery habitats by juvenile penaeid shrimp in a Gulf of México salt marsh. Unpubl. MS.

Zimmerman, R.J., and T.J. Minello. 1984b. Densities of *Penaeus aztecus*, *P. setiferus*, and other natant macrofauna in a Texas salt marsh. *Estuaries* 7:421-433

Logical Framework Analysis as a Tool for Management of a Tropical Fishery

PETER A. MURRAY [1], KEITH E. NICHOLS [1], VASANTHA M. CHASE [1]
and ROLAND HODGE [2]
[1] OECS Natural Resources Management Unit
Morne Fortuné
P.O. Box 1383, Castries
St. Lucia, West Indies
[2] Department of Fisheries and Marine Resources
Office of the Chief Minister
The Valley
Anguilla, British West Indies

ABSTRACT

With the current state of exploitation, and economic importance, of the marine fishery resources of the eastern Caribbean, it may be timely to consider the development of a new approach to management of fishery resources. It has been suggested that there should be a reorientation of fisheries science toward the management of small stocks such as those prevalent in developing countries, most of which are tropical. The perception that management, and management planning, cannot begin until the stocks have been assessed is a view which has proven detrimental to fishery management in developing countries. OECS States have followed a management sequence of: preliminary fishery assessment; policy establishment; management strategy formulation; fishery assessment, including stock assessment; and finally implementation. This sequence focuses more on individual stocks or species groups than on the fishery as a whole. In this paper we consider the utilisation of the Logical Framework Analysis approach for management of the shallow shelf and reef fishery of the island of Anguilla in the British West Indies. Based on the stated goals and objectives of fishery management in that territory, we derive a Logical Framework model; and suggest that this approach could provide for the comprehensive and holistic view of the industry that is required by Small Island Developing States like those of the Eastern Caribbean region.

KEYWORDS: Anguilla, fishery management strategy, Logical Framework Analysis

INTRODUCTION

Fishery management measures are normally based to some extent on statistics derived from fish catches, giving in most instances an incomplete picture of the state of targeted stocks. What is not often featured in the decision-making process is the impact of anthropogenic activities on the ecosystems which these populations inhabit. The socio-economic patterns of resource allocation and distribution relative to access of the fishery resources, marketing or local importance of the fishery are also not given much consideration in the decision making process. Thus, current systems of management do not allow for timely interventions or in fact cater to the externalities created by resource use in other sectors. The recent experiences of developed countries in managing fisheries based strictly on mathematical models, have shown the limitations of this approach. It has also been suggested, elsewhere (Murray and Nichols, in press), that the so-called "ecosystem" approaches also have their limitations in that they consider the fish as part of a multi-species ecosystem without including the impact of the top predator: man! The link between the strictly mathematical, or "analytical methods of stock assessment" approach, and the ecosystem approaches is in fact *Homo sapiens*, and his impact on the habitats which the populations of marine fishery resources inhabit. It is now timely to consider the development of a new approach to management of these resources, in a manner which considers all of man's interactions with the ecosystem: in other words, the whole fishery. This must be seen in the context of the current state of exploitation, and economic importance, of the marine fishery resources of the eastern Caribbean. This is more so with increased cognisance being given to the linkages between fisheries and other sectors/industries such as tourism. Palfreman and Insull (1994) have suggested a number of issues impacting on the development of the fisheries sector including (but not limited to):

i) the fish resources as strategic assets
ii) sustainable development
iii) linkages
iv) conflicts between interest groups
v) monitoring and enforcement of management regulations
vi) the role of human resources in the process of development
vii) industry structure
viii) foreign exchange
ix) marketing
x) hygiene standards, and
xi) credit.

Any new approach should have the potential to address these issues as an integral part of the action plan for fisheries management and development.

In 1994, Member States of the Caribbean Community and Common Market (CARICOM) participating in the joint Canadian International Development Agency (CIDA)/CARICOM-funded CARICOM Fisheries Resources Assessment and Management Program (CFRAMP), recognised the stated purpose of the project to "enhance the basic institutional capacity and information base and to manage and develop fisheries ..." (CFRAMP 1994). It was concluded that the consequence of (the) deficiencies in knowledge concerning the status of the resource, is poor management, leading to possible overexploitation with increased demand and consequent degradation of the resource. It was also determined that "all this pointed out the need for proper assessments of the region's fish stocks ... (with) the major point to be grasped ... (being) that more information was needed on the scalefish resources in order for them to be assessed and managed on a sustainable basis" (*ibid.*: parenthesis and emphasis ours). CFRAMP went on to note that "these assessment activities would provide the raw data the analysis of which would guide the formulation of appropriate management actions for implementation by CARICOM countries".

Recently Mahon (1997) has suggested that there should be a reorientation of fisheries science toward the management of small stocks such as those prevalent in developing countries, most of which are tropical. He goes on to suggest that the perception that management, and management planning, cannot begin until the stocks have been assessed is a misplaced view, which has proven detrimental to fishery management in developing countries (*ibid.*). The CFRAMP conclusions noted above suggest that this perception of the primacy of stock assessment is widely held in the Eastern Caribbean. While distinguishing between fishery and stock assessment, Mahon (1997) goes on to suggest that management cannot begin until the fishery is assessed. Such a fishery assessment should incorporate not just an analysis of the biological status of the stocks, but also the social, cultural, economic, political, habitat and other considerations relevant to the fishery (see also Campbell and George 1994, Campbell and Townsley 1996). The holistic approach of fishery assessment would also incorporate consideration of the externalities created by resource use in other sectors. This management objective driven (MOD: *sensu* Mahon 1997) approach has the two advantages of (a) bringing industry input early in the action sequence, and (b) encouraging thought on the management actions which are feasible, so that the (often limited) financial and human resources can be optimally allocated early in the process (*ibid.*). We concur with Mahon (1997) that in the stock assessment driven (SAD) approach, industry input tends to be sought after implementation of the management strategy. Often by that time, problems in implementation start to be encountered.

It must be admitted that, in recent times, English speaking member countries of the Caribbean Community and Common Market (CARICOM) have begun to use an approach which combines elements of both MOD and SAD management. We suggest that fisheries management in the region has followed the MOD sequence of: preliminary fishery assessment; policy establishment; management strategy formulation; fishery assessment, including stock assessment; and finally implementation; but that the focus in this sequence has been more on individual stocks or species groups rather than on the fishery as a whole. This can be referred to as a Astock/species-related management objective driven (SMOD) approach. This sequence of management events has been followed in OECS Member States, in one way or another (by accident or design), since at least the early 1980s.

It has been stated that while the "balancing act between present-day benefits and future rewards had been at the forefront of fishery management long before it became popularized under the name of 'sustainable development', the focus in the fishery had been on output ... (but) it is time to replace (the) old emphasis on sustainable output with a new emphasis on sustainability of the fishery system as a whole" (Charles 1998a, parentheses ours). Mahon (1997) puts forward the treatise that "fishery management could be approached using one of the structured project development approaches now required by most international funding agencies, for example, Logical Framework Analysis ...". The LogFRAME matrix is considered (Sartorius, 1996) to be "the embodiment of the design decisions and agreements reached by a project team, it is the team process and corresponding exchange of ideas that increases the overall quality, appropriateness, realism and responsiveness of the design ... The underlying core concept of the approach is Management by Objectives". In this paper we consider the utilisation of the Logical Framework Analysis approach (Sartorius 1996, EEAA 1997, European Commission 1993, cited in Mahon 1997) as a management and development tool for the shallow shelf and reef fishery of the island of Anguilla in the British West Indies. Based on the stated goals and objectives of fishery management in that territory, we derive a Logical Framework model for utilisation by the fishery authorities.

THE SHALLOW SHELF AND REEF FISHERY OF ANGUILLA

The target species for the shallow shelf and reef fishery of Anguilla are: Hinds (*Serranidae*); Parrotfishes (*Scaridae*); Squirrelfishes (*Holocentridae*), Grunts (*Pomadasydae*), Surgeonfishes (*Acanthuridae*), Triggerfishes (*Balistidae*), Angelfishes (*Balistidae*), Groupers (*Serranidae*), Butterfish (*Ephinephelus fulvus*), Snapper (*Lutjanidae*), Spotted eagle ray (*Aetobatus narinari)*; and Stingray (*Dasyatis sp*). While lobsters (*Panulirus argus*) are also caught on the shallow shelf and reef areas, the lobster fishery is so important as to be considered in its own right. The fish are caught primarily by wood framed traps, preferred due to their tendency to harbour moss which are said to attract fish. However, metal framed traps are also used, and have greater longevity although they are more expensive to manufacture. Traps are laid at various depths throughout the extensive shelf area. The bank targeted and depth at which the pot is laid will determine the size of the species caught. A boat will typically haul a set of traps twice per week. Each vessel hauling on average 40 - 50 traps. Handlining is also a common method used for reef fish. Hinds (*Serranidae*), Butterfish

(*Ephinephelus fulvus*), and Snapper (*Lutjanidae*) are mainly targeted using hand lines. Yellowtail snapper (*Ocyurus chrysurus*) are primarily targeted using handline or bamboo rods from land. Beach - seine netting is practiced from Crocus Bay, Meads Bay and Island Harbour. Spear fishing is practiced by a few persons for home consumption.

The management unit of the Anguillan shallow shelf and reef fishery is considered to be the island shelf for juveniles & adults (Anguilla shares an island shelf with St. Martin/St. Maarten and St. Bartholomew: the "Anguilla Bank") and the eastern Caribbean for planktonic larvae. Resource status is largely unknown. However, fishermen complain of having to lay pots further offshore suggesting that there is a decline of stocks on near-shore reefs. Further study is required to determine resource status and to establish a base line for future monitoring. High value species such as Grouper are largely found on offshore reefs. While a Coastal Resource Atlas was prepared for the Department of Fisheries and Marine Resources and published in 1995, a substantial decline in coral cover is described by fishermen in the aftermath of Hurricane Luis in 1995. Monitoring is therefore required to update the current habitat map. Species, area or gear specific catch and effort data are not recorded. There has been no catch and effort data collection post Hurricane Luis.

The main market for reef fish is for consumption within Anguilla and in French St Martin. The majority of reef fish caught by fishers from southern and western ports are sold twice a week in St Martin. Fishers from northern and eastern ports are essentially lobster fishermen who partake in some finfishing. This catch is sold locally on Anguilla, mainly to a company called Fishery and Fishing Supplies. Groupers and Snappers may be sold to hotels and restaurants and are high value species.

FISHERY MANAGEMENT IN ANGUILLA

The overall goal of fishery management in Anguilla is to ensure that the fishery is effectively managed (c.f. also Campbell and George 1994), and that development objectives are identified, priorised and implemented towards obtaining maximum sustainable benefits for the people of Anguilla. This is seen as taking place within a multiple use framework, while promoting intersectoral co-operation and co-ordination. This should ensure that fishery and marine resource management policies and legislation are integrated within the national decision making framework. The stated national fishery management objectives are to:

i) develop a framework for sustainable exploitation of the national marine resource base without compromising the potential for future use;

ii) promote development of responsible marine resource use, specifically the rational exploitation of under-utilised marine resources for the optimum benefit of the people of Anguilla;

iii) meet national nutritional needs by maximising sustainable exploitation of the fishery and through development of marketing and processing systems that ensure the development of equitable benefits for the people of Anguilla; and,

iv) facilitate participatory management, through development of management tools which enable consensus based regulation and monitoring of the multiple systems of use of the national marine resource base.

Current regulations include mesh size regulation. The use of gillnets is banned. The use of dynamite, poisons and other noxious substances is prohibited; and spear fishing is only permitted by local fishermen. The specific management objective for this fishery is summarised in the draft fishery management plan (OECS 1998) as being to ensure sustainable exploitation without any conflict resulting from multiple use of the resources. The Government of Anguilla perceives the primary management measures for this fishery as being 1) effort control; 2) closed areas and / or seasons; 3) size and gear limits; and 4) co-management arrangements

The relevant management issues can be summarised as:

i) lack of stock assessment data;
ii) inaccurate habitat inventory coupled with potential habitat destruction;
iii) no catch and effort data for management;
iv) lack of socio-economic data on the dynamics of the fishery to guide management and development;
v) inadequate market research;
vi) pot theft;
vii) ciguatera poisoning;
viii) lack of a participatory management framework for integrating multiple use; and,
ix) destructive fishing practices.

From the developmental point of view, the Government of Anguilla has the perception that the required measures are: some form of supply and demand control; and the need to improve quality of the product. The issues pertinent to these measures are the fixed price of fish; the absence of adequate market research; the inadequacy of berthing, moorings, and fish storage; and the growing demand by importing countries for adherence to international health and safety regulations.

LOGICAL FRAMEWORK ANALYSIS AS A FISHERY MANAGEMENT TOOL

The need for a fishery assessment (*sensu* Mahon 1997), as opposed to stock assessment, is predicated on the necessity to have a comprehensive and holistic view of the industry. Such a view in turn will better inform a development policy and management plan for the fisheries industry. To date very little effort has gone into understanding the fishery; into conflicts among users; or into developing policies and management strategies. Consequently, essentially uninformed decisions have been made regarding the level of investment in the industry.

It has been suggested (Mahon 1997) that "the greatest hope of managing the myriad of low total-revenue stocks that sustain the majority of fishers worldwide lies in a greater emphasis on planning and process in fishery management problem solving and decision making where financial and human resources are severely limited". Additionally the requirements of recent international agreements in fisheries emphasise responsible fishing and the adoption of a precautionary approach. This creates an urgent need in small states in general, and the OECS sub-region in particular, to re-conceptualise the management and development of fisheries. The conceptual changes must also lead to the inclusion of mechanisms that incorporate greater participation of all stakeholders in the industry, and that will promote better organisation and thus more sustainable development.

Underscoring the urgent need for a new initiative and approach to fisheries, we propose that the conceptual change required to meet the real needs of the fisheries industry can be approached by modeling the management and development of the industry within a logical framework matrix (EEAA, 1997). Such an analytical tool will assist in developing a management strategy and action plan which is clearly thought out, and which will include only those activities which will achieve the stated objectives (c.f. Campbell and George, 1994). In the current context, the Logical Framework Matrix can be used as the basis of the action plan for the management and development of the Anguillan shallow shelf and reef fishery, over the two-year lifespan of the draft fishery management plan (OECS, 1998).

Ideally, the Logical Framework Approach begins with a situational/stakeholder analysis in which a problem tree is developed, identifying the core problem affecting the fishery. Possible solutions are then determined for dealing with this and other perceived problems. In the next step, the actual development of the Logical Framework Matrix with the stakeholders effects a multi-disciplinary, participatory, management approach. This permits an analysis of the fishery which provides a clearer picture of the current situation and the management and development objectives. As a consequence, one of the primary functions of the MOD (*sensu* Mahon 1997) approach is achieved. Another advantage of the Logical Framework Approach is that it can help point to and account for inter-and intra- sectoral linkages which impact on management decision making. Additionally the approach allows for flexibility in the planning process, providing a foundation on which clear actions can be implemented to meet the management and development objectives.

Table 1 shows the proposed Logical Framework Matrix (LFM) for the management and development of Anguilla's shallow shelf and reef fishery. The Logical Framework matrix incorporates some actions to deal with the broad issues suggested by Palfreman and Insull (1994) and mentioned in the introduction to this contribution. These matrices can include externalities created by resource use in other sectors as critical assumptions or risks.

We recognise that within the context of the broader regional or even national situation, a separate logical framework matrix will have to be developed for each individual fishery. This may appear a tedious process, but in the final analysis, the Logical Framework approach allows for "the design of fishery management that is robust, adaptive and precautionary, and ... the development of appropriate institutions to implement such a management approach ... (and thus) may be exactly what is needed to move us in the right direction, towards sustainable fisheries" (Charles 1998b: parenthesis ours). We are of the view that this approach will provide for the comprehensive and holistic view of the industry that is required by Small Island Developing States like those of the Eastern Caribbean region.

ACKNOWLEDGMENTS

The authors would like to thank Ms. Sophie van der Meeren of the OECS Natural Resources Management Unit, Mr. Robert J. Campbell of Integrated Marine Management Ltd., U.K., and Ms. Patricia I. Murray for their valuable comments on an earlier draft of this contribution.

Table 1. Logical Framework Matrix for the management and development of the Anguillan Shallow Shelf and Reef Fishery

Narrative Summary	Objectively Verifiable Indicators	Means Of Verification	Critical Assumptions
GOAL The provision by the fishery of maximum sustainable benefits for the people of Anguilla is ensured	Contribution to GDP is increased by 5% by 2001 Contribution to the nutritional needs of increased 15% by 2001	National accounting publications	Political will Availability of human and financial resources
PURPOSE A participatory framework for sustainable exploitation of the shallow shelf and reef fishery is developed	Fisheries Advisory committee is established and effectively functional by 2001 Fishery contribution to sectoral outputs increases by 5% by 2001 Fisheries Management Plan operational by 2001	Reports of meetings of Fisheries Advisory Committee Reports of meetings of Executive Council Annual reports of Department of Fisheries and Marine Resources National accounting publications	Stakeholders willingness to co-operate Availability of human and financial resources Mechanisms for monitoring fisheries and fishery related habitats in place

Table 1 continued.

Narrative Summary	Objectively Verifiable Indicators	Means Of Verification	Critical Assumptions
OUTPUTS Sustainable increases in economic rent are achieved	Catch composition increased	Annual reports of Department of Fisheries and Marine Resources	Availability of human and financial resources
	Fishers' profits increased by 5%	National accounting publications	Fisheries registration and Licencing system is developed, utilised and operational
	Capital availability to fishers improved	Licensing and registration database	
Full stakeholder involvement in decision making is provided	Stakeholder participation in resource management enhanced		
ACTIVITIES Fishery management information is collected	A system for collection and analysis of catch, effort, biological and socio-economic data is developed and implemented	National accounting publications	$$$ Money available
			Resource persons
			Data base system
Fishing effort is regulated	An efficient vessel inspection registration and licensing regime is operationalized	Licensing and registration database	Sampling frame

Table 1 continued.

Narrative Summary	Objectively Verifiable Indicators	Means Of Verification	Critical Assumptions
Fishery-related habitat management improved	Habitat information is updated and monitoring system established Fisher involvement in monitoring the adverse effects of other sectors is encouraged Habitat degradation/destruction is discouraged	Updated Coastal Resources Atlas Reports of Development Control Authority Department of Fisheries and Marine Resources reports Reports of Fisheries Advisory Committee	Fishers are willing to participated Government is willing to share decision making power/responsibility
Participation of all stakeholders in the formulation of policy related to fishery management is increased	Formation of fisher organisations encouraged Formation of effective Fisheries Advisory Committee is facilitated Community level consultations to discuss Fisheries Management and Development decisions are held.		

Table 1 continued.

Narrative Summary	Objectively Verifiable Indicators	Means Of Verification	Critical Assumptions
	Collaboration with other sectors in the management of intersectorial impacts of/on the fishery is encouraged		
Factors affecting marketing of fish are improved	Research into all aspects of marketing is supported	Consultants' reports	
	The results of market research incorporated into the management strateyy		
Fishery-related infrastructure is improved	Berthing and mooring facilities for fishing vessels are improved		
	Storage facilities for fish and fish products are improved		
Economic status of fishers is improved	Capital, insurance, social security and other economic instruments for fishermen made available	Central Bank Commercial Banking and National Accounts Statistics Fisher organization reports	

LITERATURE CITED

Campbell, R.J. and K.M. George. 1994. *Sub-sector Guidelines for the Fisheries Sector.* Unpublished draft document prepared for the European Commission. Integrated Marine Management Ltd. U.K.

Campbell, J. and P. Townsley. 1996. *Participatory and Integrated Policy: A Framework for Small-scale Fisheries.* Integrated Marine Management Limited. Southernhay West, Exeter, United Kingdom. 45 pp.

CARICOM Fisheries Resources Assessment and Management Program (CFRAMP). 1994. Large pelagics, reef and deep slope fishes assessment subproject specification workshop, 18-26 January, 1994, Basseterre, St. Kitts/Nevis. Final Report. *CARICOM Fishery Research Document* No. 14, CARICOM Fisheries Resources Assessment and Management Program, St. Vincent and the Grenadines. 160 pp.

Charles, A.T. 1998a. Fisheries in Transition. *Ocean Yearbook* **13**:15-37.

Charles, A.T. 1998b. Living with uncertainty in fisheries: analytical methods, management priorities and the Canadian groundfishery experience. *Fisheries Research* **37**:37-50

Egyptian Environmental Affairs Agency (EEAA). 1997. *Project Cycle Management, guidance and checklists.* Technical Cooperation Office for the Environment, Maadi, Cairo, Egypt. 81 pp.

Mahon, R. 1997. Does fisheries science serve the needs of managers of small stocks in developing countries? *Can. J. Fish. Aquat. Sci.* **54**:2207-2213.

Murray, P.A. and K.E. Nichols. 1998. Management issues in the fisheries of OECS Member States. *Proc. Gulf Carib. Fish. Inst.* **50**:1053-1064.

Organisation of Eastern Caribbean States (OECS). (1998). *Fisheries management plan for Anguilla* (draft). Natural Resources Management Unit, Castries, St. Lucia. 47 pp.

Palfreman A and D. Insull. 1994. Guide to fisheries sector studies. *FAO Fish. Tech. Pap.* 342 pp.

Sartorius, R. 1996. The third generation logical framework approacg: dynamic management for agricultural research projects. *J. Agr. Educ. Ext.* **2**(4):49-62.

Evolution of the Fisheries Fleet in Barbados: Causes and Implications

STEPHEN WILLOUGHBY[1] and ROBERT G. CECIL[2]
[1]Division of Fisheries
Government of Barbados
Bridgetown, Barbados
[2]Dept. of Geography, University of Western Ontario
London, Ontario, N6A5C2 Canada

ABSTRACT

This paper covers a half century of evolution of the Barbados fisheries fleet, from early 1950 to the present, with an emphasis on local efforts and causes. Using oral history, archives, and media files, three major evolutionary stages are identified, along with a few minor deviations. The trigger mechanisms that fostered each stage are discussed in some detail. Where possible, the roles of government and entrepreneurs are identified. The social and economic implications of the changes are reviewed for the various stages.

KEY WORDS: Evolution, causes, implications

INTRODUCTION

The current Bajan fishing fleet is the cumulative expression of elements of fisheries that have been part of the scene for more than 50 years. The fleet has evolved from inshore sail boats, to offshore deep-sea vessels. But, there are still remnants of old boat styles, equipment and methods in use today. So to a certain extent, while parts of the fleet are modern, others, in a sense, are a functioning historical exhibit. Several types of fishing vessels were introduced over the last 50 years. Many were unsuccessful and fell by the way, while others are presently represented in the local fishing fleet. The present Barbadian fishing vessels range from small open, wooden boats, propelled by oars to large offshore, inboard-powered, steel hull longliners with hydraulic steering and outfitted with the latest navigation and communication equipment.

The objective of this paper is to trace the evolution of the Barbadian fishing fleet from the 1950s to the present, focusing on the vessels that were successfully introduced to, and retained by, the fishing fleet over the past fifty years and on the factors that may have been responsible for the success.

THE PRE-INDEPENDENCE FLEET

By the early 1950s the local fishing fleet was using open wooden vessels powered by sails. They were 6 to 8 metres in length, fitted with two sails - a main sail and a jib - and carried ballast of up to one ton of scrap iron. These vessels were cumbersome and unstable. Sail designs, and the shifting of the ballast led to many capsizings and drownings, as many old fishermen remember. The government was alarmed.

During the early 1950s the sailboat fleet was severely damage by a hurricane. This lead to the introduction of a new boat called the Calvert, similar in structure to the sail boat. The keel was different and the ballast was more stable, but this slower boat did not gain favor with the fishermen. Simultaneously, small open boats, propelled either by oars or outboards, were used mainly for inshore demersal fishing. These moses, as they are called, are still in use currently.

Fisheries officials saw motorisation as the way forward. They were looking for speed, stability and maneuverability, and pressed for change. The odd motorized boat had appeared before, but the process began in earnest in the 1950s. The fleet went from five motorized boats in 1951 to 446 in 1959 (Wiles 1959). Sail boats co-existed with them for awhile, but eventually disappeared.

These new motorised vessels, called launches, ranged between 7 and 10 m in length and carried engines of 10 to 15 hp, amidship. The cabins evolved also over time, leading to more shelter and more storage space. The launches, which are still very much in use, never fished beyond the sight of land.

Three events catalysed the transition: devastation of the sail boats by a hurricane; availability of timber from felled trees; and, government, seizing the opportunity, offering loans, up to $2,160.00 to acquire motorized boats.

The media covered the advent of the new boats, but only in factual terms, with little or no commentary. It reported that a new type of boat was been built, with a view to modernizing the fishing industry, and that fishermen were experimenting with the boat (Advocate, January 10 and April 9, 1952). The first cement designs did not catch on (Bair, 1962). But, when wood became the building material, the boats were praised by the Fisheries Officer. He was particularly pleased with the co-operation between the people in the industry and the government (Advocate, August 16 and December 14, 1955). The 1961 - 1965 Development Plan, however, was calling for professional advice to design better boats (Hess 1961). It is noteworthy that the call for better boat designs has continued, on and off, in the media to this day.

By the end of the 1970s the launches were bigger; 8 to 12 metres, and faster, using 45 to 180 hp outboard motors. These faster vessels reduced the travel time to and from the fishing ground, allowed for fishing further afield than sailboats, and landed larger catches.

The matter of fisheries infrastructures became a crucial element in the evolution of the fleet, in the late 1960s and 1970s, when very little attention was paid to handling, marketing and distribution of fish. Also the cold storage facilities at the Barbados Marketing Corporation were not adequate.

By the late 1970s and early 1980s larger and faster day boats, up to 36 feet with 180 hp engines, were well in evidence, and they were fully equipped (Mahon, Oxenford and Hunte 1985). In fact, those same authors noted that between 1978 and 1980, as a result of increasing costs of operating day boats, two long-range ice boats, with 8 to 10 tons capacity, were introduced by entrepreneurs. Therefore, it would appear that cost factors were becoming causal factors in the evolution of the fleet.

The larger catches of the 1970s compared with the 1950s, often gave rise to gluts, in a fish distribution system that was essentially coastal, with little movement inland, and few infrastructures for storing, transporting and handling fish. There was a lot of spoilage, and prices dropped a great deal in such situations. The handling of the fish on board the launches, without ice, also damaged the product. The implementation of ice boats was an idea whose time had come.

ICE BOATS

As the day boat owners wrestled with marketing problems, the evolution of the fleet was ongoing. Out of a desire to land larger catches came the introduction in the mid 1970s of a long-range vessel capable of fishing further afield, staying at sea for several days, and carrying ice to sea. These vessels, called iceboats, were promoted in the 1973-77 Development Plan (Government of Barbados). The first iceboat introduced in the mid 1970s failed due to consumers' resistance to iced fish. But, by 1980 there were three iceboats, between 13 and 15 m in length, registered with the Fisheries Division. The iceboats departed from the traditional design and fishing pattern of the launches. One of the iceboats was made from ferro-cement with the wheelhouse and cabin forward and an icehold aft. The other two iceboats were wooden with a wheelhouse and a cabin aft. One carried a freezer while the other was fitted with an icehold. However, it was the former design that took favour with the local fishermen. These vessels made trips lasting 4 to 14 days, instead of less then onr day as did the launches. They took ice to sea and iced the fish as soon as they were caught.

By 1983 the number of iceboats had grown to 20. Prior to 1983 the iceboats were having problems with crews, and finding markets for their catches. There was a high turnover in crew, as many fishermen were untrained and, therefore, did not have the skills required to operate the iceboats. Fishermen skilled in navigation, seamanship and fish handling were scarce. They were accustomed to leaving early in the morning and returning in the evening the same day. They did not fish beyond the sight of land, nor did they pay much attention to the care of the catch.

Fishermen who were used to being at sea for short periods were expected to stay fishing for up to two weeks, a change that caused a great deal of uncertainty. There is no doubt that the change had sociological impacts on the fishermen, especially on their personal and family relationships. It is astounding that men who fished on day boats were able to transform themselves into offshore sailors, with very little if any formal training, such as seamanship, navigation, safety at sea, fish handling, and the like. Also, it must have meant changing their mind sets considerably. However, no studies were conducted to examine the changes in behaviour. But is known that catches improved.

By 1989, there were 59 ice boats in the fleet (Hunte and Oxenford). They were between 12-18 m in length, with 5 to 12 tons capacity ice holds. Many of the boats were converted day boats, 70% being wooden and 30% fibreglass. The boats employed the same fishing techniques for catching large pelagics and flying fish as the day boats. Those authors said that the change from day boat to ice boat represented the second major structural transition undergone by the Barbados pelagic fleet.

The media explained the nature of the ice boats to the public in 1983. A great deal of emphasis was put on the ice component of the operation, an important consideration in an island where fresh fish was valued above all other. The media lauded them as bringing a new thrust to fishing (Advocate , October 22, 1983). Jones (1984) singled out ice availability, and the reduced costs of operating bigger boats, as major factors in the adoption of larger vessels. Yet, the evolution was not without problems, particularly as day boat owners modified their crafts almost as if they wanted to keep pace with the ice boats. According to FAO (1982), the day boats, in most case,s had engines that were grossly overpowered, causing unnecessarily high fuel consumption. The fishermen overpowered their vessels, because they could return to base quicker, and in time to catch the afternoon market, in order to receive higher prices for their fish. Construction and maintenance of the day boats was becoming expensive and difficult due to lack of timber. So, as the ice boats ascended, the day boats evolved too, but apparently not always in the directions most desired by the experts.

The development of the iceboat was catalysed by several private and public sector initiatives. In the latter case, government officials recognising the economic and social importance of the fishing industry, provided a fisheries complex, in the south of the island, at Oistins. It was equipped to furnish all the necessary modern amenities required to store, process and sell fish.

The opening of the complex ushered in a new era in fisheries development in Barbados. The new facilities afforded the users the opportunity to land a better quality fish, by icing the catch at sea, and to reduce post-harvest spoilage, by handling the fish under hygienic conditions and icing or chilling fish for short term storage. Consequently, a better quality fish product was available to the consumers.

When the complex opened, there was a universal lack of training in all aspects of seafood handling. So, the government moved swiftly to provide all manner of training, for individuals involved in all facets of fisheries operations.

The private sector focused on acquiring iceboats, whether as a newly constructed vessel or a converted launch. Many of the new iceboats were fibreglass vessel made by a local boat building company, that was opened in the early 1980s to capitalise on the high demand for iceboats. However the majority of the first iceboats were wooden.

Within two years of the opening of the complex in 1983 the number of iceboats doubled from 20 to 51, but markets were not keeping pace. Consequently, there was intense competition among iceboats, as they landed supplies that outstripped demands, but their numbers grew. Boat owners either converted their wooden launches to iceboats or purchased new ones.

The emergence of small processing plants, capitalising on the large quantity of cheap fish in the market, was a major factor in the development of the iceboats. The processors filleted and froze the fish. They also introduced vacuum packed flying fish. The processors also developed foreign markets. At that point in time you had a sort of symbiosis, the processors needed large amounts of fish, and the boats could supply them, with the day boats still satisfying the consumers' day to day needs. The processed fish was sold to supermarkets, restaurants and hotels. The product became available all year round, which was a new situation for Barbados, where the availability of fish was always seasonal. Moreover, the fish were already processed, filleted and boned.

In the late 1980s, the future of the iceboats as a viable investment started to look doubtful. Iceboat owners were faced with gluts, a high turnover in crew, consumers' resistance to ice-fish and rising maintenance cost. Also, Oistins was straining under its load. During this period fishermen often took two to three days to get rid of their fish. In many cases the fish spoiled before they could be sold.

One way that the fishermen met the challenges facing them was to create the freezer iceboat. They needed to stay at sea longer. So, the fishermen resorted to installing insulated shells of old domestic freezers in their boats, to store and ice fish at sea. Consequently, they were able to increase the time spent at sea and the sizes of their landings.

As with the development of the launches the owners invested in iceboats despite the uncertainty of markets for fish. The vendors, who purchased significant quantities of fish from launches, were not initially interested in iced fish from the iceboats, as consumers really did not accept the product, and it was not unusual for the quality of the iced fish to be poor, as a result of bad handling practices. It took several years to convince the vendors and the consumers that iced fish, if handled properly, was superior in quality to fish from launches, and to convince the fishermen of the importance of proper handling of fish at sea.

Eventually, demands for filleted fish emerged. New markets for them opened in restaurants, supermarkets, hotels and households. Also, there were noticeable changes in the lifestyles of Barbadians, who were becoming more health conscious, were turning to fish. They preferred a product that was already filleted, easy to handle, store and prepare. Consequently, the demand for the filleted product increased. Pressure was now placed on the vendors to supply filleted products, and they bought more iced fish. People began to eat out more, and restaurants increased their purchases of fish.

Processing plants were built as the ice boat emerged, a factor which encouraged their development. The ice boats were bringing in large catches, much larger than the day boats, and they needed some outlets. They were landing more fish than they could sell. Now the patterns of sales were changing. They could not sell all of their fish on the same afternoon, and the processors were offering different outlets to ice boats. They were buying whole catches from them. The fishermen, boat owners, vendors and processors all benefited from the renewed activities in the fishing industry.

The ice boats were a comparatively expensive addition to the evolutionary line. Yet, other than a thesis by McConney (1987), there are no systematic studies analyzing the reasons why individuals adopted new fishing vessels. McConney studied the owners who operate ice boats. He concluded that ice boat acquisition stemmed from the owners' knowledge of the industry, and their calculations of profitability. It was pure and simple entrepreneurship.

An alternative explanation, on the introduction of two long-range ice boats, was attributed to the increasing costs of operating day boats (Mahon, Oxenford, and Hunte 1985). There are no other studies of the reasons for boat acquisitions and adoption of new vessels.

LONGLINERS

Longliners did not become popular until the early 1990s, although the first longliner entered the fleet in the 1980s. The longliners, like the iceboats, carry an ice hold but differ in being larger, outfitted with longline gear, and in targeting tuna and swordfish. The lengths of the vessels reach as high as 25 m and carry engines as high as 800 hp. The number of longliners in the fishing fleet moved from about 4 in 1990 to 23 in 1997.

The role of longliners was summed up, in a specific vein, by the Sunday Sun: "Longlining is seen as the key to fishing. The first Fisheries Officer tried the technique, but nothing came of it, because the boats and the equipment were not adequate. Fairly large boats were needed. The ice boat "Supertest" was the first. The owner studied at the Fisheries Institute in Trinidad and Tobago. Also, when the American longliners were here in 1981, a couple of the current entrepreneurs were on those American boats, in some capacity or other. Now there are 11 locally-owned longliners at the Bridgetown Fisheries Complex, ranging from 38 to 46 feet. Four of the boats were made locally. The others are made of glass re-enforced plastic fibreglass. The boats cost $400,000.00, while a converted day boat would only cost $40,000.00 (August 27, 1989).

In the late 1980s, the development of longliners was stimulated by a development agency's, institutional strengthening project (Crown Agents 1990). Other factors included the sight effect of American longliners operating from Barbados, and their practice of transhipping their catches through local processors. The opening of the second fisheries complex, in Bridgetown in the late 1980s, was also instrumental in stimulating the development of the local Bajan longline fishery.

Crown Agents personnel conducted experimental longline fishing and provided training for fishers. Some local fishers served as crew on board the American longliners, which enabled them to learn the skills needed for longline fishing. Concurrently, local processors gained valuable experience in the shipping of fish to American markets, while obtaining tremendous information about those markets' demand for tuna and swordfish, at a price that was more attractive than that of local markets. The local processors used their acquired knowledge to access the American markets for their products. So a "foreign experience" element entered the "equation" at this point.

The climate was now right for investment in longliners. There was an outlet in the USA for the local longline catches, at profitable returns. Moerover, the Bridgetown Fisheries Complex at the time provided adequate facilities for off-loading local longliners and handling their catches. However, much of the investment in longline fishery came from outside of the fishing sector, as many fishers did not have the financial resources to acquire longliners. The investors were targeting primarily tuna and swordfish, mainly for the export market, which is still the case today.

The government also assisted in these latter phases of boat development. In the 1980s the government had obtained an international aid project, which looked at longlining. Authorities conducted a number of trials around the island. They also did some training of fishermen, in longlining techniques and the operation of navigation equipment. The Barbados Fisheries complex allowed fishermen to off-load their catch next to the dock, unlike Oistins where boats had to be off-loaded on the jetty, which was a less efficient situation.

OUTBOARD LAUNCHES: A PRACTICAL RETURN TO OLDER

During the early 1990s there was a noticeable proliferation of launches with outboard engines. The register of local fishing vessels showed records of outboard launches in the 1960s but they were never popular. The outboard launches reemerged in the mid 80s, but because of the focus of iceboats their increasing numbers went almost unnoticed, until the early 1990s.

The outboard launches evolved out of changes to the traditional moses and launches. In the case of the moses a cabin was added. Many of these converted moses (outboard launches) shifted from fishing on the reefs to fishing offshore for flying fish and dolphin during the pelagic fishing season.

In the case of original launches the outboard engines replaced the traditional inboard engines. It was usually the smaller launches that were fitted this way. Other outboard launches were deliberately constructed as launches to carry an outboard engine. By 1993 there were 82 outboard launches. It was usually the less fortunate fishers who used outboard engines in their launches.

Also, it is still not uncommon for some moses boats, especially those with higher-end specifications, to engage in activities aimed at landing flying fish. They are quite capable of conducting inshore fisheries operations, especially when their traditional demersal resources are in their off-season phase.

There were two major factors that favoured the proliferation of outboard launches during the late 1980s and early 1990s. During this period, money was difficult to acquire to purchase the expensive inboard engines. In addition, the repair and maintenance costs of inboard engines began to increase as the government started to phase out the free inboard repair service that was offered by the Fisheries Division. Therefore, vessel owners were forced to pay for the service in the private sector. Consequently, many fishermen found themselves faced with un-affordable acquisition and repair costs for inboard engines. Therefore, many persons opted for the less costly outboard engines, even though many experts felt that in the long term the outboard engines would be more expensive.

VIEWS OF THE EXPERTS AS THE EVOLUTION PROCEEDED

Over the years, the structures of all boats, including those now in use, have come under expert scrutiny. None is seen as ideal, in terms of naval architecture, nor is any accorded the distinction of being a particularly efficient vessel. Crown Agents (1990) sounded an ominous note about the day boat fleet, suggesting that it might eventually fail completely. They saw the boats as unprofitable, and not able to cover either loans or depreciation. The day boats are old, and there is no renewal. The implications of such observations are extremely serious. But, they stand, more or less, alone, as they fail to heed or cite other analysts. For example, they might have looked more closely at Hunte and Oxenford (1989), who saw a traditional fleet component existing side by side with a modem one. Hunte and Oxenford draw their recommendations from sound data.

Some concerns have been raised about the ice boats that have been adapted for longlining. As in years past, the government continued its ongoing search for a new design for the boats. One concern was a need to increase the crews' work space (Advocate, January 28, 1990). Some ice boats went through a refurbishing and lengthening process, to allow the fitting of a mini longline reel, with 3 to 5 miles capacity (Jones 1993). But that process was not advocated universally by a consulting firm. The latter saw ice boats as being designed primarily for the capture of flying fish. Crown Agents (1990) saw the ice boats suited for the pursuit of flying fish, but being far from ideal. The craft are unsuited for the open Atlantic.

Another view on this subject was provided by Hunte, McConney, Oxenford and Dharmaratne (1994). They make reference to earlier work by Hunte, in the late 1980s. That earlier study suggested that, for longlining, boats similar in size to the ice boats should be used, but modified to include hydraulic spools and perhaps 10 to 15 miles of longline. The report then informs the reader that five ice boats, between 38 and 42 ft., had been outfitted for long-lining. One local manufacturer was constructing a 52-foot GRP boat for long-lining. There were two 75-foot longliners, but no 100-foot vessels. A final recommendation was that 40 to 50 foot boats were suitable for longlining, and that 100 foot boats were unsuitable. Seventy-five foot boats were considered to be marginal.

In the recent past, the issue of boat design, and hence the evolution of the vessels of the fleet, was still ongoing, more in terms of detail than radical change. The issue appeared in the media a number of times in 1997 (Advocate, June 20, June 22, and June 23). Fishermen are still questing for better boat designs up to this day.

INDIVIDUAL INITIATIVES DURING THE EVOLUTION OF THE FLEET

The systematic literature does not offer any insights into individual initiatives as the fleet evolved. Such coverage is the province of the media. Over the years, the press has criticized boat designs occasionally, and some entries have speculated on the inability of the rank and file fishermen to be able to afford newer and bigger boats. Now and then, the papers cover the arrival of some "unique" boat, with some fanfare. No such boat has survived the sands of time.

The more noteworthy efforts coincide with the advent of the ice boats and the general trend towards bigger and costlier vessels. In 1981, one man suggested that glass-reinforced plastic be used as the basic material for the boats (Advocate, September 10, 1981).

In 1982, there was an article about a man who had a $200,000 boat, said to be the biggest boat in Barbados. The owner had a mini mart, and was a small farmer. He was arguing about fishing, with some fellows and decided to "plug into. His only problem was getting ice (Advocate, November 17, 1982).

Every now and then somebody was trying something new in fishing boat design. In 1983, there was a man interviewed who was launching a multi-purpose boat, which he considered would be lucrative. It would fish all year long, not just in the flying fish season. He had all manner of new and modern equipment on board. (Nation, May 20, 1983). The media also pointed out that some owners were modifying their moses boats into "punts", i.e., putting cabins on them in order to go to sea longer. One man, a janitor, who was in the process of such a conversion, caught the attention of the press (Nation, January 27, 1987). There were a few other occasional entries of a similar nature, with no follow up. None of the boats was ever used as a fleet prototype.

In 1995, the media ran a story on a man in the island who was building a 42-foot fibreglass boat himself. The article went on to say that a firm in Barbados used to build such boats, but had not produced any in recent years. The man doing the building had visited several boat-building places in the United States. He chose his own design, from what he saw, and was now building his own boat. He was working with a local shipwright (Advocate, January 7, 1995).

Individual initiatives seem to have occurred as independent events. Less than a dozen are mentioned over five decades. So, on average, such initiatives have been few and far between. But, they have occurred, and, by all appearances, they have been independent of "standard development forces".

It would seem that individuals would opt for boats that were bigger, and better equipped, than those in the erstwhile contemporary fleet. The newspaper would put the stress on bigger, and the implications were always that, somehow, such boats would perform in some outstanding form. Basically, the reporting amounted to signalling that the boats had arrived, but the articles made no provision for either *a priori* or *a posteriori* analysis.

Each entry, regardless of the date, either said, or suggested, that the boats were the results of entrepreneurship, and that they would create jobs, and generate incomes. Also, such boats could normally sail further, and stay out longer, than the day boats. So, such innovative vessels were attempted to evolve out of the existing types. It would be interesting to measure the extent of the influence of the presence of such vessels on contemporary and/or future entrepreneurs.

None of the individual initiatives are given credit for influencing the evolution of the fleet as a whole. Whether or not they did is left to be determined. At this point, neither the popular press, nor the systematic literature, gives any indication of major influences in boat evolution generated by individual initiatives. But, if nothing else, those initiatives attest to the capital-generating efforts and capacities of individual owners. Each new vessel, increasingly costlier through the years, must have required funds in excess of those normally available to the average day boat owner. Unfortunately, the popular press missed the opportunity to contribute to historical analysis, by overlooking the entrepreneurial aspect of capital formation for such ventures. In retrospect, individual initiatives attest to the independent spirit of the owners. In short these efforts attested to positive individualism, which is something that economic developers should study in great detail.

SOCIO-ECONOMIC IMPLICATIONS OF BOAT EVOLUTION

Some socio-economic implications of boat evolution have been touched on, from time to time, primarily in the popular press. In the early 1950s, it seemed that government loans were a necessary part of the fishing scene. Their availability meant that boat crews could aspire to owning their own boats (Advocate, August 11, 1951).

At that juncture the costs of the new boats was still within reach of the average fishermen, if given financial help. The government did step in and the development process went into operation at the level of existing participants. So, the socioeconomic conditions favoured the industry's active participants. Unfortunately, the owners did not repay their loans very well (Cecil 1988). However, the "small" man was still at the very centre of the economic development of that early period.

When the second major wave of innovation brought the ice boats, there was one major socio-economic implication above all others. In simple terms, the Fisheries Officer saw the ice boats menacing the owner of smaller vessels. He summed up his views in the press as follows: rising fuel costs, and the effects of inflation on boat building, equipment, and other fishing costs, will force many of Barbados' small fishermen out of business in the near future. It costs $40,000 to launch a 35 foot boat, and over $100,000 for larger and better vessels. He also said that, if one had that kind of money, one would probably want to invest it elsewhere (Nation, May 11, 1982).

The sizes of boats received some notice. It was seen that sizes would change, and, in turn, ownership patterns would change, leaving "the little fellow" behind. The articles in the 1980s seem to have vindicated the earlier ones, especially where boat ownership was concerned. By 1984, they were being referred to as "the mighty giants", collectively capable of becoming dominant in the fishing industry. And, when there was too much fish, the small producers would loose out, not the ice boats. One article went as far as to identify a peasant/ice boat dichotomy, explaining that the larger boats would knock out the peasants. The article, purely and simply, implied class warfare (Advocate, January 28, 1984).

It is not yet clear whether or not the ice boats will indeed "knock out" the "peasant" day boats, or whether the latter will be severely curtailed. One thing for certain, is that there are still a lot of day boats in operation. Conversely, ice boats are not guaranteed survival by their mere existence. They too can go out of operation. An 1989 article about the man reputed to have begun the ice boat era, is a case in point: "In 1979, (he) really began the ice boat era. He saw a rosy future in them, and indeed there was. But, it has not been so for the past five years. Flying fish are scarce. They are plentiful in Trinidad and Tobago, but Bajan fishermen can not fish there. However, fish from that country is coming into Barbados. His boat is now up for sale, and his son, a master fisherman, is out of the business"(Sunday Sun, May 28, 1989).

The exit from fisheries of the man who pioneered ice boats was explained in terms of increased resource scarcity, coupled with exclusion from the flying fish stocks that swim the waters of Trinidad and Tobago. So, his losses stem from biology and geopolitics. One wonders if those two factors have harmed other boats (Cecil 1999). Unfortunately, the media's sample of one leaves no grounds for a proper analysis. But, it does put a human face on the first owner of an ice boat.

Of course the actual vagaries of fishing affect the lives of those who practice the art. Boats are lost at sea, e.g., a $100,000 ice boat sinks after hitting a log (Advocate, March 28, 1991). Unfortunately, lives are also lost. In the days of sail boats there was a heavy loss of life at sea, with an average loss of 12 people a year, from the 1930s to the 1950s (Nation, July 12, 1994). The day boats are safer, but not that safe. Lives are lost every year, albeit fewer than during the sail boat era. The question remains as to whether the ice boats and longliners are the safest vessels yet to be used in Barbados. Indications are that such might be the case.

The evolution of boats, especially in times of major change, i.e., first motor boats, then ice boats, seems to trigger considerable socioeconomic impact. Looking strictly at the press, the main effect appears to be in ownership patterns. The "little man" looses out, while entrepreneurs take over ownership. Only when the motor boats wiped out the sail boats was the little man able to compete. But, the ice boats are in a different financial league than the day boats, forming a distinct fleet, separate from the day boat fleet. However, both fleets operate in the same storage and market spaces, and competition is, apparently, fierce.

As early as 1986, a warning was sounded, by Hunte and Oxenford (1989) which effectively points to an "evolutionary boundary" being reached with the advent of longliners. Fifty-foot, or so, longliners seem to be the end of the evolutionary line, in terms of the growth of the size of boats, and of their functions. In their writings, bigger boats are either discouraged or not recommended.

In a country such as Barbados, evolution may never reach the levels of technol-ogical sophistication found in large developed countries. There are limits to growth and modernization. In fact, less than modem vessels, such as day boats, may prove to be highly desirable. Will master plans ever discuss limits, co-operation, competition, and differential development, and plan for each identifiable sector separately?

CONCLUSION

The government was instrumental in one major evolutionary phase, when the fleet adopted engines over sails. Otherwise, as the media points out now and again, government financing did help to launch bigger boats. But, the ice boats and longliners that sail from Bajan shores were all launched by private initiatives. Granted, the government backed many financially, but the boats were designed and adopted privately.

On the whole, the evolution of the fishing fleet is only partial. The fleet still exhibits many elements that represent technology, techniques, and operational organizations that are really quite historical. They co-exist quite harmoniously with the most recent modern inputs found in fisheries the world over. Consequently, the key to approaching an analysis of the Bajan fleet is to recognise the comparatively wide heterogeneity of the fleet, not only in its outward appearance, but in all its technological elements. Moreover, the human elements add their own heterogeneous dimensions to the fleet. Furthermore, no design is ideal.

The evolution of the fishing boats is one of the prime indicators of development, and in a true sense of economic development, as the evolution is concurrent with increased technological sophistication, which brings the sector into increasingly higher levels of financial activities.

There have been several major changes in the nature of Bajan fishing crafts over the last half century. The literature on evolution as such is sparse, but reasons for change have surfaced in some studies and in the popular press. In the latter case, the focus is often on an individual entrepreneur's reasons for launching a boat with innovations that distinguish it from its counterparts in the fleet.

As we near the 21st century, the Bajan fleet is a mixture of vessels that differ in size, sophistication, equipment and technological organisation. The fleet is a veritable continuum of boats ranging from the traditional to the modern. Evolution has gone on for over half a century, but excepting sail boats, elements of all the stages in the evolutionary continuum are still about and in use. This evolution will continue to proceed if only in an incremental way. But, the fleet has reached a plateau for the time being.

LITERATURE CITED

Advocate, Daily newspaper. Advocate Publishing Company, Bridgetown, Barbados.

Bair, A. 1962. *The Barbados Fishing Industry*, Geography Department Publication Number 6, McGill University, Montreal. 57 pp.

Cecil, R.G. 1988. The heterogeneity of southeastern Caribbean inshore fisheries: a problem for industry wide development planning, *Caribbean Studies* **21**(1-2):215-236.

Cecil, R.G. 1999. *Half a century of fisheries in Barbados: a quest for socioeconomic interpretations in the systematic literature and the popular press.* Fisheries Division Report No.l. Fisheries Division, Ministry of Agriculture and Rural Development, Barbados. 165 pp.

Crown Agents for Overseas Governments and Administrations. 1990 *Institutional Strengthening of the Fisheries Division of the Ministry of Agriculture, Food and Fisheries,* Barbados, Final Report, Vol., 1, Technical Co-Operation ATN/SF-2474 BA, St. Nicholas House, Sutton, Surrey. unpaginated.

FAO. 1982. *Report of the Barbados Fisheries Development Project Preparation Mission,* Main Text And Annex I To 10, FAO, unpaginated. Government of Barbados. *Development Plan 1973-77.* unpaginated.

Hess, E. 1961. *Barbados: Fisheries Development Programme, 1961-1965,* Barbados Government Printing Office, Bridgetown, 41 pp.

Hunte, W. and P. McConney. 1983. *Barbados: National Report on Fisheries Statistics,* Multilith Working Paper. 41 pp.

Hunte, W., P. McConney, H. Oxenford, and G. Dharmaratne. 1994. *A Feasibility Study of Longline Fishing in Barbados,* Technical Report for the Barbados Development Bank, Bellairs Research Institute. 191 pp.

Hunte, W., and H. Oxenford. 1989. The Economics of Boat Size in the Barbados Pelagic Fishery. *Proc. Gulf Carib. Fish. Inst.* 37:78-85.

Jones, R.V. 1984. *Some Notes on Fisheries,* Multilith Document for Internal Circulation in Fisheries Division, unpaginated.

Jones, R.V. 1993. *Information Note: Expert Consultation on the Development of Offshore Fishing Craft for the Caribbean Region,* Curacao. 4 pp.

Mabon, R., H. Oxenford, and W. Hunte. 1985. *Development Strategies for Flying Fish Fisheries of the Eastern Caribbean, Proceedings of an IDRC-sponsored Workshop at the University of the West -Indies,* Cave Hill, Barbados, October 22-23, 1985. 148 pp.

McConney, P.A. 1987. *Small-scale Fisheries Planning in Barbados: The Roles of Information Exchange and Participation.* MES Thesis, Dalhousie University, Halifax. 138 pp.

Nation: Daily newspaper, Nation Publishing Company, Bridgetown, Barbados.

Sunday Sun, Nation Publishing Company, Bridgetown, Barbados.

Wiles, D. 1959. Mechanization of the Barbados Fishing Fleet *West Indies Fisheries Bulletin.* 4:1-12.

A Fisherman's Dilemma and Fisheries Management

ANDERSON KINCH
Oistins Fisherfolk Association
The Boatyard, Christ Church
Barbados

ABSTRACT

As a fisher you are most interested in catching plenty fish and making more money each trip. But suddenly realization hits you. You are not catching as much fish as before. You are not making enough money. Why? Fisheries management information is not reaching fishers as it should, and what little they get they do not pay any attention to. This is cause for concern and there is no simple solution. Fishermen and fisheries managers often work in different worlds and do not communicate as they should. This causes mistrust and disrespect for the knowledge of both sides. What is needed is more collaboration between fisheries science, management, and resource users to emphasize sharing information and discoveries. This paper describes some of the problems encountered in this area, and possible solutions for consideration.

KEY WORDS: Barbados, fisherman, management

INTRODUCTION

Barbados is the most easterly of all the east Caribbean states. It is located along the 13° 02'N latitude and 59° 30' W longtitude. The island is 166 square miles with a population of over 260,000. Our major industry is tourism, followed by sugar, manufacturing and the fishing industry. Our national dish is flyingfish and cou-cou.

I started fishing when I was 11 years old, still at school, but going fishing on weekends to accumulate lunch money for the next week. My parents were very poor. My father was a fisherman and my mother a general worker in the Ministry of Communication and Works at the time. So it was not difficult for me to choose fishing as a profession after the death of my mother at a very tender age, for I am one of those single parent children. The only time I ever have regrets is when the elements are against me. And that is when the winds are high, the seas are rough and the fish not biting. Fishing is not only my profession, it is also a hobby to me. That is why I can appreciate people leaving the office work behind and going fishing on weekends for relaxation.

BARBADOS FISHING INDUSTRY

Our fishing industry is relatively a large one, consisting mainly of four groups of fishers. That is the reef fisher, day boat fisher, iceboat fisher and longline fisher.

The reef fisher targets the reef fish by way of fish pots (traps), hand casting, or to a lesser extent, spear fishing. The day boat fisher targets the flyingfish, dolphin (mahi mahi) king fish, (wahoo) tunas and many other large pelagics. The boats are around 20-32 feet. With a crew of two they leave early in the morning, returning late that same day. There are approximately 289 day boats.

The ice boat fisher targets the same species as the day boat fisher, but with the availability of ice can remain at sea much longer (as long as ten to twelve day duration). There are approximately 147 ice boats carrying a crew of three.

The longline fisher targets the yellow fin tuna, blue marlin and black marlin, swordfish, and many other large pelagics. With a crew of four the boat range in sizes of 40 to 85 ft long and stay at sea for a period of ten to thirty days. There are approximately 30 longline vessels.

Having given a background of who we are, where we are and what we do, it is time to highlight the hardship we often face.

THE FISHERMAN'S DILEMMA --- UNSOLVED MYSTERIES

Just recently, our reef fish have been dying at an alarming rate, not only in Barbados, but in the neighbouring islands of Grenada, St. Vincent and the Grenadines and Tobago. It is estimated that about 75 thousand pounds of dead fish were disposed of by the National Conservation Commission in Barbados. That figure has nothing to do with what was swept away by the currents and what remains on the seabed. International assistance was sought and preliminary indications are that the cause in Barbados was a *Streptococcus* bacterium which was found to have killed fish in other parts of the world including the USA, Israel, Canada, Japan, and Australia. Thought not conclusive, I am glad that fishermen are not in any way implicated in the fish kill.

In Barbados a white-spined sea urchin (*Tripneustes ventricosus*) locally called "sea egg" was, and maybe still is, a delicacy. Over the years the sea eggs began to gradually disappear. The Government upon advice imposed a moratorium in an effort to save the sea egg industry. At the same time, the black sea egg (*Diadema antillarum*) though not harvested, disappeared. All kind of theories were advanced for the disappearance of the black sea egg (a pathogenic microbe was found to be the cause), but the fishermen were blamed for the low yields of the white sea egg. Indiscriminate harvesting out of season before the sea eggs where mature enough to reproduce, and breaking sea eggs and throwing them back on the seabed were thought to be contributing factors. Although I will concede that these can be contributory factors, no one has been able to convince me that these were the primary factors. With the recent fish kills taking place all over the world which have scientists baffled, I am more convinced that something similar happened to the white sea egg as to the black.

FISHERS, FISHERIES MANAGERS AND SCIENTISTS

Over the years, our government, through the Fisheries Division, has embarked on programmes to save the reef fish, such as to make it illegal to fish on certain parts of the west coast, much to the displeasure of the fishermen on the west coast. There is a move to make it illegal to use 1.25 inch mesh size wire. Instead the 1.50 inch is preferred, with the use of bio-degradable wire as a trap door to prevent of ghost fishing, so allowing the fish to escape therefore reaching a more mature size. This is promoted through education that responsible fishing today guarantees you brighter tomorrow. Having seen and read about the amount of fish that died in the fish kill, I am wondering where were the reef fish fisherman fishing or are the methods employed not effective. I think, in light of what has transpired, that there should be some sort of moratorium imposed, and fishermen should be given assistance to construct artificial reefs.

We in the Caribbean often talk about a common market (CARICOM). Over the years, all kinds of treaties have been signed. It boggles the mind how all CARICOM states can sign a free trade pact, and two countries cannot work out a fishing agreement. Barbados and Trinidad and Tobago over the years have not been able to sign any major fishing agreement after the initial signing of the first agreement which our fisherman found not favourable. I am yet to see a treaty where our fishermen have benefited.

Sometimes, our fishermen take matters into their own hands, venture into Trinidad and Tobago territorial waters, are caught, arrested, jailed, fined, and their catch confiscated. It is easier for foreign vessels to be re-flagged in Trinidad and Tobago than for us to reach a fishing agreement. Under the free trade treaty, the by-catch from those foreign vessels reaches Barbados as CARICOM originated goods. This is wrong, and there must be ways to deal with circumstances such as these.

Over the years I have read and heard a lot of declining fish stocks, and as usual fishermen has been blamed for overfishing. I think that government, scientists and fishermen alike all have to take their share of the blame. Scientists all over the world invent or develop means for us to enhance our fishing by way of satellites, GPS, depth sounders, sonar, water temperature gauges, direction finders, fish finders, the latest in gear technology and literature. Governments have provided the incentives, the support and sometimes the finance. Yet, without them shouldering any of the blame, we fishermen are accused of overfishing. Try telling this to a Barbadian flyingfish fisherman.

The past two years have been the most successful years in our flyingfish history. I am not here to knock scientists or anyone else. All of us are to blame. Sometimes I wonder if we are not too quick to start finger pointing when we are mystified or do not have concrete evidence.

The point I am making is that so long as we have a problem and we accept that there is a problem, let us work together to correct it and address the cause. Over the years, fishermen and scientists have always been at loggerheads as to where the bluefin tuna feeds. The scientists claim it feeds at the surface, and the fishermen claim at the bottom. According to National Fisherman, the February 1997 edition, the fishermen appear to be right. I think the time has come for fishermen and scientists to work harder and closer than ever before. In that way, they will have more respect for one another assessments, and that way the fishing industry will be the benefactor.

Fishermen work hard and long hours exposing themselves to constant danger without even getting the recognition they deserve. I cannot and will not attempt to speak for fishermen world-wide, but what I can do is talk about fishermen in my country. In Barbados, a fisherman is seen as nobody because of the stigma attached to the profession. In former years, fishermen were not well educated, some never had the opportunity for an education, and some never saw it as paramount. What they were interested in was providing food and shelter for their families. Today things have changed, and in order to be successful you must be educated and competitive, no matter what aspect of life from which you come.

Fishing is not an easy task. Sometimes you wonder what ever made you choose it as a profession. You are constantly away from your family. Sometimes you face the most adverse conditions, rough weather, stormy weather, water too cold, fish not biting, not getting enough money for your fish because of market conditions over which you have no control. Just recently last month, I experienced stormy conditions, caught at sea with a broken down vessel during the passage of hurricane Jose. Believe me, it was not a pleasant experience. In Barbados because of the recent fish kill, all fisherfolk are feeling the economic squeeze. Yet when things pick up, as a responsible fisher you still have to be conservative and think about sustainability. It makes all the common sense in the world but, but it does not make economic sense at this time.

There is a campaign in the United States to try to stop fishers from catching small fish, especially the swordfish. The authorities are asking the public not to purchase fish under a certain size. I know what they are trying to do, and that is to preserve the swordfish, making sure there is no market for the small ones, therefore discouraging fishers from taking them. It makes all the sense in the world to throw that fish back in the water if it is alive and hope that it lives, matures and reproduces. But tell me what good will it do if that fish is dead when it is taken from the gear? Should I throw it back and feed other fish, or should I bring it ashore and sell it at a fraction of the cost and help to feed some poor families. Either way, that fish is lost forever. I urge decision-makers not to only think about the developed countries but also the lesser developed ones. Remember that if it is bad in the developed countries, it is worse in the lesser developed, and what might have minor implications may have adverse consequences in the lesser developed. The gap between the affluent and the poor has not lessened. I think we should look at alternatives.

SOME SOLUTIONS

The only way we can overcome these problems is through ongoing education programmes. Having gone to Belize and participated in the First CARICOM Fisherfolk Strengthening Workshop for regional fishermen from 12-23 July, 1998, on reflection I can honestly say it has changed the way I think and fish. I know now that there is more to fishing than financial gain. I urge decision-makers to continue to support all ongoing education programmes so that fishers worldwide will realise that responsible fishing today guarantees a brighter tomorrow.

Organising Fisherfolk in Barbados Without Completing a Clean Round

PATRICK A. McCONNEY
Fisheries Division
Ministry of Agriculture and Rural Development
Barbados

ABSTRACT

In equestrian terms a clean round describes the completion of a prescribed course of jumps without time faults or jumping faults. In Barbados a prescribed course was set for the organisation of fisherfolk into formal groups such as associations or co-operatives. This was through a Fisherfolk Organisation Development Project that has been reported upon at previous meetings of the Gulf and Caribbean Fisheries Institute. The course was prescribed and bounded in time by the project work plan. From inception, most of the obstacles were spotted in the distance and prepared for, but some proved deceptive and difficult to overcome. Likewise the timing was calculated, but with each obstacle the course proved longer. We did not complete a clean round. This paper reviews the Fisherfolk Organisation Development Project that was implemented between June 1997 and July 1999 in Barbados. Features of the project and its results are analysed in relation to a framework of factors derived from the literature on co-management and institutional analysis. Some alternative approaches and future options for extending the work of the project into a long-term programme are explored.

KEY WORDS: Barbados, co-management, fisheries

INTRODUCTION

At the 51st Annual Meeting of the Gulf and Caribbean Fisheries Institute in 1998, a paper was presented on the history of fisherfolk organisation in Barbados and the new initiative of the 1997 to 1999 Fisherfolk Organisation Development Project (FODP) (McConney Atapattu and Leslie 2000). Based upon studies of social processes in the fishing industry an incremental introduction of co-management was advocated (McConney 1997). The FODP was expected to contribute towards the feasibility of introducing co-management through the establishment of formal fisherfolk organisations that could play meaningful roles as fisheries management partners with government (McConney and Mahon 1998). The government was expected to undertake whatever other action was necessary to provide a climate conducive to co-management.

This paper evaluates the FODP and supporting government action in the context of certain key conditions described in the literature as important to the introduction and sustainability of fisheries co-management. Some condition proved to be obstacles too great to be overcome, at least in the period available for execution of the project, while others were taken in stride.

In many respects the course of the project resembled equestrian competition where horse and rider, working as a team, negotiate challenging courses with obstacles of different types and difficulty within a prescribed time. A "clean round" occurs when the process of overcoming the obstacles does not result in award of penalties. But the course can be completed, and the event won, without completing a clean round. In such cases success is judged by a level of performance that is short of perfection, but still satisfactory. Without suggesting who is horse and who is rider, perhaps the roles change with the obstacle encountered, the fishing industry and government similarly work in close partnership to overcome the obstacle course of establishing co-management.

The next section sets out the framework used for analysis. Results and discussion follow. Noting that the Fisheries Division was satisfied enough with progress during the FODP to incorporate continuity of the project into its regular work programme, some thoughts are shared on where to go from here to tackle the remaining obstacles in another round.

FRAMEWORK FOR ANALYSIS

The Fisheries Management Plan for Barbados clearly states that the term "fisheries management" encompasses both fisheries conservation and development (Figure 1).

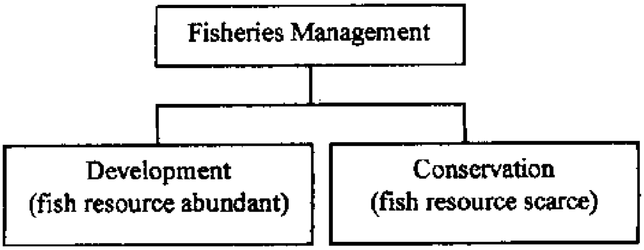


Figure 1. Dichotomy of fisheries management

We are aware that others may use management to mean conservation and to oppose development. This use is not politically correct in a developing country. But it also does not mean that development is seen only as increased fish harvest. This would be equally naïve. Fishery development has many dimensions, some of which are illustrated in Figure 2. Often the non-harvest aspects are of most interest to the fishing industry since catching more fish does not necessarily result in a better life. These are important points for appreciating how co-management is usually viewed and approached in Barbados by both the fishing industry and government.

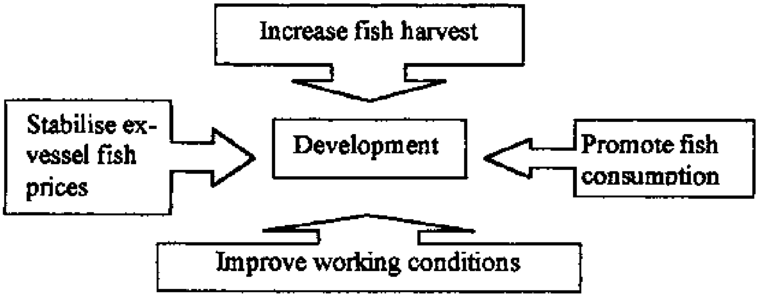


Figure 2. Some dimensions of fisheries development

The framework for analysis in this paper is drawn primarily from work by Pomeroy, Katon and Harkes (1998) on key conditions and principles that facilitate the successful implementation of fisheries co-management. These they drew from Asian experiences with co-management. The twenty-eight conditions and principles suggested for successful fisheries co-management in Asia are given in Table 1.

Table 1. Evaluation of fisheries co-management conditions in Barbados

Co-management condition	Score	Co-management condition	Score
Individual incentive structure	2	Political and social stability	1
Recognised resource management problems	2	Networking and advocacy	2
Leadership	2	Enabling policies and legislation	2
Stakeholder involvement	1	Provision of financial resources/budget	1
Empowerment	2	Government agency support	2
Trust between partners	2	Social and cultural fit	2
Property rights over resource	3	Partner sense of ownership	2
Local political support	2	Effective enforcement	2
Capability building	2	Partnerships and contractual agreements	1
Organisations	1	Overlap of interests	2
Conflict management	3	Flexibility	1
External agents	1	Appropriate scale	2
Clear objectives form a well-defined set of issues	2	Co-ordinating body	1
Effective communication	2	Social preparation and value formation	2

Scores: 1 = fully present; 2 = partially present; 3 = not present

RESULTS

Table 1 also contains the results of analysis derived from qualitatively evaluating how close the FODP and government of Barbados have come to making these part of the fisheries environment in Barbados. A score of 1, 2 or 3 points is given to each attribute depending upon whether it is judged to be: (1) fully present; (2) partially present; or (3) not present.

Individual Incentive Structure

This is partially present amongst most stakeholders, but needs to be developed on more a sustainable basis than the present tendency, particularly amongst fishers, to see co-management as a response to crisis.

Recognised Resource Management Problems

This is also partially developed, but alarming proportions of fishers continue to deny that overfishing, particularly of pelagic species, is possible. More are ready to accept that inshore reef resources are declining, but they see threats from other coastal uses as the major problems, not fishing.

Leadership

Good leadership is present in only a few organisations. Development of leadership skills is an ongoing process that will take additional time.

Stakeholder Involvement

Other coastal users such as dive operators and recreational fishers have been the main stakeholders outside of the fishing industry. Their involvement is full, with an increasing appreciation of the issues related to coastal and fisheries management.

Empowerment

The fishing industry, and particularly the harvest sector, still considers itself relatively powerless in comparison to other coastal and ocean users. A process of enlightenment and empowerment to raise the consciousness and confidence of fisherfolk continues.

Trust Between Partners

The fishing industry generally continues not to trust the government fully, but trust appears to be growing. The government has demonstrated increasing trust in the fisherfolk organisations.

Property Rights over Resources

There is no system of traditional or modern property rights in the fishing industry. Such a system could only apply to the inshore reef fishery in the absence of multilateral arrangements for the pelagic fishery.

Local Political Support

Given the small size of the island and absence of local government, there is little distinction between local and national political support. Political support is present, but needs to be improved upon.

Capability Building

Building capability and capacity was one of the main components of the FODP, but still there is need for much more. This is especially with respect to leadership and the effective management of the fisherfolk associations.

Organisations

Organisation formation and strengthening was the focus of the FODP. In the two year period fourteen fisherfolk organisations became active. Their legitimacy is recognised by government through administrative registration, but to date legal status has only been accorded to co-operatives.

Conflict Management

There are no established mechanisms for conflict management or resolution in the fishing industry. Conflicts are usually mild, and most often between the harvest and postharvest sectors or within the latter.

External agents

The Fisheries Division has employed external agents throughout the FODP period and beyond. Fisheries authorities are acutely aware that a "foreigner factor" exists whereby information or advice from an outsider may be more readily accepted than from a local. This is a serious weakness.

Clear Objectives Form a Well-defined Set of Issues

Issues and objectives are not entirely clear to all parties. In particular, the denial of overfishing in some quarters makes it difficult for fishery conservation and recovery objectives to be shared by all stakeholders.

Effective Communication

Communication is improving between the fishing industry and government, but is not altogether satisfactory. One reason is that many in the fishing industry are not familiar with concepts in fisheries management.

Political and social stability

Barbados is extremely stable socially and politically.

Networking and Advocacy

Networking, more than advocacy, was a central theme of the FODP experience. Emphasis was placed on community linkages and exchanges of ideas and information between fisherfolk and other stakeholders.

Enabling Policies and Legislation

The inadequacy of the legal framework was hinted at above concerning organisation registration, but it also extends to jurisdiction and powers. Right now, fisherfolk organisations are recognised entirely due to a supportive government policy which needs to be reflected in legislation.

Provision of Financial Resources/budget

The budget of the Fisheries Division is small, but since 1998 funds have been allocated to provide direct assistance through a small grant facility administered by the Fisheries Division with a minimum of red tape. Funds are available to fisherfolk organisations from a variety of sources, but they need to demonstrate willingness to manage their own affairs properly.

Government Agency Support

As stated throughout, government support has been excellent and the driving force behind co-management. Some types of support, particularly those requiring legal bases have, however, been too late in coming.

Social and Cultural Fit with Existing and Traditional Institutions

Although there were fisheries co-operatives 30 years ago, and the harvest sector in particular has yearned for more unity amongst fisherfolk, collective action is not the norm in the fishing industry. However in a country with an increasing number of interest groups, the social and cultural fit is much better now than before.

Partner Sense of Ownership of the Co-management Process

The sense of ownership in the fishing industry is only partially developed. There is still a considerable degree of dependence on government in paternal rather than partnership relations.

Effective Enforcement

There are relatively few fisheries management laws, and what few exist are only poorly enforced or complied with. However, there has been a noticeable increase in the provision of information from the fishing industry about law-breaking activities which offers hope for future development.

Partnerships and Contractual Agreements

Such agreements are not yet in place, although informally there are close links between the Fisheries Division and the more active fisherfolk organisations. These links are aimed at improving the ability of the groups to be sub-contracted on fisheries issues. This relationship has developed in the awareness that as non-governmental organisations (NGOs) the fisherfolk organisations stand a better chance than government in attracting some types of funding and resources for conducting participatory activities.

Overlap of Interests

There is a significant overlap of interests amongst coastal and ocean resource users. This is now being realised through dialogue between the groups which did not exist before the FODP.

Flexibility

Notions of flexibility and adaptation are fundamental to the design of the co-management process in Barbados due to the acknowledged high levels of uncertainty in many aspects of the fishing industry.

Appropriate Scale

Scale poses a problem. Because nearly 90% of the total fish harvest in Barbados consists of migratory pelagic fishes, a region-wide scale for co-management is most appropriate. This is presently not possible due to the absence of appropriate inter-governmental organisations or arrangements. A smaller scale is appropriate for the inshore resources, but the absence of discrete geographic fishing communities and fishing grounds, results in large sections of coast, virtually the entire island, having to be the shared management area for several groups. Again the scale is too large in relation to the present capacity of the fisherfolk organisations. Hence the focus on the operation of a viable secondary level, national fisherfolk body.

Co-ordinating Body

The secondary level, national fisherfolk body co-ordinates fisherfolk organisations, while the Fisheries Division and Coastal Zone Management Unit bring together many government interests. The Fisheries Advisory Committee (FAC) which operates under the Fisheries Act serves as the ultimate co-ordinating body by including major stakeholders at policy level.

Social Preparation and Value Formation

There was insufficient time under the FODP for social preparation, and it is still an ongoing process fundamental to the sustainability of the co-management process. A significant difference between wanting co-management and functioning within its framework has been revealed by experience. Society still sees the fishing industry as individualistic and not capable of collaborative initiatives in true partnership with government. This is slowly changing, and successful practical demonstration of the potential of co-management has been the most effective change agent.

DISCUSSION

As shown in Table 1, the majority of key conditions and principles identified as important in the Asian experience are present to some extent in the Barbados situation. Because co-management, and indeed the whole concept of fisheries management, is a very recent introduction to Barbados, it is not possible to tell whether the same level of importance will be attached to these enabling conditions and principles in the longer term. However, it seems likely from this very preliminary and superficial analysis that several of them may be critical.

The results also reveal some policy implications that point the way forward if Barbados is to proceed with co-management as far as is feasible. One of the most fundamental concerns is scale. The sharing of fishery resources with neighbouring islands makes national scale, community-based co-management virtually impossible for many fisheries. It is enticing to think that regional scale co-management will be feasible once the appropriate networks of public and private sector stakeholder can be brought together, but this is futuristic. What is more feasible now is the forging of partnerships aimed at establishing the environment for such arrangements. In this context the current setting up of a regional inter-governmental fisheries mechanism is of considerable interest, and it needs to be paralleled at the community level among fisherfolk organisations.

Other major considerations concern the rate of progress of establishing co-management in Barbados, particularly in the wider context of including development aspects. While the process cannot be rushed, it is clear that the government must provide a more favourable climate for co-management through policies, procedures and legislation. There has been much good progress towards meaningful partnership between the fishing industry and government based on good will and trust, but this is insufficient. Unless the rights and duties of the stakeholders are more securely established, with flexibility of course, there will always be some holding back in fear of trust betrayed. Government needs to act swiftly to improve the climate and demonstrate its commitment in a more tangible manner.

Finally, there is the perspective and capacity of the fishing industry stakeholders, and particularly the fisherfolk organisations. The latter need to recognise and begin to exercise their true potential. At this stage there is understandably much dependence upon government, but a greater degree of selfreliance must be demonstrated if they are to avoid co-optation by government, even if unintentional. Access to resources and expertise through NGO networking must be more fully explored. The organisations are still very weak in terms of their administrative capacity and ability to conduct their own affairs. Without significant advancement in this area, given the absence of a traditional fishing community organisational structure, their chances of meaningfully participating in co-management are small. As concluded in last year's presentation at the GCFI, their continued strengthening is of vital importance to the sustainability of co-management.

LITERATURE CITED

McConney, P.A. 1997. Social strategies for coping with uncertainty in the Barbados small-scale pelagic fishery. *Proc. Gulf Carib. Fish. Inst.* **49**:99-113.

McConney, P.A and R. Mahon 1998. Introducing fishery management planning to Barbados. *Ocean and Coastal Management* **39**: 189-195.

McConney, P.A., A. Atapattu and D. Leslie. 2000. Organizing fisherfolk in Barbados. *Proc. Gulf Carib. Fish. Inst.* **51**:299-308

Pomeroy, R., B.M. Katon and I. Harkes. 1998. Fisheries co-management: key conditions and principles drawn from Asian experiences. Paper presented at the 7th Conference of the International Association for the Study of Common Property, 10-14 June 1998, Vancouver, Canada. 23 pp. Unpubl. MS.

The Changing Face of Fisheries in the Economies of OECS Member States

JAMES D.K. WILSON[1] and PETER A. MURRAY[2]
[1]P.O. Box 2873
Maputo, Mozambique
[2]OECS Natural Resources Management Unit
Morne Fortuné
P.O. Box 1383, Castries
St. Lucia, West Indies

ABSTRACT

Fisheries continue to play an important and sometimes underrated part in the economies of OECS Member States, providing both full-time, part-time and seasonal employment, contributing significantly both to domestic food security and national GDP. Offshore resources and larger pelagics are felt to have the best potential for increased exploitation and present an identifiable path for technical and economic diversification. At national levels there are indications of fishers becoming both more professional and more committed. The sub-regional fisheries sector is now managing to attract private investment from persons who, with few traditional links with fishing, are bringing a new level of professionalism to it. There has, in recent years, been a significant investment in fisheries infrastructure in the OECS sub-region, contributing to a transformation process in some Member State fisheries. In some areas moves have been made to manage existing resources, but the existing technologies are inadequate to take up the challenge of resource management and the exploitation of new resources. Additionally, national policy statements and objectives for fisheries need to be further developed and clarified to assist the implementation of national fisheries strategies.

KEY WORDS: Economic diversification, fisheries, OECS

FISHERY RESOURCES OF OECS MEMBER STATES

The distribution of fishery resources in the Member States of the Organisation of Eastern Caribbean States (OECS) can be broadly divided up on a geographical basis (Wilson 1999). The northern (Leeward Islands) group have significant shallow water resources (coastal reef demersals and pelagics) and more distantly located offshore pelagic and deep slope resources. Amongst the southern (Windward Islands) group, shallow water resources are less significant. Deeper water pelagic and deep slope resources are relatively near and easily accessible. The inshore resources, particularly shallow water reef resources, shallow water lobster and conch are under some stress. These resources have historically been the easiest and cheapest to exploit due to their proximity and the suitability of traditional fishing methods. In some States there is very little targeting of demersal deep slope (shelf edge) fishery resources. Ciguatera poisoning of fish poses a particular problem in specific areas of the Leeward Islands, making some coral reef resources unexploitable and creating a lack of consumer acceptance for some species. In general, offshore resources and larger pelagics are felt to have the best potential for increased exploitation, based upon some existing resource studies and, particularly in the Windward islands, the apparently high profitability of the fishery (Wilson 1999). Potential for expansion of effort on certain pelagic stocks, however, may be limited by decisions of the International Commission for the Conservation of Atlantic Tunas (ICCAT).

FADs are thought to have performed well in the sub-region, attracting fishers away from coastal resources and making the fishing of some pelagics less energy intensive. In some States artificial reefs continue to serve both the fisheries and the tourist industries. In others, initial results were good, followed by a gradual degradation in performance as the artificial reef disintegrated.

There is very limited production in the sub-region from culture based activities, although studies have indicated that there is some potential for cage grow-out of shallow water demersal species (Wilson 1999). The promotion of cage based culture may conflict with other coastal resource uses, particularly tourism. In St Lucia and St Vincent there is some small-scale aquaculture production of tilapia and freshwater shrimps (George 1999, Ryan 1999). The viability of culture activities is very dependent upon the availability of suitable feed, land and water resources.

FISHERIES IN THE ECONOMY

The importance of fisheries to individual economies shows significant variation across the member nations. Table 1 shows the most recent values of selected indicators relating to the contribution of fisheries to the economies of OECS Member States. Some concern has been expressed by fisheries officers, at various fora, about the correct evaluation of the contribution of fisheries to Gross Domestic Product (GDP), with export valuations based on the ex-vessel rather than Freight on Board (FOB) price, thus overlooking any incremental economic benefit of the external market.

Table 1. Selected indicators of the contribution of fisheries to the economies of OECS Member States (Barrett 1999, Conservation and Fisheries Department 1999, Finlay 1999, George 1999, Guiste 1999, Jeffery 1999, Murray 1999a, O'Garro 1999, Ryan 1999, Simmonds 1999) n/a = not available.

Country	Contribution to GCP (EC$M)	% of Total GDP	Landings Quantity (tonnes)	Employment	
				# of Persons	% Fishermen
Anguilla	4.71	2.6	333	400	n/a
Antigua and Barbuda	21.8	1.8	653	1,200	n/a
British Virgin Islands	10.87	0.9	1,000	387	97
Dominica	8.84	1.6	1,079	2,891	94
Grenada	11.41	1.7	1,260	1,949	90
Montserrat	0.52	0.6	46	60	n/a
St. Kitts and Nevis	8.35	1.4	339	458	88
Saint Lucia	8.76	0.8	1,523	1,957	91
St. Vincent and the Grenadines	13.08	2.0	809	657	n/a

Marketing and Trade

The primary marketing and distribution of fish in the sub-region is characterised by:

i) Direct sales to the consumer;

ii) Sales to many small traders, who sell to the consumer with little further processing; and

iii) Sales to State supported marketing organisations at a guaranteed price. The national markets are, in the main, strong enough to be able to absorb all of the current production. In addition, the position which fish has in the local diet is becoming more significant as countries develop and consumers become more health conscious. If a weakness exists, it is in the internal distribution of fish products, the failure of which has been known to lead to spoilage and dumping (cf. Guiste 1999). Domestic consumption in many of the OECS Member States is focussed upon inshore species, particularly small pelagics and shallow reef demersals.

In some States the tourist market for fish products is strong, consuming a large amount of fish compared to the domestic market (George 1999). The intra-regional trade in fish is product specific, and is often focussed on added value products such as flying fish fillets. In four OECS Member States, the government intervenes directly in the purchasing of fish products with the view to providing a guaranteed price and the capacity to purchase and store large quantities of fish, even during peak production periods. Some of the same State marketing ventures are also closely involved with the importing of fish products, particularly for the tourist industry.

The principle export markets for fish products have been the United States, Europe and their dependents, but these have become more inaccessible recently (particularly the EU) following more stringent application of health and sanitary regulations. These restrictions have also had severe implications on the regional export of fish products to the French Antilles and exporting is perceived as becoming more uncertain and risky. The export trade in conch has been suppressed in recent years due to restrictions imposed by CITES, reducing foreign exchange earnings, and encouraging fishers away from the conch fishery. However, it may have been to the benefit of the resource which was under severe pressure.

Technology and Infrastructure

Most of the inshore and coastal fishing in the region is carried out in open or half-decked vessels, powered by conventional outboard motors. There is a transformation under way and the more traditional wooden vessels are being replaced by more seaworthy 7 - 10 m glass fibre vessels powered by outboard engines rarely smaller than 75hp and often over 115hp. Of particular concern is the level of safety, the range and carrying capacity of the present fleets. Typically fuel is the most significant (around 50%) of direct operating costs. In some areas moves have been made to manage existing resources through modification of existing gears (mesh size regulations). However, the existing technologies are inadequate for the challenge of resource management and the exploitation of new resources (Wilson 1999).

The distribution of fisheries infrastructure is very uneven across the sub-region. Grant funded projects have financed the establishment of landing sites, processing infrastructure and facilities for storage of both fish and equipment (Murray 1999b). OECS Member States are unified in the opinion that the development of physical infrastructure, particularly port and storage facilities is a vital part of modernisation of the fishery, encouraging private sector investment (Wilson 1999). It is thought that this has contributed to the use of more seaworthy vessels, more consistent operating patterns, and more diversified fisheries (*ibid.*).

Investment and Financing

In each of the OECS Member States there already exists at least one development finance institution lending at subsidised rates (typically 3-5% lower than the commercial banks) and, in principle, these funds are available for investment in the fisheries sector. Grenada, Dominica and St Lucia have lines of credit specifically for fisheries sector investments (Finlay 1999, Guiste 1999, George 1999). Commercial banks are also lending into the fisheries sector, but under more stringent loan conditions. Overall, there is a high rate of delinquency and defaulting on repayments by fishers who borrow from the banks. Thus, the commercial banks will only invest with caution and require real guarantees. Although funds may be available, they are in reality difficult for fishers to access. Particular problems from the fishers' viewpoints include:

i) High initial fees (commissions, inspection fees);

ii) Cost or unavailability of insurance;

iii) Difficulty in the provision of sufficient guarantees and collateral.

In some instances loan ceilings of the development finance institutions are too low (typically around EC$ 45,000 per loan) to facilitate diversification, particularly the acquisition of higher technology investments. Recently, there are some signs that both the banks and the development finance institutions are improving their attitude towards the sector (George 1999, Ryan 1999, Wilson 1999). This is more closely associated with the appearance of a new generation of investors rather than a change in the behaviour of existing operators with the sector.

Subsidies

OECS governments have facilitated investment in the sector through subsidy schemes and tax holidays, which have given concessions including import duty reduction on capital equipment, rebates on the duty paid on fuel and tax exemptions. Across Member States, subsidies vary in their degree, the inputs that are covered, and the method of implementation (Wilson 1999). They may have long term cost implications and possibly negative effects on other sectors of the economy if investment capital is in short supply.

INSTITUTIONAL CAPACITY

Through the CARICOM Fisheries Resources Assessment and Management Program (CFRAMP) and the OECS, there has recently been a significant improvement in the training of fisheries officers (Haughton 1999). In many cases, the core staff of the fisheries departments are now young, dynamic and well qualified. However, the institutional capacity of Fisheries Departments is still inadequate (Wilson 1999), for the objectives and activities identified in the national fisheries management plans. Many elements of the fishery are in transition. Thus, with the need for a more pro-active part in the guidance of the evolution of fisheries, the maintenance and development of institutional capacity is very important, particularly if current expertise is to be retained in the sector.

FISHER DEVELOPMENT

In general, attempts at the organisation and association of fishers have not met with success, with the result that no active and truly representative bodies exist in the sub-region. There are examples of economic co-operatives (cf. George 1999) which allow members to qualify for fuel duty rebates, but they have limited involvement with other services for members and provide little political representation. The latter has proved an obstacle to facilitate fisher participation in decision making (Wilson 1999). Since fishers feel they have nothing to gain by associating, or little to lose by not associating, there is no appreciable motivation to make the effort. Key issues are recognition and empowerment, and if the *raison d'être* of an association is to spread beyond simply direct economic benefits then the association must be given power to influence the conditions which surround its members. This can present both political and strategic problems, because the view of the user group may not necessarily be in line with the government's overall objectives. On an individual level, there are indications of development amongst fishers who are becoming both more professional and more committed (Jeffery 1999). In addition, the sector is managing to attract private investment from persons who have few traditional links with fishing and they too are bringing a new level of professionalism to the sector.

FISHERIES MANAGEMENT

Advances in the management of the sub-region's fisheries have been made in the recent years, and Member States have draft fisheries management plans, installed systems for data collection and analysis, better trained fisheries division staff, and have contributed to a harmonised draft high seas fishing bill. There is however still a widely felt lack of data and background fisheries information in fields including technology, biology, catch and effort, and marketing (cf. Murray and Nichols, in press). Specifically, a serious lack of conclusive biological and catch/effort information is felt throughout the Member States. While data may be collected, there is limited capability within fisheries departments to interpret and use the information practically to guide resource management and development.

There are some problems in finalisation of the recently developed Fisheries Management Plans (FMPs), the creation of an appropriate legal framework, and the execution of some identified activities. A consultative process with stakeholders, through the establishment of Fisheries Advisory Committees (FACs), is one of the final stages in the drafting of the FMPs. The need for stakeholder participation seems to be universally accepted, but the reality has proven difficult to implement. The FACs are seen as not being truly participatory as the stakeholders have become involved in the management process at an already advanced stage (cf. also Murray et al. this volume). In some cases fisheries departments have not yet been able to establish FACs, have difficulty in allocating sufficient human and financial resources to the execution of FMPs. This difficulty is compounded by the fact that resource use conflicts in the coastal zone continue, particularly between fisheries, tourism, agriculture (pollution and erosion) and urbanisation (domestic and industrial waste).

Surveillance and the policing of national resources are seen as necessary but difficult tasks. Many OECS Member States have indicated that foreign vessels poach in their national waters, but surveillance operations are limited by the absence of funding from the profits of licenses or other rents.

DISCUSSION

Supply and Demand

The general situation of fisheries in the OECS sub-region bears many similarities to other emerging small scale fisheries around the world (Wilson

1999). The resources which are the most *accessible* to the fishers, considering their geographic location, capture technology, capital availability and markets are coming under increasing pressure and in some instances already showing signs of collapse. The inshore shallow reef resources fall into this category (Wilson 1999). On the other hand, the exploitation of offshore pelagic and deep slope resources has some important implications. Demersal deep slope species are slow maturing and stocks are easily depleted, as has already been manifested in Grenada (Finlay 1999). The exploitation of larger pelagics (tunas, wahoo, dolphin and sailfish) imply higher energy costs and the need for larger, more seaworthy vessels. In addition, the fishery for pelagic migratory stocks is by definition seasonal, with characteristic high and low periods, making financing more challenging both for the lending institution and the borrower. Politically and socially the issue is particularly poignant as such resources have traditionally supplied a large part of the domestic (non-tourist) market (Wilson 1999).

The demand for fish and fish products is growing in the OECS sub-region as a result of economic growth, population growth, and changes in diet and consumption habits. In light of the growing stress on inshore resources, States are increasingly challenged by the difficulty of answering to their own domestic fish demands, even without the additional demands of the tourist industry. The development of tourism in the sub-region has resulted in demand to which national fisheries have been unable to respond completely and, consequently, growing fish imports (Wilson 1999). In guaranteeing a minimum and permanent market, governments are effectively subsidising their fisheries and reducing the marketing risk (*ibid.*). It questionable whether these State marketing ventures are viable in commercial terms. In fact, if completely privatised, they might be forced to adopt very different operational and pricing strategies.

Stressed traditional fishery resources, national market demands and growing imports result in increasing pressure for diversification of the fishery. This may have implications in the fields of technology, financing, processing, and resource management. This is against the backdrop of a lack of conclusive biological and catch/effort data on the resources in the sub-region, and limited capability of fisheries departments, making it difficult for resource managers to make confident decisions regarding the management and development of the fisheries.

Loans and Investments

It is important to touch on the causes of the loan repayment delinquency problem mentioned earlier. The possible causes are that borrowers are either ***unable*** or ***unwilling*** to make repayments. The inability to make repayments can be the result of issues such as:

i) The fact that loans require regular payments and yet the fishing activity is highly seasonal or payments can only be made irregularly;

ii) The inability of the investment to support the total cost of financing;
iii) Natural disasters.

On the other hand, unwillingness to pay will be influenced by:

iv) Fisher education and training, particularly regarding small enterprise management;
v) The reality of previous financing activities in the sector. A culture of defaulting is very easily created by the concession of loans without the implementation of agreed penalty measures in the event of non-payment; and,
vi) Extended grace periods at the start of a loan induce unwillingness to pay.

Technology and Marketing

Energy costs are a growing issue in OECS fisheries and would become more important with the exploitation of more distant resources, especially if trip lengths remain short. Technical solutions are available (Wilson in press), and promotion of the knowledge of the existence of these alternatives should form part of a strategy for offshore fisheries development. Operation costs could also be reduced through the implementation of subsidies, such as a rebate on fuel duties (Wilson 1999). Already implemented in some States, these could be implemented in others if the loss of duty was seen to be offset by other financial and non-financial gains (*ibid.*). As operating cost reduction will make a depleted resource economically more harvestable, there would need to be a concomitant implementation of suitable management and development measures. For example, more selective gears are needed as an appropriate substitute for traps, and known technologies for offshore fishing (such as surface long lines) are seen to require active promotion. Particular need has been highlighted (Ryan 1999) for the development of multipurpose vessels capable of using a variety of gears rather than specialised craft. In one or two instances the viability of larger fishing vessels has been questioned and a study on vessel viability may be needed before an appropriate technology can be indicated for a given fishery. Experience with FADs has been particularly positive, attracting fishers away from near-shore resources. Expansion of the FAD program and the development of low technology FADs can continue to build upon this experience. FAD maintenance and management becomes increasingly important as other resource users (such as sport fishermen) begin to show interest.

The promotion of added value processing, particularly orientated towards import substitution and export promotion, is an area with clear potential. The tourist market can be viewed as a small piece of the export market that is conveniently logistically and bureaucratically much more accessible. The tourist market will, however, continue to be equally as demanding as the export market in terms of quality. HACCP based quality control programs will need to be implemented if the market is to be developed.

Management

Fishery policy objectives and the search for improved returns from the fishery have led those States with suitable resources and fleets to focus upon export markets in the European Union and United States, particularly for lobster and snappers. However, under current EU health and sanitary requirements (c.f. Satney 1999), only Antigua & Barbuda and Saint Vincent & the Grenadines can easily make exports to the EU, and even then they are restricted just to fresh fish. The EU requirements have had a particularly negative effect on those other States trading with the French Antilles (c.f. Barrett 1999, Murray 1999a), which have now become inaccessible markets.

Fisheries Department budgets are limited, qualified staff are overloaded, and there are insufficient personnel available to cope with an ever-increasing portfolio of departmental responsibilities. In addition to this, consultative and participatory approaches in fisheries management call for the development of new skills as well as new attitudes, and appropriate training may not be seen as a priority. Problems with the establishment and use of FACs in the development and execution of FMPs have shown that specific interventions need to be considered to raise awareness in fisheries department staff of participatory issues and provide appropriate training. The need for stakeholder participation in resource management is increasingly necessary in fisheries where the State does not have the requisite means to effectively police the resource. In cases where co-management has been successful, the management targets and mechanisms have often been based upon the legalisation of stakeholder objectives i.e. the State supports the stakeholder.

The earlier development of the FMPs may not have involved stakeholders at a sufficiently basic level and thus the implementation of co-management, based upon what are primarily State objectives, may run into inherent contradictions. This may make the FMPs prove socially impossible to implement, and lead to a drastic reduction in catches and subsequent food security and financial risks. The reduction in fishing effort in small-scale fisheries is historically difficult to achieve. At best, immediate goals can be attained through the diversion of effort rather than absolute reduction, if fisher livelihoods are to be sustained. Additionally, national policy statements and objectives for fisheries need to be further developed and clarified to assist the implementation of national fisheries strategies.

A number of areas can be suggested for consideration as strategic for intervention in OECS fisheries. In some instances sub-regional co-operation may be possible, whereas in others interventions can only be considered on an individual national basis. The promotion of offshore fisheries presents for many States an identifiable path for technical and economic diversification. This would need to be accompanied by suitable investigation and monitoring activities. These could be considered in the framework of a program such as the proposed CARICOM regional fisheries mechanism. Coupled with this, the training of fishers and the identification and extension of appropriate technologies could be a focus for the development for national fisheries. This could make use of regional training institutions, and/or specific intra-regional co-operation with a country, which already has an established fishery or experimental experience.

CONCLUSION

Fisheries continues to play an important and sometimes underrated part in the economies of OECS Member States, providing both full-time, part-time and seasonal employment, contributing significantly both to domestic food security and national GDP. The regional fisheries sector has made some significant advances in recent years: Member States are now equipped with fisheries management tools, better trained staff, improved fisheries infrastructure and, in some States, the beginning of a more technically advanced and diversified fleet. Both fisheries managers and fishers continue to face challenges such as the depletion of easily accessible fish resources, the need for further training (amongst both government staff and fishers), and the difficulty of the integration of stakeholders in to the management process. Exploitation of more distant resources, and the required fleet modernization, have brought particular challenges including increased capital intensity and the importance of credit, higher fuel costs and the requirement for improved vessels, gear and port facilities. In addition, developments in external markets have made many, such as those in the EU, less accessible as import regulations have tightened up. Some of these issues could be addressed through regional initiatives such as training, the development of resource materials and aspects of research, whilst others, including policy development require urgent national attention.

ACKNOWLEDGEMENTS

The authors would like to acknowledge the support of the Commonwealth Secretariat in providing funds for the studies on which this paper is based, through its Export and Industrial Development Division. We would also like to thank Ms. Angelique D.N. Antoine for her comments on an earlier draft of this contribution.

LITERATURE CITED

Barrett, A. 1999. *Nevis - A Review of the Fisheries Sector.* Background paper prepared for the Symposium on the development of a Fisheries

Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

Conservation and Fisheries Department. 1999. *Country Presentation: Virgin Islands (British).* Background paper prepared for the Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

Finlay, J. 1999. *Grenada Profile.* Background paper prepared for the Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

George, S. 1999. *Fisheries Sector Review for Saint Lucia.* Organisation of Eastern Caribbean States, Natural Resources Management Unit, and Department of Fisheries, Ministry of Agriculture, Fisheries, Forestry and the Environment. Castries, St. Lucia. 42pp.

Guiste, H. 1999. *Country Paper on the Fisheries Sector of the Commonwealth of Dominica.* Background paper prepared for the Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

Haughton, M.O. 1999. A Regional Mechanism for Sustainable Fisheries in the Caribbean. In: Organisation of Eastern Caribbean States, 1999. *Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999. Symposium Report.* Natural Resources Management Unit, Organisation of Eastern Caribbean States, Castries, St. Lucia.

Jeffery, Cheryl. 1999. *Country Paper: Antigua and Barbuda.* Background paper prepared for the Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

Murray, P.A., 1999a. *A Review of the Fisheries Sector of Anguilla.* Background paper prepared for the Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

Murray, P.A., 1999b. Some Perspectives in Investment and Financing for Fisheries in the OECS. *IN* Organisation of Eastern Caribbean States, 1999. *Symposium on the development of a Fisheries Management and*

Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999. Symposium Report. Natural Resources Management Unit, Organisation of Eastern Caribbean States, Castries, St. Lucia.

Murray, P.A. and K.E. Nichols, 1999. Management issues affecting the fisheries of OECS Member States. *Proc. Gulf Carib. Fish. Inst.* **50**:1053-1064.

Murray, P.A., K.E. Nichols, V.M. Chase and R. Hodge. 2001.Logical Framework Analysis as a tool for management of a tropical fishery. *Proc. Gulf Carib. Fish. Inst.* **52**:258-270.

O'Garro, M.O.V. 1999. *Montserrat Country Paper: Status of the Fisheries Sector.* Background paper prepared for the Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

Ryan, R. 1999. *Fisheries Status Report 1997.* Background paper prepared for the Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

Satney, A.O. 1999. International and Regional Fish Marketing: perspectives for the OECS. *IN* Organisation of Eastern Caribbean States, 1999. *Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999. Symposium Report.* Natural Resources Management Unit, Organisation of Eastern Caribbean States, Castries, St. Lucia.

Simmonds, J. 1999. *Country Paper: St. Kitts.* Background paper prepared for the Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999.

Wilson, J. 1999. Country Report Summary Paper. *IN* Organisation of Eastern Caribbean States, 1999. *Symposium on the development of a Fisheries Management and Development Strategy for the OECS Region. Administrative Centre, Kingstown, St. Vincent and the Grenadines, April 14-16, 1999. Symposium Report.* Natural Resources Management Unit, Organisation of Eastern Caribbean States, Castries, St. Lucia.

Wilson, J. In press. Fuel and Financial Savings for Operators of Small Fishing Vessels. *FAO Fisheries Technical Paper.* 70pp.

Implications of Optimal Effort Reductions in the Florida Commercial Spiny Lobster Fishery

J. WALTER MILON[1], SHERRY L. LARKIN[1] and NELSON EHRHARDT[2]
[1]*Food and Resource Economics Department*
University of Florida
Gainesville, Florida 32611-0240 USA
[2]*Rosenstiel School of Marine and Atmospheric Science*
University of Miami
Miami, Florida 33149-1098 USA

ABSTRACT

An integrated bioeconomic analysis – using alternative surplus production and harvesting cost functions – provided a range of effort that would maximize economic yield in the Florida commercial spiny lobster fishery. Despite a 35% trap reduction since 1992, when a transferable harvest rights program was implemented, effort remains too high and trap certificates appear undervalued. It is unlikely that the optimal effort level will be achieved under the existing program given the lack of a reduction goal, prolonged reduction approach, and waning support for the program from delayed benefits. A comprehensive long-range management plan is needed to reap the benefits – economic and environmental – that were predicted and desired from the trap certificate program.

KEY WORDS: Bioeconomic model, fisheries management, surplus production model

INTRODUCTION

Florida's spiny lobster (*Panulirus argus*) fishery is one of the state's most important fisheries, ex-vessel landings were valued at nearly $30 million in 1996 (NMFS 1997). From 1960 to the early 1990s, commercial fishing effort expanded from less than 100,000 traps (the dominant gear type) to more than 900,000 (Hunt 1994). Despite the significant increase in effort, total commercial landings varied little, fluctuating between 2,400 and 3,580 metric tons (mt) per year since 1969. Even though the significant increase in effort did not have an effect on landings that would cause concern for the health of the stock, it did raise several other concerns. In 1991, the Florida Legislature observed that:

> Due to rapid growth, the spiny lobster fishery is experiencing increased congestion and conflict on the water, excessive mortality of undersized lobsters, a declining yield per trap, and public concern over petroleum and debris pollution from existing traps (Florida Statute 370.142(1)).

The number of traps was eventually regulated in 1992 when the Florida

Legislature implemented the Trap Certificate Program (TCP). The mandated goal of the TCP is "to stabilize the fishery by reducing the total number of traps, which should increase the yield per trap an d therefore maintain or increase overall catch levels" (Florida Statute 370.142(1)). The TCP ended an era of open-access management of the spiny lobster fishery in Florida by establishing a cap on total effort. The program is one of the first individual transferable effort programs in the United States. Under the TCP, qualified commercial fishers own "certificates" that entitle the owner to fish a specified number of traps (each certificate allows the use of one trap). All traps are identical, since trap size and design are regulated. Each year, fishers pay an annual certificate fee ($1.00 in 1998 - 1999) and, in return, receive a tag for each certificate owned. The tags are attached to the traps and indicate the trap is legal for that season (tags are color-coded each season and stamped with a certificate number that can be used to identify the owner). Certificates are transferable, all or in part, among fishers.

The total number of certificates, which is considered a proxy for the total level of effort allowed in the fishery, has been periodically reduced in accordance with the stated goal of the program (Florida Statute 370.142(1)). The Statue that established the program did not, however, specify the total number of traps to eliminate from the fishery. Since 1992, periodic reductions in the total number of certificates have eliminated approximately 35 percent of the commercial traps (Milon et al. 1998). It is not clear, however, whether these reductions have been too much or too little relative to an "optimal" (e.g., profit maximizing) number of traps in the fishery.

The purpose of this study was two-fold. The first goal was to determine the total number of traps that would maximize the net economic benefits in the commercial fishery, and thus, test the hypothesis that previous regulatory actions have achieved an economically optimal and sustainable number of traps in the fishery. This was accomplished by estimating biological production and harvesting cost models for use in an integrated bioeconomic analysis. The second goal was to assess the effects of moving the fishery toward the optimal solution. The evaluation of these effects will focus on the number of participants, the market for trap certificates (including observed prices and transfers), and the implications for future management of the fishery.

MODELING APPROACH

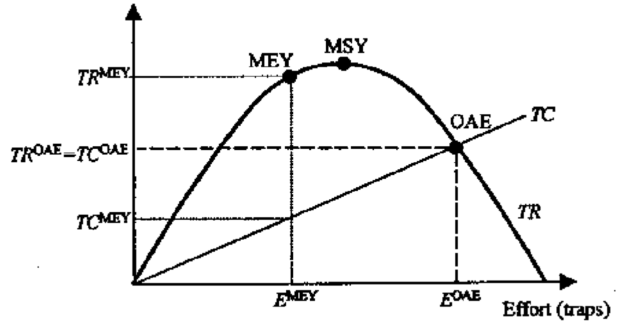
Bioeconomic theory for a commercial fishery posits that the socially optimal level of catch and effort is determined by the biological dynamics of the stock, harvesting costs, and the products' market price (Hartwick and Olewiler 1998). This is because society is interested in stock conservation and the profitability of the industry. Without entry or effort restrictions, harvest continues to the breakeven point – an effort level where total revenues just cover total costs (*TR* = *TC*) – which is known as the open-access equilibrium (OAE). Using the well-known Schaefer (logistic) yield-effort curve, and assuming constant costs and prices, the OAE solution is shown in Figure 1(a) (Gordon 1954). The OAE (unregulated) equilibrium is socially inefficient (suboptimal) because the same total revenue can be achieved at a lower cost. At the OAE solution the additional effort incurs additional costs, which completely offset the total revenues (profits are zero). In addition, the level of catch at OAE is less than at the maximum sustainable yield (MSY), which occurs at the height of the total revenue (*TR*) curve. The relative catch levels can be identified using the *TR* curve since the *TR* curve retains the same shape as the underlying sustainable yield (catch-effort) curve when price is constant. The MSY catch level represents the largest quantity that can be harvested on a sustainable basis (i.e., without compromising the stock); it is the harvestable surplus. Throughout the remainder of the analysis the terms catch, yield, harvest, and landings (variable *C*) will be used interchangeably.

From society's point of view, the maximum economic yield (MEY) is the optimal solution since industry effort is increased only to the point where additional revenues are offset by harvesting costs (Gordon 1954). This solution is identified by equating the slopes of the total revenue and total cost curves (i.e., where marginal revenue equals marginal cost). TR^{MEY} minus TC^{MEY} represents the maximum profit per unit effort in the fishery. In the traditional example shown in Figure 1(a), the MEY effort is less than needed to take the MSY. This solution would also provide the maximum rents to the fishery if costs included the opportunity cost of capital and labor (i.e., the market value of alternative uses for the resources). Since opportunity costs are often difficult to measure in fisheries, most empirical studies attempt to measure only profit changes (Hartwick and Olewiler 1998).

To estimate an MEY solution for the Florida commercial spiny lobster fishery, we must first estimate a sustainable yield curve. The sustainable yield curve, also known as the surplus production function, describes the aggregate effects of natural mortality, growth, and recruitment in a single compensatory function. According to Menzies and Kerrigan (1980), surplus production models can be used when the relationship between the local stock size and future recruitment is weak or unknown. In addition, these models have relatively modest data requirements and are particularly useful as first approximations (Clarke, Yoshimoto, and Pooley 1992). The shape of this curve depends on assumptions regarding the growth rate of the stock. For example, the traditional logistic model in Figure 1(a) assumes a density-dependent growth pattern whereby the sustainable annual harvest is dependent on the size of the local population in previous years. This specification is characterized by the potential for complete depletion of the stock since catch can be driven to zero at excessive levels of effort.

Recent studies have concluded that spiny lobster recruitment in Florida is dependent, at least in part, on the size of the spawning stock in waters adjacent to Florida (Ehrhardt 1994). In addition, the Florida fishery prohibits harvest (1) during spawning season, (2) of egg-bearing females, and (3) of juvenile (undersize) individuals. According to Clarke, Yoshimoto, and Pooley (1992), if recruitment into a fishery is exogenous or local regulations are sufficient to maintain recruitment, a logarithmic production function is most appropriate. A logarithmic production function, such as shown in Figure 1(b), assumes the sustainable yield is not entirely dependent on stock size so increasing effort eventually has no effect on total catch.

1(a) Logisitic (Schaefer) Production Function



1(b) Logarithmic Production Function

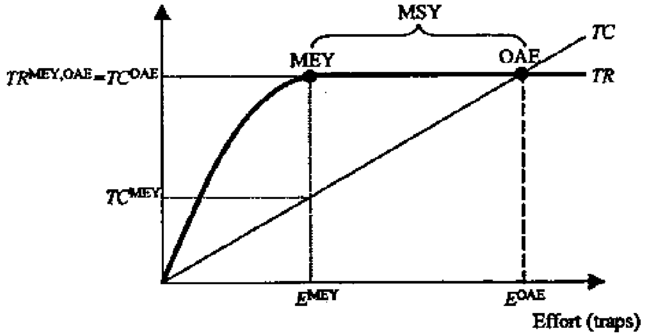


Figure 1. Equilibrium solutions with alternative production functions

As illustrated in Figure 1(b), a logarithmic or "flat-top" sustainable yield curve has a wide range of effort levels that produce the MSY solution. Effort at the open-access equilibrium (E^{OAE}) is greater than effort that maximizes economic yield (E^{MEY}). Since total revenues are the same at either E^{MEY} or E^{OAE}, society is not making the best use of its resources by increasing effort from E^{MEY} to E^{OAE}. The additional effort at the OAE solution dissipates profits that would be earned at E^{MEY} since costs are higher. Thus, even if the biological relationship indicates that additional effort will not threaten sustainability of the stock, the bioeconomic framework shows that it is necessary for management to restrict effort in the fishery to achieve an economically efficient allocation of resources.

DATA

Catch and effort data, as well as records of certificate transactions since the inception of the TCP, were obtained from the State of Florida. The catch and effort data consist of annual landings and trap use for the 1960 - 1961 through 1997 - 1998 seasons. Data reported on the East Coast from 1964 to 1975 were adjusted to remove landings and traps associated with fishing in the Bahamas (data and correction details are available in Milon, Larkin, and Ehrhardt 1999). Landings are the quantity purchased (whole weight) by licensed wholesale dealers and are assumed to equal total catch (C). Fishing effort (E) is the total number of traps operated by commercial fishermen. It is implicitly assumed that fishing practices have not changed over time and do not differ among fishers. These are valid assumptions given that trap size and construction have been regulated since the 1960s (Milon et al. 1998). Also, fishing technology changes may have increased the rate of harvest – which is accounted for in the cost information and biological coefficients – but would not have affected resource availability and, therefore, estimation of the surplus production function. These data exclude the recreational sector since statistics are not available for the entire period and effort is measured differently. Omitting the recreational data will not, however, affect the shape of the production function since recreational landings have remained a relatively constant share of total landings since recreational data collection began in 1991 (Hunt et al. 1998).

Annual landings in Florida averaged approximately 1,500 mt during the 1960s, but have averaged 2,850 mt since and fluctuated without an apparent trend. Total effort increased significantly from 1960 to 1992, from less than 100,000 to nearly one million traps. The dramatic increase in traps with relatively stable landings caused the average trap yields to fall approximately 75 percent from 1970 to 1990. Since the TCP was implemented in 1992 the number of traps has been reduced to approximately 544,000 (Milon et al. 1998).

Cost data needed to estimate the marginal cost per trap were obtained during interviews conducted with a stratified sample of lobster fishers (n = 53) in the Florida Keys (Milon, Larkin, and Ehrhardt 1999). Variable costs included trip costs (fuel, bait, groceries, ice, supplies, and labor payments), equipment leasing and repair, and maintenance expenses incurred during the 1996-97 season. These costs averaged $16,366 per vessel exclusive of labor. Labor payments equaled $12,950 assuming the captain and crew were paid minimum wage ($5.15 per hour). Using the minimum wage was necessary since preliminary surveys indicated a variety of compensation methods were us ed and this information was a sensitive issue that many did not wish to discuss. Basing labor costs on the minimum wage provides an estimate of the minimum opportunity cost associated with work hours expended in this fishery. Fixed costs averaged $21,238 per vessel annually and included interest payments, docking fees, depreciation (vessels and gear), and licensing.

RESULTS

Biological Production Models

Two flat-top production models were estimated for this fishery. The empirical models and corresponding catch-effort curves are shown in Figure 2. The first, dubbed the "Effort-Corrected Schaefer" model, incorporated the effects of trap density into a traditional Schaefer production model by specifying the catch rate as an inverse function of total effort. In addition, a relative trap efficiency parameter was estimated and used to standardize effort over time. Both parameters in the Effort-Corrected (E-C) Schaefer model were statistically significant at the one percent level and the estimated model was highly significant overall ($F_{1,34}$ = 69.9). The second flat-top production model, referred to as the "Biomass Utilization" model, assumed that catch was a function of the catch rate and the maximum catch possible. As with the E-C Schaefer model, the catch rate in the Biomass Utilization (BU) model incorporated the effect of trap density on yield. Catch was estimated as the difference between the asymptotic (maximum) catch minus the catch that survived fishing effort. This model is unique in that catch is a function of the available "catchable" biomass without taking population regeneration into consideration. Using the nominal data, the estimated BU model was statistically significant at the one percent level ($F_{1,34}$ = 71.1). See Milon, Larkin, and Ehrhardt (1999) for further detail.

Figure 2 shows the relationship between the total number of traps and sustainable landings predicted by each model. Both models predict landings would increase at a decreasing rate until approximately 400,000 traps. Landings would then remain constant at approximately 2,800 mt as effort increased.

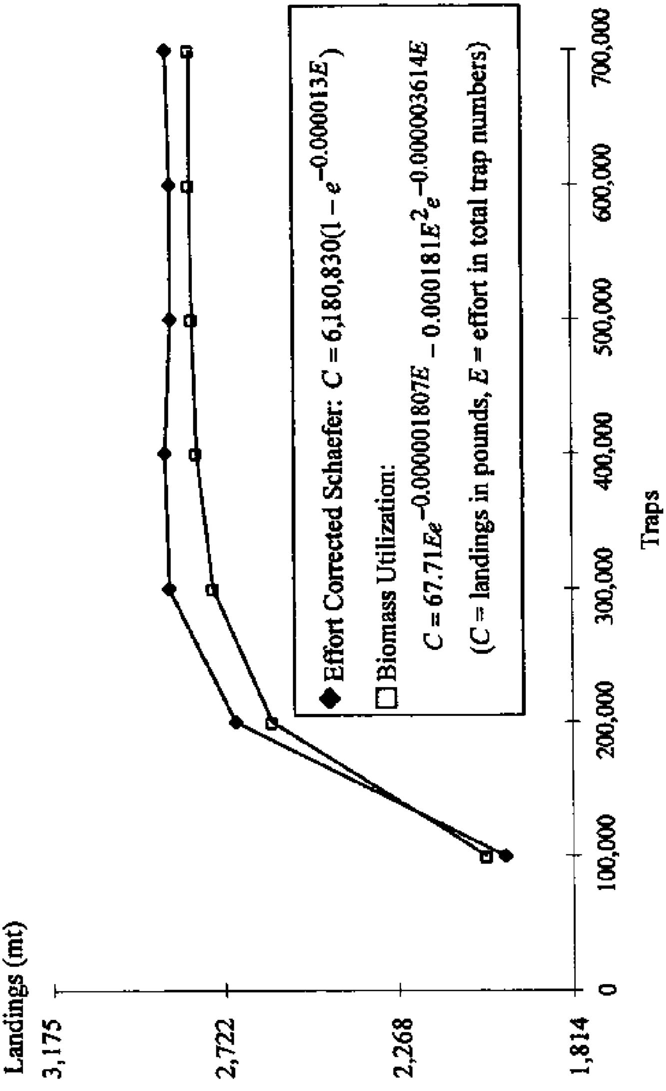


Figure 2. Estimated biological production functions: C - f(E)

Cost of Production Models

Two cost models were estimated for this study. The first followed Prochaska and Cato (1980) in which the total annual cost of lobster fishing for each vessel (TC_i) was regressed against the number of traps operated by the firm (E_i): $TC_i = \alpha + \beta E_i$. This equation provided an estimate of the annual fixed cost for each vessel (α) and the corresponding marginal cost per trap (β). The costs estimated from this specification represent the short-run costs of fishing. The second model assumed all costs are variable and, in particular, depend on the level of effort (i.e., number of traps fished): $TC_i = \beta E_i$. T his is the appropriate specification for a long-run analysis.

The two cost models were estimated using a least squares estimator with the 1996 - 1997 survey data. Both marginal cost estimates were statistically significant at the one-percent level. The short-run cost curve estimated annual fixed costs at $14,901 and marginal cost at nearly $29.73 per trap ($R^2 = 0.50$). The long-run cost curve estimated marginal cost at $38.81 per trap ($R^2 = 0.87$), approximately 30 percent above the short-run cost.

Integrated Bioeconomic Analysis

Four OAE solutions were found by equating the two total revenue curves (*TR*) – for the E-C Schaefer and BU models – with the short- and long-run total cost curves (*TC*). These solutions, where profits are driven to zero, occur at effort levels between 565,729 and 614,269 traps. For comparison, the fishery was operating at approximately 605,000 during the 1996 - 1998 seasons (Milon et al. 1998). Average yields under the OAE solutions ranged from 4.6 to 5.0 kg per trap, closely matching observed yields. Table 1 summarizes the OAE solutions.

Table 1 also includes a description of the four MEY solutions, which were found by equating the two marginal revenue curves (*MR*) with the two marginal cost estimates (*MC*). The marginal revenue curves were derived by multiplying the estimated marginal productivity curves (i.e., the slope of the biological production functions in Figure 2) by the average unit price in 1996 ($8.36 per kg, NMFS 1997). The profit-maximizing number of traps in the fishery ranged from 158,619 to 198,523, each yielding from 13.5 to 15.4 kg for total landings of 2,446 mt to 2,681 mt. Industry profits ranged from $13.7 million to $14.8 million, or $71 to $90 per trap. Using the long-run cost curve resulted in fewer traps and lower landings but higher landings per trap. It is notable that the range of MEY solutions encompasses early estimates by Prochaska and Cato (1980) who found 169,335 traps landing 15.5 kg annually would maximize net revenues. Also, these estimates are consistent with survey data from the early 1970s that showed average trap yields of 14.5 kg per trap when approximately 147,000 traps were in the fishery (Williams and Prochaska 1976).

Table 1. Range of Equilibrium Solutions

	Total Revenue Specification			
	OAE Solutions		MEY Solutions	
Cost Specification	Biomass Utilization	E-C Schaefer[a]	Biomass Utilization	E-C Schaefer[a]
Short-run Cost:[b]				
Effort (E)	565,729 traps	575,592 traps	179,159 traps	198,523 traps
Catch (C)	2,802 mt	2,851 mt	2,529 mt	2,681 mt
	(5.0 kg per trap)	(5.0 kg per trap)	(14.1 kg per trap)	(13.5 kg per trap)
Total Profits	$0	$0	$13,720,529 ($77 per trap)	$14,182,009 ($71 per trap)
Long-run Cost:				
Effort (E)	603,350traps	614,269 traps	158,619 traps	178,655 traps
Catch (C)	2,803 mt	2,853 mt	2,446 mt	2,599 mt
	(4.6 kg per trap)	(4.6 kg per trap)	(15.4 kg per trap)	(14.5 kg per trap)
Total Profits	$0	$0	$14,278,428 ($90 per trap)	$14,785,497 ($83 per trap)

[a] Where $q = 0.000001807$.

[b] Assuming each vessel fished 1,279 traps (the sample average) in order to calculate total fixed costs.

From the bioeconomic analysis, we know the value of each certificate (trap) if the total number of traps were optimal (i.e., from approximately 160,000 to 200,000). If the transfer market for trap certificates is working properly (e.g., buyers and sellers can exchange easily and at a reasonable cost), the observed certificate transfer price should closely match the estimated optimal certificate value. The difference between the average reported price of a certificate and the estimated optimal certificate value could be used as a rough approximation of the gains from certificate reductions, that is, the gains to moving toward the MEY solution.

IMPLICATIONS

The bioeconomic analysis revealed that if reductions were to continue until the economically efficient number of traps is reached, economic efficiency in the fishery would reach a maximum. Certificate values would range from $70 to $90 per certificate, which translates to earnings of 77 to 128 percent above average annual costs reported during the 1996 - 1997 season (assuming 1,279 traps per vessel). However, the MEY solutions occur at effort levels that are approximately 35 percent of current trap numbers. The need for significant effort reductions should be expected to affect fishery participants, the market for certificates, and the future management of the fishery. The implications of such reductions for each sector are considered below.

Industry Participants

The bioeconomic analysis indicates that future effort reductions in the commercial spiny lobster fishery could significantly increase the profit per trap and the value of certificates. The estimated optimal number of traps, however, could imply a significant reduction in the total number of vessels in the fishery. For example, using the average number of traps per vessel (1,279) reported in the recent cost study by Milon, Larkin, and Ehrhardt (1999) and the optimal MEY effort levels from this study (158,619 to 198,523; Table 1), optimal fleet size would range from 124 to 155 vessels. Each vessel would earn profits of from $91,497 to $115,149 annually assuming 1996 costs and $8.36 per kg price. A fleet of 124 vessels represents a 75 percent reduction in the number of full-time operators. Although the TCP contains restrictions that ensure at least 76 certificate owners, greater concentration of certificate ownership can have effects on other segments of the industry. For example, fewer harvesters could cause consolidation in the processing sector and further reduce employment. Previous reductions have not, however, significantly increased concentration in the Florida spiny lobster fishery (Milon et al. 1998).

Certificate Market

During the 1996 - 1997 season, 604,920 certificates were available – approximately three times the optimal number of traps – and reported certificates prices averaged from $4.47 to $15.52 depending on the certificate type and calculation method (Milon et al. 1998). Although lower than the price that would be expected if the number of traps were near optimal (i.e., $70 - $90, table 1), these prices are near the value expected with the current number of traps (544,000 in 1998 - 1999). For example, given an average annual yield of 5.8 kg per trap and average price of $8.36 per kg, gross returns are approximately $48.50 per trap. Net returns would range from $9.69 to $18.77 using the estimated long- and short-run marginal costs, respectively. For comparison, the annual profit per trap from the BU model would equal $13.30 at the 1998-99 certificate level. Consequently, the trap values from the estimated bioeconomic models are similar to the average transfer prices reported to date.

The total number of certificates, volume of certificate transfers, and certificate composition (by type) is important since each affects the revenue collected by the State of Florida. The total number of certificates determines the maximum annual revenue from certificate fees, which were $1 for the 1998 - 1999 season. If reductions were to continue toward the MEY solution, certificate fee revenue would decrease proportionately. The volume of certificate transfers is important since each certificate transferred is subject to a $2 fee. The number of certificates transferred annually fell from approximately 91,000 in 1994 to just over 35,000 in 1998 (Milon et al. 1998). If the total number of transfers continues to decline, so will the revenue collected by the State of Florida. A decline in the number of certificates and transfers can further reduce revenues by reducing surcharges, which depend on the type of certificate transferred. There are three types of certificates. Certificates originally issued are Type A-1. Certificates sold to an immediate family member are Type A-2 and are exempt from a transfer surcharge. Certificates sold to non-family members are Type B and are subject to a one-time transfer surcharge equal to 25 percent of the value of the transaction. From 1993 to 1999, the composition of certificates changed as the number of Type B certificates increased to 31 percent of the total (Milon et al. 1998). The percentage of Type B certificates is important since the State of Florida will not collect a surcharge from the subsequent transfer of these certificates. Revenue reductions to the State of Florida could compromise the fiscal self-sufficiency of the TCP.

Future Management

The bioeconomic optimal solutions (MEY), which maximize rents to the industry, offer several advantages over the open-access alternative. First, there would be a reduction in the amount of labor needed in the fishery, which could increase returns to certificate owners. Second, it would eliminate excess gear. Fewer traps would reduce the negative environmental consequences (e.g., debris and "ghost" fishing) from lost and abandoned traps. The relatively low cost of a trap is a disincentive to retrieve traps at the conclusion of the season. Lastly, the MEY solution allows for the possibility of residual rents. Resource managers could redistribute, if desired, a portion of the increased returns to the citizens of Florida in the form of an "equitable rent per trap" as stipulated in the original legislation (Florida Statute 370.142(2)).

Moving the industry toward the MEY solution could also create additional management issues that need to be addressed. For example, there would likely be impacts on local communities and other fisheries as participants are initially displaced. On the other hand, increased rents could attract effort from other fisheries and cause poaching and illegal trap use, requiring the need for increased enforcement of regulations. Of course the status quo situation, which is near the open-access (OAE) solution, will need to contend with many of these issues regardless of the course of action pursued.

DISCUSSION

The Florida spiny lobster TCP was implemented to alleviate several problems faced by rapid growth in the industry including: declining trap yields, increased congestion and conflict on the water, and environmental concerns from trap debris (Florida Statute 370.142(1)). The program allocated trap certificates, defined a mechanism to periodically reduce certificates, and allowed transfer of ownership. However, the TCP failed to state an overall effort reduction goal even though estimates were available (e.g., Prochaska and Cato 1980, Waters 1996). Consequently, six years after implementation, there has been little effect on trap yields, which is not surprising given that effort reductions fall far short of the estimated reductions needed to reach MEY in the fishery. The relatively slow adjustment to lower effort levels is due, in part, to the 10% cap on annual reductions. This approach lowers subsequent reductions, which slows the movement toward the optimal solution. Since the beneficial effects of trap reductions are not expected until total trap numbers fall below 300,000, at least six future reductions are required to see results. Ten reductions in total would be required to achieve the MEY range of optimal effort estimates. In addition, with the alternate year reduction schedule (which began with the 1998 - 1999 reduction), the MEY level would not be reached until 2019. It is unlikely that support for the program could be maintained over the 20 years needed to show these results. As Johnson and Libecap (1982) suggest "fishermen are more likely to support arrangements that do not affect status quo rankings and that increase their total catch – such as season closures, hatcheries, gear restrictions to protect juvenile fish, and controls on fishing by members of other groups."

This is because "those programs raise rents for existing fishermen above open access conditions, even though dissipation continues along other margins. In the absence of political support from fishermen, politicians, and bureaucrats facing periodic reelection and budget review will not pursue efficiency goals in regulation if the programs are controversial, as is likely" (Johnson and Libecap 1982, p. 1019).

The bioeconomic analysis and the evaluation of the TCP provide valuable insights into the strengths and weakness of one of the nations first transferable rights programs. It provides an example of how economic analysis can be used to evaluate the effects of one approach to correcting problems in fisheries management. The lack of a measurable goal (i.e., number of certificates to be eliminated) has already led to dissention between resource managers and fishers that has undermined the effectiveness of the program (by delaying effort reductions) and compromised future reductions. Consequently, there is an immediate need for a more comprehensive approach to deal with program goals and the mechanisms to achieve those goals (Milon et al. 1998)

LITERATURE CITED

Clarke, R.P., S.S. Yoshimoto, and S.G. Pooley. 1992. A bioeconomic analysis of the Northwestern Hawaiian Islands lobster fishery. *Marine Resource Economics* **7**:115-140.

Ehrhardt, N.M. 1994. The lobster fisheries off the Caribbean coast of Central America. Pages 133-143 in: B. Phillips, J. Cobb, and J. Kittaka, (eds.) *Spiny Lobster Management*. Blackwell Scientific Publications Inc., Cambridge, MA.

Gordon, H.S. 1954. The economic theory of a common-property resource: The fishery. *Journal of Political Economy* **62**:124-142.

Hartwick, J.M., and N.D. Olewiler. 1998. *The Economics of Natural Resource Use*, 2nd ed. Addison-Wesley Educational Publishers Inc., New York, NY, 432 pp.

Hunt, J.H. 1994. Status of the fishery for *Panulirus argus* in Florida. Pages 158-168 in: B. Phillips, J. Cobb, and J. Kittaka, (eds.) *Spiny Lobster Management*, Blackwell Scientific Publications Inc., Cambridge, MA.

Hunt, J.H., W.C. Sharp, T.R. Matthews, R.G. Muller, R.D. Bertelsen, and C. Cox. 1998. Status of the spiny lobster fishery in Florida, 1998. Florida Department of Environmental Protection. Marathon, FL. 15 pp.

Johnson, R.N., and G.D. Libecap. 1982. Contracting problems and regulation: the case of the fishery. *American Economic Review* **72**(5):1005-1022.

Menzies, R.A., and J.M. Kerrigan. 1980. The larval recruitment problem of the spiny lobster. *Fisheries* **5**(4):42-46.

Milon, J.W., S.L. Larkin, and N. Ehrhardt. 1999. A bioeconomic analysis of the Florida spiny lobster fishery. Florida Sea Grant, Report Number 117, Gainesville, FL. 99 pp.

Milon, J.W., S.L. Larkin, D.J. Lee, K.J. Quigley, and C.M. Adams. 1998. The performance of Florida's spiny lobster trap certificate program. Florida Sea Grant, Report Number 116, Gainesville, FL. 71 pp.

NMFS (National Marine Fisheries Service). 1997. *Fisheries of the United States, 1996.* Current Fishery Statistics No. 9600. Silver Spring, MD. 168 pp.

Prochaska, F.J., and J.C. Cato. 1980. Economic considerations in the management of the Florida spiny lobster fishery. *Fisheries* **5**(4):53-56.

Waters, J. 1987. Economic source document for analysis of limited entry alternatives for the spiny lobster fishery in Florida. NMFS. Baeufort, NC. 14 pp.

Williams, J.S., and F.J. Prochaska. 1976. Maximum economic yield and resource allocation in the spiny lobster industry. *Southern Journal of Agricultural Economics* **9**:145-150.

Description de las Temperaturas Usadas en el Cultivo de Larvas de *Strombus spp.*

DALILA ALDANA ARANDA[1], CÁRDENAS BAQUEIRO[2] and VICTORIA PATIÑO SUÁREZ[1]
[1]CINVESTAV IPN Unidad Mérida
Laboratorio de Biología marina
Km. 6 antigua Carretera a Progreso
C.P. 97310 Mérida Yucatán México
[2] CRIP de Campeche-IPN
Campeche, Mexico

RESUMEN

Strombus gigas, S. costatus, S. pugilis, S. raninus, S. gallus, and *S. alatus* son especies de importancia comercial en el Caribe. Economicamente *Strombus gigas* es la mas importante y consequentemente la mas estudiada. En esta descripción se analizan los effectos de la temperatura en el desarrollo de las etapas de crecimiento de la concha de los veligers, la taza de metamorfosis, la taza de establecimiento y la taza de la superviviencia. Por mucho tiempo se ha sugerido que la temperatura como un factor importante en la regulación de la taza de desarrollo, la duración de la etapa pelagica y de la mortalidad de las larvas de invertebrados marinos béntonicos. Es bien conocido que el establecimiento temprano de las larvas detro de ciertos límites, se relaciona directamente con la temperatura del agua. El propósito de este articulo es analizar los efectos de la temperatura (24-32 ⁰ C) en el desarrollo del veliger hasta metamorfosis. El cual se logra en approcimadamente en 14 diás a temperaturas de 29°C y en 60 diás a temperatures de 27°C, Por que?

PALABRAS CLAVE: Conch, larvas, temperatura

Overview of Temperatures Used in Larviculture of *Strombus* spp.

ABSTRACT

Strombus gigas, *S. costatus*, *S. pugilis*, *S. raninus*, *S. gallus* and *S. alatus* are of commercial importance in the Caribean. Economically, *S. gigas* is the most important and consenquently the most widely studied. In this overview is analyzed the effects of temperatures on the development of shell growth stages of conch veligers, metamorphosis rate, settlement rate an survival rate. Temperature has long been suggested as an important factor regulating the developmental rate, length of pelagic life, and mortality of larvae from benthic marine invertebrate organisms. It is known that the rate of early cleavage, within certain limits, is related directly to water temperature. The purpose of this paper is to analyze the effects of temperature (24 - 32°C) on the veliger's development until metamorphosis which is attained within 14 days at 29°C to 60 days at 27°C, Why ?

KEY WORDS: Conch, larvae, temperature

INTRODUCCION

De Jesús (1999) analizo los temas de los artículos científicos publicados sobre Strombus gigas (n = 230) y encontró que ha habido una evolución de estos desde los años 1950s en que se escriben los primeros. Así en los años 1960s la mayoría de los artículos fueron sobre la biología genera, en los 1970s el principal tema fue la biología pesquera. Al inicio de los 1980s y como consecuencia de la sobre-explotación del recurso en la mayoría de los países del Caribe, se da una fructífera producción de información sobre el cultivo larvario, producción de juveniles y los primeros ensayos de siembra de éstos en el medio natural con fines de repoblamiento. Posteriormente, en los años 1990s, cuando los trabajos de cultivo no proporcionan los resultados que se habían proyectado, se inicia una novedosa línea de investigación en Ecología de larvas y juveniles con el grupo de trabajo de Stoner y Davis, estudiándose el reclutamiento de la especie (Stoner and Davis 1994, Stoner and Ray 1993, Stoner et al. 1996; Stoner et al. 1998, Stoner and White 1990, Stoner 1990, Ray-Culp et al. 1999).

Los estudios de ecología y biología de larvas son de utilidad para comprender mejor los procesos de reclutamiento en el medio natural, así como la aplicación de esta misma información para mejorar los resultados de las técnicas de acuicultura para la producción de semillas en el laboratorio.

Una utilidad adicional de los experimentos realizados en laboratorio.sobre biología y ecología de larvas, es la ayuda que proporcionan para comprender los procesos de asentamiento, desarrollo y sobrevivencia, dado que el seguimiento de los organismos, en particular de larvas en el océano es difícil. Así, la mayor parte de la información disponible acerca de la vida larvaria proviene de estudios en laboratorio.

Davis, et al. (1996), señalan que el comprender los factores que afectan el crecimiento, desarrollo y sobrevivencia en larvas de invertebrados marinos en el campo, proporciona un conocimiento pertinente para determinar la dispersión larval y el proceso de reclutamiento

Con el objetivo de incrementar el conocimiento científico sobre la biología de larvas y su aplicación a mejorar las técnicas de cultivo y comprender los procesos de reclutamiento, en el presente trabajo se realizó una revisión del efecto de la temperatura en el cultivo larvario que han sido utilizadas para *Strombus gigas*, *Strombus costatus* y *Strombus pugilis*. Se resume también información de la posible relación entre la temperatura con los resultados de crecimiento, desarrollo, metamorfosis y sobrevivencia.

Condiciones de Cultivo y Temperatura para la Crianza de Larvas Velígeras de *Strombus gigas*

S gigas es una de las especies más importantes de especies pescadas en el Caribe, y su cultivo ha sido ampliamente estudiado, en menor grado lo han sido *S. costatus* y *S. pugilis* , incluso las técnicas desarrolladas para la primera especie han sido aplicadas indistintamente para las otras dos.

En la Tabla 1 son resumidas las diferentes condiciones que han sido utilizadas para el cultivo larvario de *S gigas* , los resultados están expresados en términos de tasa de crecimiento, longitud máxima, duración de la metamorfosis y sobrevivencia.

De la revisión de artículos publicados sobre las condiciones de cultivo larvario de *Strombus gigas*, se observa que solamente 3 trabajos han sido específicamente para estudiar el efecto de la temperatura durante el desarrollo larvario (Aldana Aranda et al. 1987, Corral y Ogawa 1985, Bradshaw-Hawkins 1982). Los demás autores han realizado sus experiencias de cultivo larvario bajo un rango de temperatura variable o pocos de ellos bajo una temperatura constante. Incluso el 30% de los autores citados en la tabla anterior, ni siquiera señalan la temperatura que emplearon para correr su experimento. Mucho menos presentan alguna relación entre la temperatura y el crecimiento, desarrollo o sobrevivencia en la vida larvaria de *S. gigas*.

A partir de la Tabla 1, se observa que en la vida larvaria de los Strombidos , en particular de *S. gigas*, la temperatura es un factor que ha sido poco evaluado su efecto en el proceso de crecimiento, desarrollo y competencia para la metamorfosis.

Respecto al rango de temperatura utilizado para el cultivo larvario *S. gigas* este ha sido entre 23 y 31°C. Del análisis visual de los datos de esta tabla se observa que los mejores resultados en términos de crecimiento y metamorfosis son observados en el rango de temperatura entre 27 y 31°C. En este rango Davis et al. (1993) obtiene la metamorfosis de *S. gigas* a los 21 días. En contraste tres décadas antes, D'Asaro (1965), reporta por primera vez larvas metamorfoseadas de esta especie en 60 días con una temperatura promedio de 25°C. Aldana-Aranda and Torrentera (1987) trabajando con larvas de *S. gigas* lcultivadas a 20 ± 2°C, 24 ± 3°C and, 28°C observaron una mortalidad masiva al quinto día de cultivo para las larvas cultivadas a 20 ± 2°C; y al noveno día para aquellas cultivadas a 24 ± 3°C. Contrariamente, la mortalidad de las larvas cultivadas a 28°C fue observada 10 días después de iniciado su cultivo y la metamorfosis de estas larvas se presento a los 20 días.

Ahora bien, los mismos valores de la Tabla 1 fueron analizados estadísticamente, para establecer posibles correlaciones entre la temperatura y los siguientes parámetros: tasa promedio de crecimiento, longitud máxima alcanzada al momento de la metamorfosis y tiempo en que alcanzan la metamorfosis. Sólo el tiempo que utilizan las larvas para alcanzar la metamorfosis presento un coeficiente de correlación por arriba de 0.75, las otras relaciones de la temperatura no presentaron una correlación alta mayor a 0.80. A continuación las curvas que presentaron la mejor correlación y su ecuación.

Table 1. Results in growth, maximum length (ML), metamorphosis (M) and survival (S) of *S. gigas* larvae, under different culture conditions. Food is indicated as: **a**. Enriched natural cultures of phytoplankton, mainly *Nitzchia* spp., *Skeletonema costatus*, and *Chaetoceros* spp., **b**. *Thalassiosira* spp., **c**. *Tetraselmis* spp., **d**. *Isochrysis* spp., **e**. *Chaetoceros gracilis*, **f**. *Nanochloris*, **g**. *Dunaliella tertiolecta*, **h**. *Platymonas tetraselmis*, **i**. *Prorocentrum minimun*, **j**. *Emilania huxleyi*, **k**. *Heterocapsa pygmacea*.

T°C	Food	Growth µm/day	ML mm	M Days	S %	Source
24-27	H	-	-	60	-	(1)
24-30	A	82*	2.2	28-33	-	(2)
24-30	A	-	-	27-35	-	(3)
28 ± 1	b, c, d, g	-	1.9	28	-	(4)
29	b, d, f, g,	-	1.9	14-35	-	(5)
26-30	A	-	-	18-21	-	(6)
23-31	c, d	-	-	19-30	mortality	(7)
26 ± 1	d, g, i, j, k	13-93	1.2	-	83-96	(8)
27-29	A	-	-	-	-	(9)
28	c, d, c+d	40	1	20	-	(10)
25-31	d, e, c	-	1.1	21-40	-	(11)
28-30	D	24	0.9	Not reached	59 ± 9	(12)
27	a, d	-	1.3	27 ± 2	-	(13)
27-30	d, e	39	1.2	21	-	(14)
29 ± 1	d, c	5-13	0.7	-	21-52	(15)
28	C	-	-	-	-	(16)
28-32	b, c, d	-	-	-	15-20	(17)
29 ± 1	b, c, d,	24-37	0.8	-	25-82	(18)
28-30	d, e		1.3	18-23	-	(19)
26-29	A	-	-	18-30	-	(20)

*Rate calculated for this table from other data in source.

(1) D'Asaro, 1965; (2) Brownell, 1977; (3) Brownell et al., 1977; (4) Siddall, 1981; (5) Davis and Hesse, 1983; (6) Laughlin and Weil, 1983; (7) Corral and Ogawa, 1985; (8) Pillsbury, 1985; (9) Buitrago, 1985; (10) Aldana-A. and Torrentera, 1987; (11) Davis et al., 1987; (12) Heyman et al., 1989; (13) Boidron-M., 1992; (14) Davis et al., 1993; (15) Domíngez, 1993; (16) Aldana-A. et al, 1994; (17) Baqueiro, 1994; (18) García-S. and Aldana-A.; 1994; (19) Davis, 1994; (20) Weil and Laughlin; 1994.

En la Figura 1 se presenta la correlación entre la temperatura y la tasa de crecimiento diaria, con un ajuste polimodal y un coeficiente de correlación de 0.63. La tendencia de estos puntos muestra que la temperatura a la que se obtiene la mejor tasa de crecimiento es a 27°C (60 um por día), luego una zona subóptima con una tasa de crecimiento de 40 um en el rago de temperatura de 26°C y 28°C. Por último una zona de bajo crecimiento a temperaturas superiores a 29°C e inferior a 26°C.

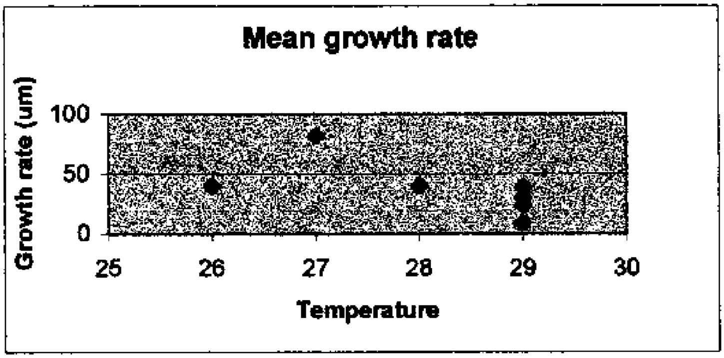


Figura 1. Correlación entre el efecto de la temperatura y la tasa promedio de crecimiento diaria de la longitud de la concha de larvas velígeras de *Strombus gigas*. Los datos para establecer la correlación fueron tomados de los autores citados en la tabla 1.. $Y= -13.039\ x^2 + 709.9x - 9597.2$
$r^2= 0.6313$

La Figura 2 muestra la correlación entre la temperatura y la longitud máxima al momento de la metamorfosis no presento correlación alguna, su mejor ajuste fue el exponencial con un coeficiente de correlación bajo de 0.32.

Por último la correlación entre la temperatura y el tiempo en que la larva alcanza la metamorfosis (Figura 3), presento un buen ajuste exponencial con un coeficiente de correlación de 0.80. Los datos muestran una tendencia a disminuir el tiempo de la metamorfosis de 26°C a 29°C, utilizando menos de 25 días a esta última temperatura. De esta misma tendencia se observa que a temperaturas superiores a 30°C, las larvas se salen de su rango óptimo y vuelven a requerir de mayor tiempo para la metamorfosis.

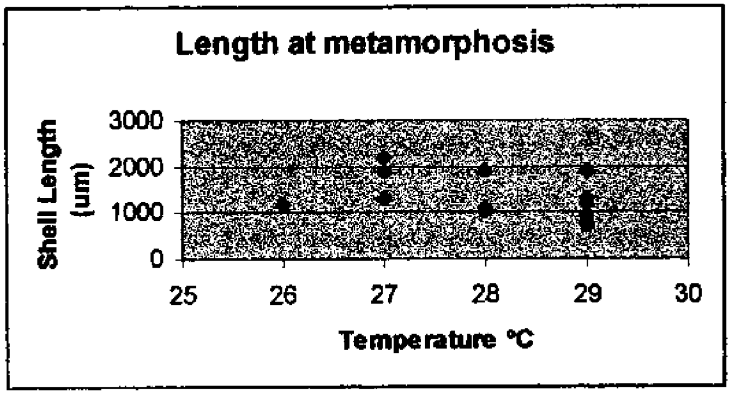


Figura 2. Correlación entre el efecto de la temperatura y la longitud máxima de la concha de larvas velígeras de *Strombus gigas*. Los datos para establecer la correlación fueron tomados de los autores citados en la tabla 1.. $Y= 138.33\ x^2 - 8075x + 1189\ r^2= 0.3195$

Por otra parte, la información obtenida de los cultivos larvarios y de sus resultados en términos de crecimiento o duración de la metamorfosis, no es relacionada con la época del año de la cual proviene la masa de huevos. El único trabajo que trata de establecer una relación en este sentido es Ogawa y Corral (1985) quienes durante tres años y de manera mensual (1983 - 1985), estudiaron cultivos larvarios hasta la metamorfosis, sugiriendo que la frecuencia de éxito de la cría de larvas hasta la metamorfosis depende de la estación del año en la que se efectúa el cultivo, con mejores resultados de junio a septiembre. Los autores obtienen un mayor numero de cultivos que llegan hasta metamorfosis al aumentar la temperatura, pero si esta sigue incrementándose, las pruebas positivas hasta metamorfosis vuelven a bajar. Estos resultados llevan implicito el conceto de la calidad del huevo de los progenitores que ha sido estudiado ampliamente en *Pecten maximus* por Dorange (1990).

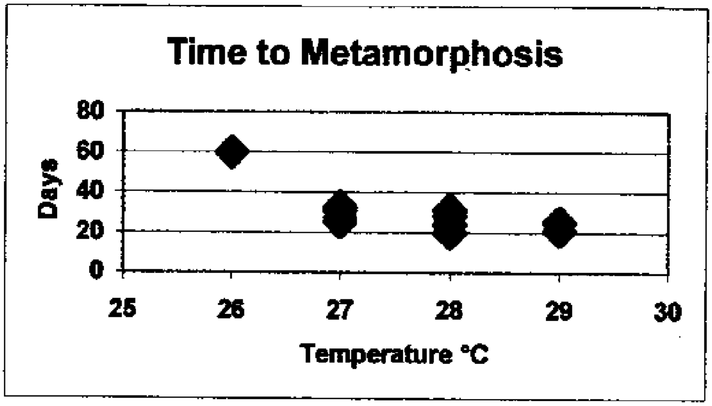


Figura 3. Correlación entre el efecto de la temperatura y el tiempo en que alcanzan la metamorfosis las larvas velígeras de *Strombus gigas*. Los datos para establecer la correlación fueron tomados de los autores citados en la tabla 1. $Y= 8.1548 x^2 - 348.86x + 4964.9$; $r^2= 0.7975$

De Jesús (1999) en un estudio sobre abundancia y distribución de larvas no encontró correlación significativa entre estas y la temperatura, salinidad o el oxígenos. Sólo observó correlación entre estas y la abundancia de juveniles.

Sin embargo, otros autores como Stoner et al. (1992) y Barilé (1994) encuentran correlación entre la abundancia larval y la temperatura y el fotoperiodo. Lo anterior podría estar asociado con la capacidad adaptativa de la larva para acortar el tiempo hacia la metamorfosis, dismimuyendo la probabilidad de depredación en el plancton y evitar la competencia con los miembros del fondo (Stoner 1997, Pechenick 1999).

DISCUSIÓN

Temperatura, Factor Exógeno y su Efecto en las Larvas de Moluscos

La atemperatura es uno de los factores físicos que son conocidos por afectar el crecimiento de las larvas de moluscos, lo mismo que el alimento (Bayne, 1983) La tasa de crecimiento de las larvas se incrementa al increm,entarse la temperatura hasta un optimo para luego declinar a temperaturas más elevadas.

Ursin (1963) describe la relación entre la temperatura y el tiempo de completar una cantidad de crecimiento como una curva catenaria simétrica definida por:

$$Y = yo \cosh p (x - xo)$$

Donde y es el tiempo, x es la temperatura, xo es la temperatura a la cual el desarrollo es mas rápido, yo es el tiempo de desarrollo al tiempo xo y p es el coeficiente de temperatura. Del reciproco de la ecuación anterior 1/y, es la tasa de crecimiento. La simetría alrededor de la temperatura óptima para el crecimiento puede no estar presente, ya que para algunas especies se presenta una transición abrupta desde el optimo hasta una temperatura superior letal (Lucas and Costlow 1979). Loosanoff (1959) para *Mercenaria mercenaria* logra describir una relación lineal entre la temperatura y el número de días desde la fertilización hasta el asentamiento, de la siguiente manera:

$$\text{Días al asentamiento} = 37.9 - T\ (^{o}C)$$

Sin embargo a 15°C y 33°C el crecimiento y desarrollo fueron anormales y la mortalidad fue alta. Ansell (1968) resume datos sobre la tasa de crecimiento de *Mercenaria mercenaria* a través de su rango de distribución geográfica y concluye que la temperatura media optima es de 20°C; lo cual contrasta con el crecimiento de la larva, que tiene su optimo cerca de los 30°C. La explicación de estas observaciones, y su significado ecológico permanece sin explicación. Algo similar se observo en el análisis de los resultados obtenidos en este trabajo con *S. gigas*; donde a 26°C se tiene la mejor sobrevivencia, a 27°C la mejor tasa de crecimiento y a 29°C el mejor resultado para la metamorfosis.

Una evaluación de los efectos de la temperatura sobre el crecimiento larval no es completa, si no se considera el período durante la cual la larva es competente para alcanzar la metamorfosis. Lutz y Jablonskib (1978) han sugerido que una correlacvión negativa entre la temperatura y la longitud de la concha a la que alcanzan la metamorfosis, puede ser de utilidad en paleoclimatología.

En el medio natural las larvas pueden estar expuestas a cambios cortos de temperaturas, sin embargo experimentos del efecto de variaciones de temperatura sobre el crecimiento son escasos.

En larvas veligeras de *Nassarius obsoletus*, Scheltema (1967) obtiene una alta tasa de crecimiento a la temperatura de 25°C. La temperatura más baja a la cual el desarrollo se completo hasta la metamorfosis fue de 16 a 17°. Este autor observo 46% de inhibición en el crecimiento de las larvas entre la temperatura optima y la mínima a la cual el desarrollo es completado. Estos resultados muestran que a bajas temperaturas (< 27°C) se incrementa la tasa de mortalidad, y el atraso del desarrollo de las veligeras. Por otra parte a pesar de que la fuenate de alimento de larvas *S. gigas* son algas unicelulares, cuyo valor nutritivo depende de la digestibilidad de las celulas algales, especialmente de la pared celular, y la producción de toxinas (Lucas 1990). Scheltema (1967) puntualiza que algunos dinoflagelados o diatomeas pueden crecer y sobrevivir igual de bien entre 15 and 30°C, sin embargo tanto el número de celulas como su valor nutritivo puede diferir marcadamente entre estos extremos. Davis y Calabrase (1964) encontraron que *Chlorella* sp. Incrementan su valor nutricional en larvas de almejas y ostiones, cuando la temperatura es incrementada por arriba de 20°C. Estos autores han sugerido que las enzimas necesarias para digerir digerir la pared celular pueden estar activas completamente a altas temperaturas.

La temperatura es un factor que controla el metabolismo del organismo e influye en el crecimientolarval y en la sobrevivencia (Pechenik et al.,1990); por lo que mucha mayor atención debe prestarse a este factor. Este autor con larvas de *Mytilus edulis* y de *M. mercenaria* y *C. Virginica*, encuentra una pobre correspondencia entre el efecto de factores ambientales y la tasa de crecimiento y el desarrollo. Lo anterior es similar a lo que se presenta en este estudi para *S. gigas*. En los estudios de Bayne (1965), el incremento de la temperatura acelera la tasa de crecimiento mas en relación a la tasa de desarrollo.

En parte por las implicaciones ecológicas que tienen el desarrollo de los moluscos (Pechenik 1990), los biólogos han examinado los efectos de la temperatura, la concentración de alimento y otros factores ambientales sobre su desarrollo larval, su sobrevivencia y su crecimiento; a través de la longitud de la concha o en el mejor de los casos con el peso seco (Pechenik 1987). Por lo que respecta a los efectos de los factores ambientales sobre las tasas de diferenciación permanecen no explorados para los moluscos, a pesar de que existe una buena razón para pensar que las tasas de crecimiento y diferenciación pueden ser afectadas en diferentes grados por cualquier cambio ambiental. En la medida que diferenciación anatómica y fisiológica puede presentarse en ausencia de crecimiento. En este sentido larvas de opistobranquios, presentan substanciales cambios anatómicos y desarrollo de competencia para la metamorfosis, posteriores varios días de haber cesado el crecimiento en longitud de la concha. Existen evidencias marcadas de la independencia entre el proceso de crecimiento con la diferenciación durante el desarrollo larvario. Así larvas de *Crepidula fornicata* desarrollsn filamentos branquiales a tallas pequeñas de longitud de la concha cuando se cultivsn a bajas temperaturas (Pechenik y Lima 1984). Si el tiempo requerido para desarrollar los filamentos branquiales ha incrementado en la misma proporción que la tasa de crecimiento de la concha ha decrecido, las branquias se deberan desarrollar mas tarde, pero a la misma talla; las tasas de crecimiento de la concha aparentemente decreciron mas por las bajas temperaturas en relación a la tasa de diferenciación de las branquias, de manera que las larvas desarrollan branquias a tallas en promedio más chicas. De manera similar larvas de *C. fornicata* cultivadas a altas temperaturas crecen mas rapido y alcanzan la metamorfosis a tallas en promedio más chicas (Pechenik 1984), lo que implica un efecto diferencial de la temperature sobre las tasas de crecimiento y diferenciación. El grado de extensión al cual las tasas de crecimiento larval y diferenciación estan unidas no ha sido específicamente estudiada en moluscos. De manera clara este efecto diferencial se presenta en este análisis realizado del efecto de la temperatura sobre el desarrollo y crecimiento larval de *S. gigas*.

CONCLUSIÓN

El efecto de la temperatura no ha sido estudiada sobre el desarrollo larvario de los Strombidos, ni en particular sobre S. gigas. El conocer el rango optimo de temperatura para el desarrollo y crecimiento de una especie, tiene una aplicación en mejorar las ténicas de cultivo para la obtención de semilla, pero que también tiene aplicación en comprender el papel de este factor físico en el proceso de desarrollo larvario, metamorfosis y asentamiento de nuevos reclutas, con la implicación que tiene en la dinámica de posblaciones de un recurso pesquero.

Al conocer el efecto de la temperatura sobre el desarrollo larval, sirve para predecir el grado de éxito que tendrá el proceso de reclutamiento. Esto esta siendo ya aplicado en peces con el modelo de Appeldoorn, que estima la mortalidad natural de éstos en base a la temperatura ambiente media.

Desde el punto de vista ecológico, la temperatura presenta un papel estratégico en la sobrevivencia larval, ya que a mayor temperatura el crecimiento y desarrollo se acelera, disminuyendo la duracion de la vida larvaria y en consecuencia el tiempo que la larva permanece en el plancton, disminuyéndo el riesgo de mortalidad por predacción.

REFERENCES

Aldana-Aranda, D., M.V. Patiño-Suárez, and T Brulé. 1994. Ingestion and digestion of eight unicellular algae by *Strombus gigas* larvae (Mollusca gastropode) studied by epifluorescence microscope. *Aquaculture* **126**:151-158.

Aldana-Aranda, D., A. Lucas, T. Brulé, M. Andrade, E. García-Santaella, N. Maginot, and M. Le Pennec. 1991. Observations on ingestion and digestion of unicellular algae by *Strombus gigas* larvae (Mollusca, Gastropoda) using epifluorescence microscopy. *Aquaculture* **92**:359-366.

Aldana-Aranda, D. and B.L Torrentera. 1987. Croissance larvaire de *Strombus gigas* (Mollusque: Gastéropode) en fontion de la nourriture et de la température. *Haliotis* **16**:403-411.

Appeldoorn, R., 1994. Queen conch management and research: status, needs and priorities. Pages 301-319 in: R.S. Appeldoorn and B. Rodríguez (eds.) *Strombus gigas Queen Conch Biology, Fisheries and Mariculture.* Fundación Científica Los Roques, Caracas, Venezuela.

Baqueiro, C.E. 1997. The Molluscan fisheries of Mexico. Pages 39-49 in: *The Histoy, present condition and future of the molluscan fisheries of North and Central American and Europe.* NOAA. Tech. Rep. NMFS 129 U.S. Dep. of Commerce.

Baqueiro, E. 1994. Cultivo de juveniles del caracol reina, *Strombus gigas,* en Quintana Roo, México. Pages 295-300 in: R.S. Appeldoorn and B. Rodríguez (eds.) *Strombus gigas Queen Conch Biology, Fisheries and Mariculture.* Fundación Científica Los Roques, Caracas, Venezuela.

Boidron-Metairon, I., 1992. A new approach to comparative studies of *Strombus gigas* larvae at the developmental and nutritional levels. *Proc. Gulf Carib. Fish. Inst.* **41**:459-467.

Brownell, W., and J.M. Stevely. 1981. The biology, fisheries and management of the queen conch *Strombus gigas. Mar. Fish. Rev.* **437**:1-12.

Brownell, W.N., 1977. Reproduction, laboratory culture, and growth of *Strombus gigas, S. costatus* and *S. pugilis* in los Roques, Venezuela. *Bull. Mar. Sci.* **27**(4):668-680.

Brownell, W.N., C.J. Jr. Berg, and K.C. Haines. 1977. Fisheries and aquaculture of the conch, *Strombus gigas* in the Caribbean. *FAO Fisheries Report* **200**:59-69.

Buitrago, J., 1985. Crias en cautiverio del huevo al adulto del botuto (*Strombus gigas* L.). Estación de Investigaciones Marinas de Margarita. Fundación La Salle de Ciencias Naturales. Margarita,Venezuela, Vol. III, 29-39.

Corral, J.L. and J. Ogawa. 1985. Cultivo masivas de larvas de caracol *Strombus gigas* en estanques de concreto. *Proc. Gulf Carib. Fish. Inst.* **38**:345-351.

D'Asaro, C.N., 1965. Organogenesis, development and metamorphosis in the Queen Conch, *Strombus gigas*, with notes of breeding habits. *Bull. Mar. Sci.* **15**:359-416.

Davis, H.C. and A. Calabrase. 1964. Combined effects of temperature and salinity on development of eggs and growth of larvae of *Mercenaria mercenaria* and *C. virginica. Fish. Bull.* **63**:643-655.

Davis, M. 1994. Short-term competence in larvae of queen conch *Strombus gigas*: shifts in behavior, morphology, and metamorphic response. *Mar. Ecol. Prog. Ser.* **104**:101-108.

Davis, M. and R.C. Hesse. 1983. Third worl level conch mariculture in the Turks and Caicos Island. *Proc. Gulf Carib. Fish. Inst.* **35**:73-82.

Davis, M. and A. W. Stoner. 1994. Trophic cues induce metamorphosis of queen conch larvae (*Strombus gigas* Linnaeus). *J. Exp. Mar. Biol. Ecol.* **180**:83-102.

Davis, M., R.C. Hesse, and G. Hodgkins. 1987. Commercial hatchery produced queen conch, ***Strombus gigas***, seed for the research and grow-out market. *Proc. Gulf Carib. Fish. Inst.* **38**:326-335.

Davis, M., C.A. Bolton, and A.W. Stoner. 1993. A comparison of larval development, growth, and shell morphology in three Caribbean *Strombus* species. *The Veliger* **36**(3):236-244.

Domínguez, T.O., 1993. Alimentación de larvas de Caracol Rosa (*Strombus gigas*), con dos especies de microalgas (*Tetraselmis suecica* e *Isochrysis* aff. *galbana*). Tesis para obtener el grado de Ingeniero Pesquero en Acuacultura. ITMAR, Boca del Río, Veracruz, México, 97 pp.

García-Santaella, E. and D. Aldana-Aranda. 1994. Effect of algal food and feeding schedule on larval growth and survival rates of the queen conch, *Strombus gigas* (Mollusca, Gastropoda), in Mexico. *Aquaculture* **128**:261-268.

Heyman, W.D., R.A. Dobberteen, L.A. Urry, and A.M. Heyman. 1989. Pilot hatchery for the queen conch, ***Strombus gigas***, shows potential for inexpensive and appropiate technology for larval aquaculture in the Bahamas. *Aquaculture* **77**:277-285.

Laughlin, R.A., and E. Weil. 1983. Queen conch mariculture and restoration in the Archipielago de Los Roques: preliminary results. *Proc. Gulf Carib. Fish. Inst.* **35**:64-72.

Lucas, A., 1990. Feeding and digestion in bivalve larvae. Pages 173-190 in: Morton B. (ed.), Proc of Memorial Symposium in honor of Sir Charles Maurice Yonge. Hong Kong.

Pillsbury, K.S., 1985. The relative food value and biochemical composition of five phytoplankton diets for queen conch *Strombus gigas* (Linné) larvae. *J. Exp. Mar. Biol. Ecol.* **90**:221-231.

Pechenik, J. A., L.S. Eyster, J. Widdows, and B.L. Bayne. 1990. The influence of food concentration and temperature on growth and morphological differentation of blue mussel *Mytilus edulis* L. larvae. *J. Exp. Mar. Biol. Ecol.* **136**:47-46.

Scheltema, R.S. 1967. The relationship of temperature to the larval development of *Nassarius obsoletus* (gastropora). *Biol. Bull.* **129**:340-354.

Siddall, S. E., 1981. Larviculture. Pages 13-23 in: J.C. Berg, Jr. (ed.). Proceedings of the first Queen Conch Fishery and Mariculture Meeting, The Wallace Groves aquaculture Foundation, Freeport, Bahamas.

Weil, E., Laughlin, R. A., 1994. Laboratory culture of *Strombus gigas* L. in the Dos Mosquises Marine Station, Los Roques National Park, Venezuela: Final results. Pages 275-294 in: R.S. Appeldoorn and B. Rodríguez (eds.). *Strombus gigas* Queen Conch Biology, Fisheries and Mariculture. Fundación Científica Los Roques, Caracas, Venezuela.

Use of Light-attracted Zooplankton for Rearing Post-settlement Coral Reef Fish

MAGGIE WATSON, ROBERT POWER, and JOHN L. MUNRO
ICLARM, Caribbean Marine Protected Areas Project
Suite 158, Inland Messenger Service
Road Town, Tortola, British Virgin Islands

KEY WORDS: Juvenile mortality, light attracted plankton, recruitment

ABSTRACT

Settlement stage *Lutjanus synagris* and *Ocyurus chrysurus* were caught in light traps set off fore reefs in the British Virgin Islands and raised in floating mesh cages tethered in water 1 - 2 m deep. Lights were designed to attract plankton into the cages and provide live natural food for the fish. Plankton taxonomic composition around lights was compared with samples from plankton tows in adjacent water, as well as with gut contents of cage reared and wild fish. Fish mortality and growth were followed over the first few weeks of life and compared with results from a field trial assessing natural mortality and growth. Cage rearing reduces the exceptionally high early post-settlement mortality typical of *Ocyurus chrysurus* from approximately 80% to 40% within the first month. We suggest two uses for this technique, a) to help speed recovery of over-exploited stocks within no-take marine reserves b) to rear settlement stage ornamental species until they are large enough to survive on artificial food. Light traps and cage culture in combination could replace destructive fishing practices on reefs exploited for the aquarium trade.

INTRODUCTION

Light attracted plankton has previously been investigated for culture of freshwater pike-perch (*Stizostedion lucioperca)* fry (Schlumpberger and Ziebarth 1981, Jaeger and Nellen 1983, Jaeger et al. 1984, all cited in Hilge and Steffens 1996). This paper reports preliminary investigations into the feasibility of collecting settlement stage reef fish and rearing them past their initial mortality hurdle by using light attracted zooplankton as food. High post settlement mortality is typical of many coral reef fishes. Shulman and Ogden (1987) found >90% natural mortality in French grunts *Haemulon flavolineatum* during the month after settlement, and this rate may be characteristic for species that settle in pulses, and/or form schooling aggregations of juveniles (Roberts 1996, and references therein).

We discusses potential uses for this rearing technology both to enhance recovery of recruitment overfished reef fish stocks within protected areas and to provide a non-destructive alternative income for fishers raising ornamental species for the aquarium trade.

MATERIALS AND METHODS

Settlement stage reef fishes were caught using light traps modified from a design by Stobutzki and Bellwood (1997). The traps consisted of 40 x 40 x 40 cm aluminium 'angle-iron' frames supporting transparent plexiglas panels. On each side of the trap a clear funnel ran the height or width of the cube, (two vertical and two horizontal) each with an opening of 12mm. The light was a 'stand alone' system inserted into the top of the main trap. It consisted of an O'ring sealed plastic box, housing two six volt rechargeable batteries and connected to a clear plastic tube containing an 8 watt fluorescent bulb. A light sensitive switch mounted on the top of each light box turned the trap on at dusk and off at dawn. An advantage of this modular design is that changing batteries and mending equipment did not require the entire light trap to be taken ashore. Buckets set beneath the main trap retained the fish as traps were hauled. Traps were set at a depth of 2 m, 100 - 200 m offshore of three fringing reefs on Tortola, British Virgin Islands around new moon periods in 1998 and 1999. Commercially important yellowtail and lane snappers (*Ocyurus chrysurus*, *Lutjanus synagris*) caught in light traps were reared in floating cages (125 L, mesh size 3 mm) tethered in 1 - 2 m of water for approximately one month to investigate the feasibility of rearing reef fish on light attracted plankton. At night, each cage was lit with either an 8 watt light module from the light traps or a 9 watt mains powered light sealed with silicone into plastic drinks bottles and appropriately fused. All cages were regularly scrubbed of algae. The order and variety of experiments described below were unavoidably constrained by unpredictable availability of fishes in monthly settlement pulses over the recruitment season.

Early Mortality in Caged and Wild Fish

Survival of *Lutjanus synagris* reared under two different stress regimes was examined to isolate the effect of handling stress on survival. For two cages (both with 45 fish initially) handling stress at first capture from the light trap was minimised by not touching the fish and by keeping them in extra large aerated buckets until placing them in the cage as the earliest opportunity. One group was counted and measured every week, while the other was sampled only fortnightly. *L synagris* reared in two other cages (with initial numbers of 35 and 40) received no special treatment at capture and were both sampled weekly. Survival of *O. chrysurus* in two cages (initial n = 35 and 63) was recorded weekly to investigate density dependence in mortality.

As part of a parallel study, early juvenile natural mortality of *O. chrysurus* was estimated from daily censuses of an area of seagrass 25 x 50 x 1-1.5 m in depth. The area was divided into 5 x 5 m squares. One observer swum concentrically around each individual section until the entire area of each square was covered. Each census took a total of 2 - 2.5 hours. Daily censuses were performed during the 11 days leading up to peak settlement in September 1999 and for six days following. Over the next nine days a different observer conducted censuses. Resumption of observations by the original observer identified bias in numbers of fish observed (but not the proportion of size classes – see section on growth below) and nine days data were dropped from the analysis of mortality. Observations then continued every other day for a further 11 days until the pulse of settlement became indistinguishable from other juveniles. Temporal patterns in abundance were assumed proportional to the processes of settlement and natural mortality (as in McGehee 1995). We also assume disappearance of fish represented mortality rather than movement, and that any emigration from the census area would be balanced by immigration. We base this assumption on the enormous size of the census area relative to observed home ranges of settlement stage *O. chrysurus (*Watson and Gell in prep and see discussion*)*, and on results from a preliminary mark recapture experiment which found elastomer tagged *O. chrysurus* moved only a few metres after settlement (Watson and Gell in prep). The day when the number of settlement size fish peaked (91 compared to only four 10 days earlier) was assumed to represent the settlement event, and numbers of fish remaining over time were expressed as a percentage of that number for comparison with mortality in cage reared fish.

Estimates of Early Juvenile Growth for *L. synagris* and *O. chrysurus*, and Comparison of *O. chrysurus* Growth in Cage Reared and a Natural Population

Early juvenile growth was estimated from weekly measurements of two groups of *O. chrysurus* (n = 35 and 63) and three groups of *L. synagris* as in the mortality section above (n = 45 'low stress', n = 35 and n = 40). Data from the 'low stress' trial measured fortnightly were excluded as there were only two data points. Total lengths of wild fish from the September settlement peak were estimated (to the nearest 0.5 cm) during daily censuses of the seagrass grid until the cohort could no longer be clearly distinguished (about 25 days). Modal length from field observations was plotted against modal length of *O. chrysurus* (grouped to the nearest 0.5 cm) reared in two cages in order to compare growth of captured and wild fish.

Food Availability and Feeding Preferences

Zooplankton was sampled from water adjacent to floating cages using a mini-plankton net (30 cm diameter aperture) towed at approximately one knot for five minutes. Zooplankton attracted into cages was sampled with two swipes of a dip-net in an empty cage every five minutes over a 25 minute period. Gut contents were analysed for seventeen cage-reared *O. chrysurus* (total length 20-40mm) caught at night. Gut contents were also analysed for ten wild *O. chrysurus* (22 - 50 mm total length) caught during the day. Percent composition by volume in the water samples and in fish guts (individuals within each sample pooled) were compared to examine selective feeding behaviour.

RESULTS

Early Mortality in Caged and Wild Fish

Survival of *L. synagris* was greater in 'low stress' higher density cages (n = 45) than in 'high stress' lower density cages (n = 35 and n = 40) (Figure 1.). Two cages where initial handling stress was minimized had approximately 80% survival after three weeks, compared to cages stocked at lower initial densities (23% and 12% less fish) where survival was around 50% within two weeks. After initial capture, weekly measuring did not appear to have a marked affect on survival. Thus a large part of early mortality of *L. synagris* (and probably *O. chrysurus*) in cages appears to be due to initial handling stress.

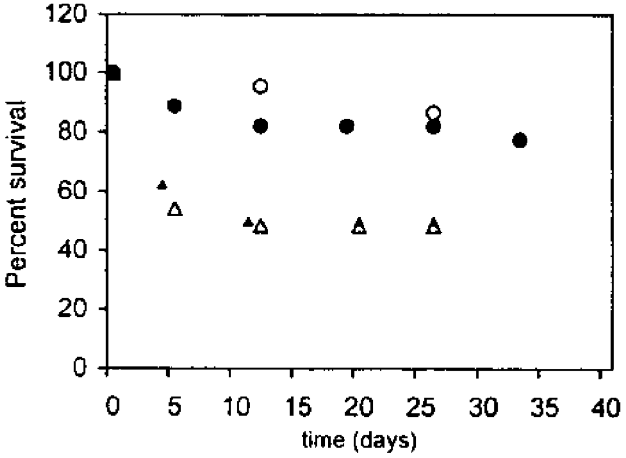


Figure 1. Percent survival of *Lutjanus synagris* against time in cages for four rearing regmines; triangle, initial n = 40; no special treatment; open triangle initial n = 35, no special treatment; circle initial n = 45, 'low stress' regime, sampled weekly; open initial n = 45, 'low stress' regime sampled fortnightly

Survival of *O. chrysurus* at initial densities of n=35 and n=63 over approximately one month were very similar (Figure 2) suggesting that mortality is not density dependent up to at least 0.5 fish per litre. Survival in these two cages was approximately 30% higher than estimates of survival from wild fish after one month (Figure 2). Daily censuses estimated mortality of *O. chrysurus* to be 80% within one month in the wild. Cage reared fish had a mortality of approximately 30-40% over the same time period. Mortality in all three groups fitted exponential curves (see equations on the graphs), i.e. 'type three' mortality. The curve fitted through the wild population was not forced through an intercept of 100% since the actual number of settlers is not known because settlement took place over a number of days during which mortality was already operating.

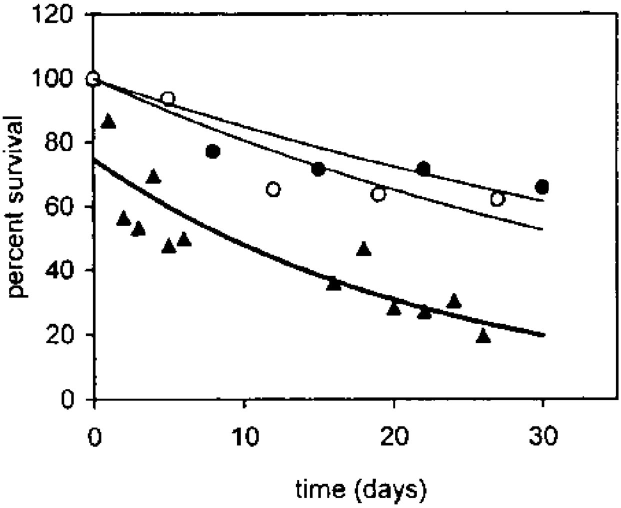


Figure 2. Percent survival against time for *Ocyurus chrysurus* < 1 month since settlement and reared in floating cages (closed circles initial n = 35, $y = 100e^{-0.0162x}$, $r^2 = 0.71$, open circles initial n = 63, $y = 100e^{-0.0125x}$, $r^2 = 0.80$) compared with percent survival in the wild estimated from daily field censuses of one monthly cohort (triangles, $y = 74.831e^{-0.0445x}$, $r^2 = 0.83$)

Comparison of Growth in Natural and Cage Reared Fish.

Early juvenile growth in cages was linear for both *O. chrysurus* and *L. synagris* and shows no signs of density dependence between 0.28 and 0.5 fish per litre (Figure 3 and Figure 4.). Modal length of *O. chrysurus* in cages was not significantly difference to modal length of the September 1999 cohort censused in the wild over the first month (Figure 5).

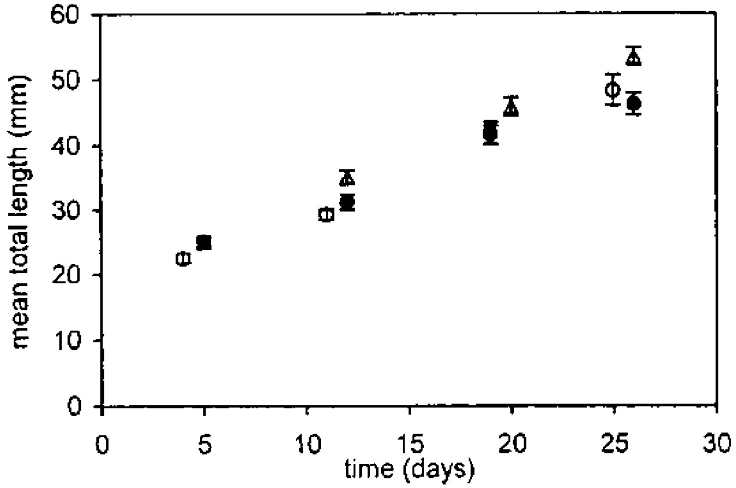


Figure 3. Mean total length (mm) of *Lutjanus synagris* against time in cages for three rearing regimes; closed circle initial, n = 45, 'low stress' y = 1.05*X + 19.7, r^2 = 0.98; triangle initial, n = 35, 'high stress' y = 1.35*X + 18.5, r^2 = 0.99; open circle initial n = 40, 'high stress' y = 1.275*X + 16.72, r^2 = 0.99. Error bars represent 95% confidence levels.

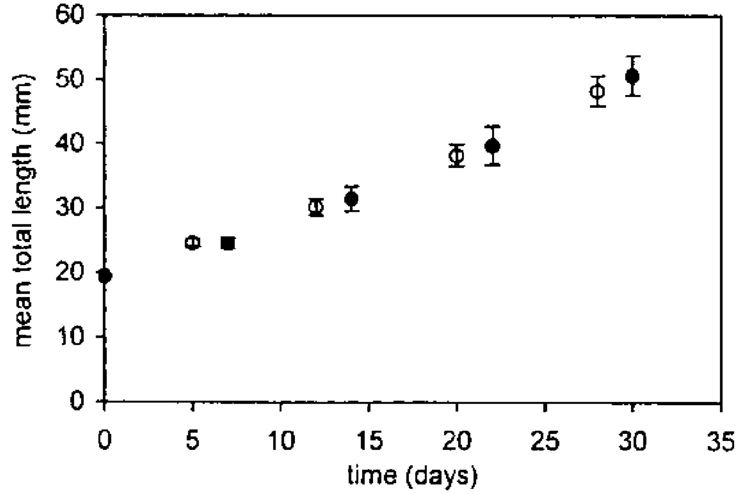


Figure 4. Mean total length (mm) of *Ocyurus chrysurus* against days in cage for two groups; closed circles initial n = 35, y = 1.0383*X = 17.99, r^2 = 0.99; open circles initial n = 63, y = 1.0081*X + 19.01, r^2 = 0.99. Error bars represent 95% confidence levels

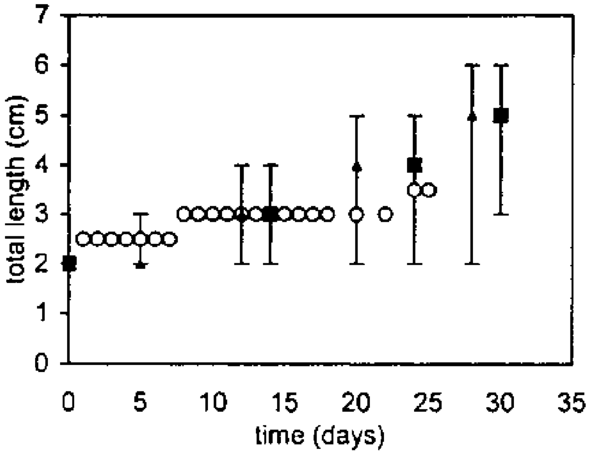


Figure 5. Modal total lengths of *Ocyurus chrysurus* (less than one month since settlement) estimated to the nearest 0.5 cm from field observations (open circles) and two cage culture experiments (square and triangle) against time in days from peak settlement and from date of capture in light traps, respectively. Error bars for caged fish indicate size range.

Food availability and feeding preferences

Zooplankton samples taken from tows in water adjacent to floating cages and from dip-net sweeps inside lighted cages show zooplankton attracted to the lights is similar to that in open water (Figure 6). Both wild and cage reared *O. chrysurus* fed selectively, and included fish (mostly juvenile 'bait fish' - Atherinidae and Clupeidae) in their diet (34% and 13% respectively). These fish, which are attracted to lights, avoided dip-nets and the plankton tow. Malacostraca make up very similar volumes in the diet of both wild and caged fish (43% and 40% respectively). Caged fish also ate annelid worms (23%) attracted up off the bottom by the light. Combined with growth estimates, the data suggest zooplankton provides an adequate food source comparable with a natural diet for early juvenile snappers.

DISCUSSION

Our results demonstrate that the 'low tech', low maintenance technique of rearing reef fish using light attracted plankton supports growth at least equal to that in the wild, and substantially reduces mortality over the first month. We suggest two potential uses of this technique. In coral reef areas where recruitment over-fishing has diminished the supply of new recruits, recovery of fish populations may be extremely slow even if fishing ceases. For example, identical studies comparing reef fish settlement to the heavily exploited reefs on the north Jamaican shelf, and to the moderately exploited reefs in the British Virgin Islands found orders of magnitude differences in settlement rates (Munro and Watson 1998; Munro and Watson 1999), implying that stock recovery will take many years. If reefs are self-seeding recovery may not occur without intervention. Light traps may be a means of catching juvenile reef fish for stock enhancement (Doherty 1994). Using the techniques described here, settlement stage fish could be collected from a relatively large area, reared for several weeks, and concentrated within a marine protected area.

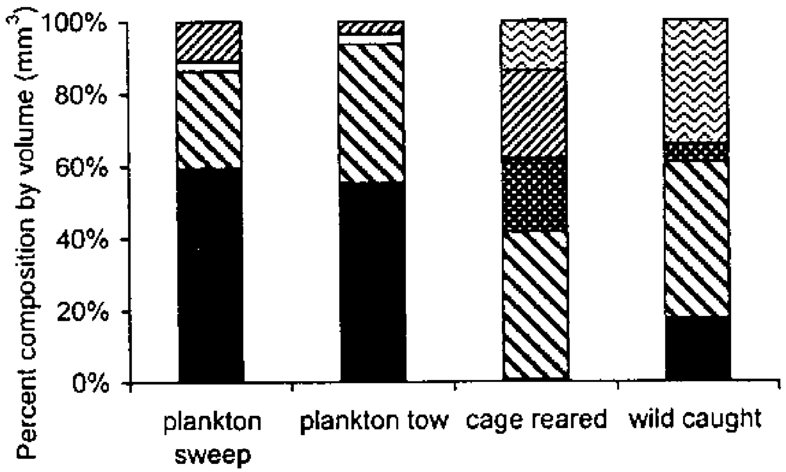


Figure 6. Percent composition by volume for plankton sweeps inside floating cages, a plankton tow in water adjacent to floating cages, and cages reared (n = 17, total length 20 - 40 mm) and wild (n = 10, total length 25 - 53 mm) *Ocyurus chrysurus*. Key: black = Copepoda, grey = Ostracoda, wide hatching = Malacostraca, thin hatchin = Annelida, wavyline = fish, dotted = Crustacean debris

This technology is intended for heavily exploited populations where even a slight increase in juvenile survival from small releases would be valuable. Stock enhancement has been heavily critizised as uneconomic (Hilborn 1998), inappropriate (Scarnecchia 1988, Meffe,1992), difficult to evaluate (Leber et al. 1996, Hilborn 1998), and potentially harmful to wild populations through introduction of disease or through genetic selection for inappropriate characteristics (Schramm and Piper 1995). However, most enhancement efforts have been large scale, expensive 'high tech.' programs raising up to several million fish from eggs to fingerlings. The present approach is low cost, low maintenance, and rears only wild caught stocks on site. Fish are fed with natural food before release to an unfished native habitat. The methods might be suitable for small scale marine protected areas projects in developing countries were coral reef fisheries resources are under the greatest threat.

In this study, cage rearing had the potential to decrease mortality in the first month after settlement from approximately 80% to around 40%. This estimate is probably conservative due to inaccuracies in the censuses of wild fish. It was not possible to measure the true natural settlement rates both because visual observations may miss many fish and because numbers of settling fish increased rapidly over several days and mortality before the settlement peaked was ignored in our analysis. For many reef fishes, mortality is thought to be highest soon after settlement (Victor 1986, Doherty and Sale 1986, Sale and Ferrell 1988). Furthermore, we compared natural mortality with *O. chrysurus* reared in cages before development of faster, low stress handling techniques at capture. Results from *L. synagris* suggest cage mortality can be reduced to around 20% with careful handling.

Perhaps one of the most important criticisms of stock enhancement is that artificially reared fish lack the behaviour necessary to ensure survival in the wild. A parallel study of 'early juvenile' snapper behaviour in seagrasses (Watson and Gell in prep) found that recently settled *O. chrysurus* and *L. synagris* (approximately 2 - 3 cm) remain almost stationary near the bottom and pick plankton from the water column. They rarely move more than 20cm from their initial position and tolerate conspecifics of the same size. However, from approximately 3 - 3.5 cm their range becomes considerably bigger (several square metres), they switch to feeding on benthic invertebrates and act aggressively towards similarly sized conspecifics. Thus cage rearing is probably most suited to rearing fish to approximately 3 - 3.5 cm (3 - 4 weeks).

The present study, which was designed to test the feasibility of the technology, has not addressed the question of whether cage reared fish subsequently released have a higher overall survival than wild fish from the same cohort. High mortality in the first few days after settlement (or release) may be due to unfamiliarity with the habitat. However, rearing in semi-natural environments (Masuda and Tsukamoto 1998 and references therein) and conditioning to the habitat (Olla et al. 1994) have been shown to reduce post release mortality. An evaluation phase is planned for 2000. Despite his many criticisms, Hilborn (1998) concedes that stock enhancement programs are most likely to succeed where wild stocks are essentially gone. This is the case for commercially fished stocks on many overexploited reefs. Even where high mortality makes cost per surviving individuals substantial, the value should be considered in terms of survivor reproductive potential rather than the contribution to stock numbers (Stoner and Glazer 1998).

The second application we suggest for rearing fish on light attracted plankton is to supply ornamental reef fish to the aquarium trade. Widespread use of destructive collecting techniques, such as sodium cyanide, often leads to high mortality in captured fish, does substantial harm to the reef ecosystem, and has made development of sustainable collection techniques an international concern. However, growing consumer demand for 'eco-labled' fish suggests sustainable aquarium fisheries could provide a valuable alternative income for fishers, particularly where overfishing is currently degrading coral reef resources. We suggest fish could be reared on light attracted plankton from late pelagic/settlement size until they are big enough to thrive on artificial food. Catching fish before they reach the 'wall of mouths (Kaufman et al. 1992) on the reef provides an opportunity to avoid high post-settlment mortality. Where natural populations are already overexploited, a proportion of the fish caught for the aquarium trade could also be grown past their initial 'mortality hurdle' and used to enhance natural populations

LITERATURE CITED

Doherty, P.J. 1994. A potential role for light-traps in enhancement of coral reef fisheries ICLARM Conf. Proc., Manila. **44**:92-93.

Doherty, P.J. and P.J. Sale. 1986. Predation on juvenile coral reef fishes: an exclusion experiment. *Coral Reefs* **4**:225-234.

Hilborn, R. 1998. The economic performance of marine stock enhancement projects. *Bull. Mar. Sci.* **62**: 661-674.

Hilge, V. and W. Steffens. 1996. Aquaculture of fry and fingerling of pike-perch (Stizostedion lucioperca L.) - a short review. *J. Appl. Ichthyol.* **12**:167-170.

Jaeger, T. and W. Nellen. 1983. Die aufzucht von fischlarven in beleuchteten netzgehegen. *Fischwirt* **33**:38.

Jaeger, T., W. Nellen, and H. Soll. 1984. Beleuchtete netzgehegeanlagen zur aufzucht von fishbrut bis zur setzlingsgröße. Eine bauanleitung und aufzuchtbeschreibung. *Ber. Inst. Meereskd.* Christian-Albrechts-Univ. Kiel **126**:72.

Kaufman, L., J. Ebersole, J. Beets, and C.C. Mcivor. 1992. A key phase in the recruitment dynamics of coral reef fishes: post settlement transition. *Environmental Biology of Fishes* **34**:109-118.

Leber, K.M., S.M. Arce, D.A. Sterritt, and N.P. Brennan. 1996. Marine stock-enhancement potential in nursery habitats of striped mullet, *Mugil cephalus* in Hawaii. *Fishery Bulletin* **94**:452-471.

Masuda, R. and K. Tsukamoto. 1998. Stock enhancement in Japan: review and perspective. *Bull. Mar. Sci.* **62**:337-358.

McGehee, M.A. 1995. Juvenile settlement, survivorship and in situ growth rates of four species of Caribbean damselfish in the genus *Stegastes*. *Environmental Biology of Fishes* **44**:393-401.

Meffe, G.K. 1992. Techno-arrogance and halfway technologies: salmon hatcheries on the Pacific coast of North America. *Conservation Biology* **6**:350-354.

Munro, J.L. and M. Watson, 1998. Comparative settlement and recruitment rates of coral reef fishes in moderately-exploited and over-exploited Caribbean coral reef ecosystems. International Society for Reef Studies. European Meeting. Perpignan, France September 1998. Unpubl. MS.

Munro, J.L. and M. Watson. 1999. *The role of marine protected areas in fisheries management and biodiversity conservation in coral reef ecosystems*, pp. 47. ICLARM Caribbean Marine Protected Areas Project. Phase 1. Technical report. ICLARM, Road Town, British Virgin Islands.

Olla, B.L., M.W. Davis, and C.LH. Ryer. 1994. Behavioural deficits in hatchery-reared fish: potential effects on survival following release. *Aquacult. Fish. Manage.* **25**:19-34.

Roberts, C.M. 1996. Settlement and beyond: population regulation and community structure of reef fishes. Pages 85-112 in: N.V.C. Polunin and C.M. Roberts (eds.) *Reef Fisheries*. Chapman and Hall, London.

Sale, P.F. and D.J. Ferrell. 1988. Early survivorship of juvenile coral reef fishes. *Coral Reefs* **7**:117-124

Scarnecchia, D.L. 1988. Salmon management and the search for values. *Canadian Journal of Fisheries and Aquatic Science* **45**:2042-2050.

Schlumpberger, W. and G. Ziebarth. 1981. Produktion von vorgestreckten zandern in beleuchteten gazekäfigen. *Z. Binnefisch. DRR* **28**:143-144.

Schramm, H.L.J. and R.G. Piper. 1995. Uses and effects of cultured fishes in aquatic ecosystems. In *American Fisheries Society Symposium 15*. 608 pp.

Shulman, M.J. and J.C. Ogden. 1987. What controls tropical reef fish populations: recruitment or benthic mortality? An example in the

Caribbean reef fish *Haemulon flavolineatum*. *Mar. Ecol. Prog. Ser.* **39**:233-242.

Stobutzki, I.C. and D.R. Bellwood. 1997. Sustained swimming abilities of the late pelagic stages of coral reef fishes. *Mar. Ecol. Prog. Ser.* **149**:35-41.

Stoner, A.W. and R.A. Glazer. 1998. Variation in natural mortality: implications for Queen conch stock enhancement. *Bull. Mar. Sci.* **62**: 427-442.

Victor, B.C. 1986. Larval settlement and juvenile mortality in a recruitment-limited coral reef fish population. *Ecol. Mongogr.* **56**:145-160.

Watson, M. and F. Gell. Settlement, early mortality and juvenile behaviour of the yellowtail snapper, *Ocyurus chrysurus*. In prep.

Mitochondrial Control Region of Striped Mullet, *Mugil cephalus*: A Tool to Restore Marine Fisheries Resources

NIKOLA M. GARBER, AMBER F. GARBER, KENNETH C. STUCK and WALTER D. GRATER
The University of Southern Mississippi, Institute of Marine Sciences
Gulf Coast Research Laboratory
P.O. Box 7000
Ocean Springs, Mississippi 39566-7000 USA

ABSTRACT

Mitochondrial DNA (mtDNA) has been frequently used as a molecular marker in fisheries stock assessment and has potential applications in stock enhancement. In some fish species, sequence data from the mtDNA control region exhibits enough intraspecific variability that the likelihood of two unrelated individuals being identical is very low to nil. Efforts to restore the depleted natural stocks of the striped mullet, *Mugil cephalus*, have been undertaken in the Pacific, through cultured fish release. In such enhancement programs the use of molecular markers, particularly during the initial pilot-scale releases, is a reliable method to estimate contributions of cultured fish to wild stocks. The purpose of this study was to create primers in the flanking tRNAs to amplify the mtDNA control region in *M. cephalus* as a prerequisite to then conduct population genetic studies supporting stock enhancement activities. Further study of the control region of *M. cephalus* will allow separation of individual stocks and the possible utilization of molecular tags to identify genetic contributions of hatchery-reared individuals to the wild population in stock enhancement activities in the Gulf of Mexico and Caribbean.

KEY WORDS: *Mugil cephalus*, control region, mitochondrial DNA

INTRODUCTION

The steadily increasing demand for seafood in the United States, together with technological improvements in harvesting methods and natural events have resulted in depletion of marine fisheries resources. About half of the commercially important marine fishes in the United States are currently being harvested beyond their maximum sustainable yield, leading to a steady decline in abundance (National Marine Fisheries Service 1997, Blankenship and Leber 1995). This trend will likely continue into the future as a result of projected worldwide increases in human population, particularly in developing countries (National Marine Fisheries Service 1997).

Restoration of depleted stocks is possible through effective traditional management procedures designed to reduce fish mortality, reduce fishing effort, and restore habitat. Since overfishing of most commercial stocks is inevitable

(Grimes 1998), traditional management procedures, along with non-traditional methods, such as stock enhancement, should be considered before stocks are significantly depleted.

Stock enhancement has been utilized in varying degrees from the mid-1880s to present time in the United States, Canada, France, Australia, United Kingdom, New Zealand, and Norway (Grimes 1998). Early attempts at marine stock enhancement often involved the indiscriminate release of eggs and newly hatched larvae; most of which were considered failures and in some instances were thought to have caused deleterious effects on existing stocks (Grimes 1998). Blankenship and Leber (1995) proposed a more responsible approach to marine stock enhancement which prioritizes and selects a target species, develops stock rebuilding goals, and uses genetic resource management to avoid deleterious genetic effects. Baseline genetic data is necessary to direct restocking efforts and minimize the negative genetic impacts caused by accidental or intentional inbreeding (Hinder et al. 1991, Blankenship and Leber 1995).

The primary objective of this study is to develop the necessary technical procedures for using mtDNA, specifically the control region, as a molecular marker to support stock enhancement activities through the United States Gulf of Mexico Marine Stock Enhancement Consortium Program (Consortium). The goals of this program are the development, refinement, field testing, and demonstration of a successful marine fishery stock management program for the U.S. coastal Gulf of Mexico that blends stock enhancement technology with traditional fishery management practices. When the identified constraints are resolved, the technology will be transferred to local and regional resource management agencies for their consideration as an additional management tool to supplement, and help replenish certain high-priority, declining stocks of marine fishes. The striped mullet, *Mugil cephalus*, was used as a test species in this study because of its initial designation as a prototype target species by the Consortium. These procedures will be used in future papers to discuss the characterization of the control region, as well as facilitation of a population genetic study of *M. cephalus* from three ocean basins. The procedures described in this study will then be adapted to other target species as designated by the Consortium.

MATERIALS AND METHODS

Mugil cephalus was collected from Biloxi Bay, Mississippi. White muscle tissue was excised and placed in SED buffer (250 mM EDTA, pH 7.5, 20% DMSO, 3.42 M NaCl; weight to volume ~ 2 - 3 grams to 16 ml). *M. cephalus* was identified using fish keys by Hoese and Moore (1977), Murdy (1983), and Robins and Ray (1986), and taxonomic specialists (Jim Franks, Stuart Poss, and Richard Waller) at the Gulf Coast Research Laboratory, Ocean Springs, MS.

Total genomic DNA was extracted from each sample, under sterile conditions, using a procedure modified from Taggart et al. (1992). DNA was then quantified using fluorescence spectrophotometry, as described in Gallagher (1994). Samples were adjusted to a concentration of 100 ng/μl in TE (10 mM Tris, 1 mM EDTA, pH 8.0) prior to storage at -20°C.

Portions of cytochrome b, 12S rRNA, and the control region, and the entire tRNA threonine (tRNA-T) and proline (tRNA-P) were amplified by PCR using universal primers CB3R (5' CACATTCAACCAGAATGATATTT 3'; Palumbi, 1996; Fig. 1) and 12SA-H1067, referred to in the present study as 12SAR, (5' ATAATAGGGTATCTAATCCTAGTT 3'; Martin et al., 1992; Figure 1). PCR amplification was performed in replicate 25 μl reactions containing 100 ng template DNA, 1.5 mM $MgCl_2$, 200 μM deoxynucleotide triphosphates (Promega, Inc.), 0.3 μM of each primer, and 1.75 units *Taq* DNA polymerase with 10X PCR buffer (Amersham Life Science). PCR cycling parameters were 3 min at 94°C, followed by 35 cycles consisting of a denaturing of 0.75 min at 94°C, an annealing of 1 min at 55°C, and an extension of 2 min at 72°C, with a final elongation of 7 min at 72°C. After visualization on an agarose gel, the appropriate PCR product was excised, purified using the QIAquick Gel Extraction Kit (Qiagen, Inc.), quantified, and direct sequenced. Species-specific primers in the genes coding for tRNA-P (MulPro, 5' CCAAGGCCAGGATTTTTACGTT 3'; Figure 1) and 12S rRNA (Mul12S, 5' CACGAGATTTACCGGCCCTATTAG 3'; Figure 1) were then designed.

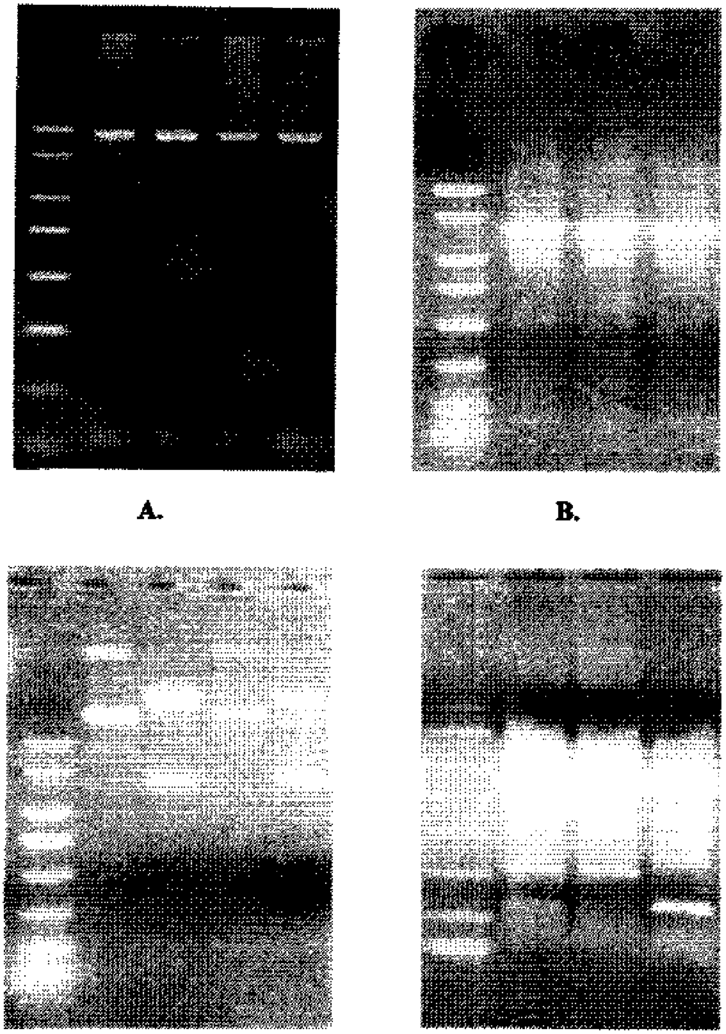
These primers, MulPro and Mul12S, were then utilized to produce a PCR fragment using the previously described cycling conditions. Purified PCR products were cloned using the pGEM®-T Easy Vector System (Promega, Inc.). Ligated vector DNA was transformed into competent JM109 cells that were then cultured on Luria-Bertani (LB)/ampicillin plates with x-gal and IPTG. Colonies containing inserts were identified by blue/white selection and used to inoculate 5 ml minipreps. The cloned plasmid DNA was isolated using the Wizard® *Plus* DNA Purification System (Promega, Inc.). Plasmid DNA was then purified using the PEG method (Nicoletti and Condorelli 1993), quantified, and sequenced.

A species-specific primer in the tRNA-phenylalanine (tRNA-F; MulPhe, 5' TCTTGACATCTTCAGCGTCGC 3'; Figure 1) was subsequently designed and used with the primer MulPro to amplify the entire control region with the above PCR cycling parameters. The appropriate PCR-product was gel-purified, quantified, and direct sequenced. All DNA sequencing was done with an ABI model 373A automated sequencer at the University of Maine DNA Sequencing Facility.

Figure 1. Relative positions of primers used to amplify the mitochondrial DNA control region and flanking genes of *Mugil cephalus*. Abbreviations of the tRNAs are threonine (tRNA-T), proline (tRNA-P, and phenylalanine (tRNA-F).

RESULTS

Total genomic DNA was extracted, and a piece of DNA approximately 2000 base pairs (bp) in length was produced by amplification with the universal primers CB3R and 12SAR (Figure 2a). Approximately 640 bp of sequence was obtained from the CB3R priming site and 400 bp of sequence from the 12SAR site. Results using the NCBI's BLAST WWW Server (Basic Local Alignment Search Tool; Altschul et al. 1990) on the resulting sequences indicated the products obtained closely matched portions of the 12S rRNA, control region, and cytochrome b, as well as tRNA-T and tRNA-P. This sequence information was used to design species-specific primers located in the tRNA-P (MulPro) and in the 12S rRNA (Mul12S) located approximately 180 bp downstream of the 12SAR site. The species-specific primers, MulPro and Mul12S, produced a 1,300 bp PCR-product (Figure 2b). This DNA fragment was subsequently gel purified, T/A cloned (Figure 2c), and completely sequenced, producing the entire control region, tRNA-F, and a partial 12S rRNA. A species-specific primer located in the tRNA-F (MulPhe) was designed from this available sequence data. Finally, MulPro and MulPhe primers produced a DNA-fragment of approximately 880 bp (Figure 2d), which contained the entire control region. BLAST searches identified these sequences as the mtDNA control region and it was most homologous with *Xiphias gladius* (swordfish). The entire 1694 bp sequence is deposited in GenBank, accession number AF108270.



A. B.

C. D.

Figure 2. PCR products of *Mugil cephalus* mitochondrial control region and flanking genes produced, using the primers: (A) CB3R/12SAR; (B) MulPro/Mul12S; (C) MulPro/Mul12S, Lanes 2 and 4 contain uncut clones, Lanes 3 and 5 contain clones from which the insert was cut out with EcoRI enzyme; (D) MulPro/MulPhe (see Materials and Methods for details). Lane 1 in all gels is the ladder of sizes 2000, 1500, 1000, 750, 500, 300, 150, and 50 base pairs (Amresco PCR marker, Amresco, Inc.).

DISCUSSION

The purpose of this study, providing the needed technical background for the eventual development of genetic tags to support stock enhancement activities in the Gulf of Mexico and Caribbean, was accomplished by sequencing the portion of the mtDNA between the cytochrome b and 12S rRNA. It was necessary to create species-specific primers in the flanking tRNAs because tRNAs are conserved within species and amplification of the smallest sequence, including the entire control region, is needed to obtain the greatest overlap in the 5' and 3' sequences. Further utilization of these primers with an increased number of samples allowed for characterization of the control region and a population genetic study of *M. cephalus* between three geographically distant populations: Gulf of Mexico, Atlantic, and Pacific Basins (Garber 1999). This thesis research determined the genetic differences within the Gulf of Mexico and between the three Basins, as well as determining if these differences constituted genetically distinct populations. The sequences could then also be utilized to evaluate whether or not a partial or entire control region can be used as a genetic tag.

Stock enhancement is considered a non-traditional option for restoration of marine fisheries, but it may become an extremely useful alternative management tool in the near future. Genetic considerations need to be defined and maintained, such as: identifying the genetic risks and consequences of enhancement, defining an enhancement strategy with adequate genetic controls, implementing those genetic controls in the hatchery, and monitoring and evaluating effects/ impacts of hatchery-reared fish on wild stocks. Utilizing mtDNA, specifically the control region, may be extremely useful in assessing adverse impacts of releases and possibly provide solutions necessary to circumvent many of these problems before they occur. A population-genetics study using control region sequence data (Rocha-Olivares et al. submitted) as well as a complete characterization of the control region (Garber et al. in preparation) will be presented elsewhere. Appropriate genetic monitoring will result in preservation of fisheries resources and will protect the natural genetic population structure of marine species worldwide. If our fishery stocks are to ever recover, fisheries management entities must work together to combine the available biological, technological, economical, and political information in an effort to identify the best possible solutions.

ACKNOWLEDGMENTS

A special appreciation is extended to Jim Franks for his extra time, effort, and direction in everyday questions and problems encountered during this research. Immense gratitude for editing assistance is given to Axayacatl Rocha-Olivares of Louisiana State University, Baton Rouge, Louisiana. Identifications were confirmed by Jim Franks, Stuart Poss, and Dick Waller to whom we are grateful. A personal thanks is given to David Geter for his technical computer expertise.

Funding was provided by National Marine Fisheries Service in cooperative agreement with the National Oceanic and Atmospheric Administration, U.S. Department of Commerce grant, number NA76FL0446. Research was conducted under the U.S. Gulf of Mexico Marine Stock Enhancement Consortium, a cooperative effort of the Gulf Coast Research Laboratory (Ocean Springs, Mississippi), the Mote Marine Laboratory (Sarasota, Florida), and the Oceanic Institute (Honolulu, Hawaii).

LITERATURE CITED

Altschul, S.F., W. Gish, W. Miller, E.W. Myers and D.J. Lipman. 1990. Basic local alignment search tool. *J. Mol. Biol.* **215**:403-410.

Blankenship, H.L. and K.M. Leber. 1995. The responsible approach to marine stock enhancement. *Am. Fish. Soc. Symp.* **15**:167-175.

Gallagher, S.R. 1994. Commonly used biochemical techniques, Appendix 3. Pages A.3.10-A.3.11 in: K. Janssen, (series ed.) *Current Protocols in Molecular Biology*. John Wiley and Sons, Inc., USA.

Garber, N.M. 1999. Application of the mitochondrial DNA control region in population structure studies of *Mugil cephalus* (striped mullet) in North America. M.S. Thesis. The University of Southern Mississippi, Hattiesburg, MS. 86 pp.

Garber, N.M., A. Rocha-Olivares, A.F. Garber and K.C. Stuck. Complete characterization of *Mugil cephalus* (striped mullet) mitochondrial DNA control region in the Atlantic, Gulf of Mexico, and Pacific Ocean Basins. In prep.

Grimes, C.B. 1998. Marine stock enhancement: sound management or techno-arrogance? *Fisheries* **23**(9):18-23.

Hinder, K., N. Ryman and F. Utter. 1991. Genetic effects of cultured fish on natural fish populations. *Can. J. Fish. Aquat. Sci.* **48**:945-957.

Hoese, H.D. and R.H. Moore. 1977. *Fishes of the Gulf of Mexico: Texas, Louisiana, and adjacent waters.* Texas A & M University Press, College Station, TX. 327 pp.

Lowe, T. and S.R. Eddy. 1997. tRNAscan-SE: a program for improved detection of transfer RNA genes in genomic sequences. *Nucl. Acids Res.* **25**:955-964.

Martin, A.P., R. Humphreys and S.R. Palumbi. 1992. Population genetic structure of the armorhead, *Pseudopentaceros wheeleri*, in the North Pacific ocean: application of the polymerase chain reaction to fisheries problems. *Can. J. Fish. Aquat. Sci.* **49**:2386-2391.

Murdy, E.O. 1983. *Saltwater fishes of Texas: a dichotomous key.* Texas A&M University Sea Grant College Program, College Station, TX. 220 pp.

National Marine Fisheries Service. 1997. *Fisheries of the United States, 1996.* Current Fishery Statistics No. 9500.

Nicoletti, V.G. and D.F. Condorelli. 1993. Optimized PEG method for rapid plasmid DNA purification: high yield from "midi-prep." *Biotechniques* 14(4):535-536.

Palumbi, S.R. 1996. Nucleic acids II: the polymerase chain reaction. Pages 205-247 in: D.M. Hillis, C. Moritz and B.K. Mable, (eds.) Molecular Systematics, 2nd ed., Sinauer Associates, Inc., Sunderland, Massachusetts, USA.

Robins, C.R. and G.C. Ray. 1986. *The Peterson field guide series: a field guide to Atlantic coast fishes North America.* Houghton Mifflin Company, Boston, MA. 354 pp.

Rocha-Olivares, A., N.M. Garber and K.C. Stuck High genetic diversity, deep inter-oceanic divergence and historical demography in the striped mullet *Mugil cephalus*. *Mol. Ecol.* Submitted.

Taggart, J.B., R.A. Hynes, P.A. Prodohl and A. Ferguson. 1992. A simplified protocol for routine total DNA isolation from salmonid fishes. *J. Fish. Biol.* 40:963-965.

Induction of Ovulation in Common Snook, *Centropomus undecimalis* [Bloch], using Human Chorionic Gonadotropin (HCG) and Gonadotropin-Releasing Hormones (GnRH)

C.L. NEIDIG[1], D.S. SKAPURA[1], and H.J. GRIER[2]
[1]*Mote Marine Laboratory*
1600 Ken Thompson Parkway
Sarasota, Florida 34236 USA
[2]*Florida Fish and Wildlife Conservation Commissio*
Stock Enhancement Research Facility
14495 Harllee Road
Port Manatee, Florida 34221-9620 USA

ABSTRACT

Common snook, *Centropomus undecimalis*, is one of Florida's most popular inshore sport fish and the subject of an ongoing stock enhancement program. In the development of hatchery techniques for spawning this species, both human chorionic gonadotropin (HCG) and gonadotropin-releasing hormones (GnRH) were investigated. Doses of HCG were compared to determine the smallest dose that would induce ovulation and produce optimum egg and larval survival. Doses of 50, 100, 250, 500, 1,000, and 2,000 IU/kg body weight (BW) were used. A dose of 500 IU/kg BW of HCG produced consistent ovulation, good egg quality and larval survival. Secondly, four analogs of GnRH were each administered in time-release pellets at a dose of 10 μgm/kg/day over five days. These were salmon (sGnRH), chicken (cGnRH-II), seabream (sbGnRH), and mammalian (mGnRH) gonadotropin-releasing hormones. With the exception of sbGnRH which was ineffective, the time to ovulation was similar with all hormones, and viable larvae were obtained. Control fish did not ovulate.

KEY WORDS: Common Snook, induced ovulation, spawning

INTRODUCTION

Common snook is a popular inshore sport fish in southern Florida. This species is an important candidate for both aquaculture and stock enhancement (Chapman 1982). The procedure to induce final oocyte maturation and ovulation in common snook has been to use HCG at a dose of 1,000 IU/kg BW. In an effort to improve spawning methods, six doses of HCG were compared to determine the lowest dose that would induce ovulation and provide the best egg quality and larval survival. In addition, four forms of gonadotropin-releasing hormone were evaluated for their ability to induce ovulation in common snook. Two of these were known forms (cGnRH-II and sGnRH) that occur in common snook (Sherwood et al., 1993). A third form of GnRH was also isolated from common snook in insufficient quantity for sequence determination. However, sbGnRH has since been sequenced (Powell *et al.* 1994) and may be the same as the unknown form in common snook. The three GnRH forms that have been isloated from fish and the mammalian analog were compared for their ability to induce gonadotropin-releasing activity in the pituitary of seabream, *Sparus aurata* (Zohar *et al.* 1995). We compared their ability to induce ovulation in common snook.

MATERIALS AND METHODS

Common snook were collected during their spawning season using either a seine or trammel net (Taylor et al. 1998). To compare doses of HCG, females were divided into six groups of five fish each and injected with: 50, 100, 250, 500, 1,000, and 2,000 IU/kg BW of HCG. Controls were injected with the carrier, 0.9% sodium chloride. Each form of GnRH was administered to groups of four female common snook as intraperitoneal, time-release GnRH pellets (10 μgm/kg/day). Controls were untreated or implanted with a placebo.

Individual fish were kept in soft-mesh net pens in recirculating tank systems where they were biopsied to follow final oocyte maturation. Percent of fertilized eggs, percent hatch, and percent survival to first feeding (72 hour post-hatch) were evaluated to determine spawn quality (Neidig et al. 2000).

RESULTS

The dose of HCG and number of fish injected versus the number that ovulated were as follows: controls 5:0; 50 IU 5:0; 100 IU 5:1; 250 IU 5:2; 500 IU 5:5; 1,000 IU 5:5; and 2,000 IU 5:5. Control fish did not ovulate. The percent fertilization and percent survival of larvae to first-feeding from fish injected with 250, 500, 1,000, and 2,000 IU/kg BW HCG were within acceptable ranges (Neidig et al. 2000).

In the GnRH study, the control, placebo, and sbGnRH-treated fish did not ovulate. Most (75%) of these fish had no advance in oocyte maturation. Three of four mGnRH-treated fish ovulated, and the ovaries of the fourth reached a preovulatory stage. All of the fish implanted with sGnRH and cGnRH-II ovulated and produced viable eggs.

DISCUSSION

The dose of HCG that produced consistent ovulation and good egg quality was 500 IU/kg BW. Implants of sGnRH, cGnRH-II, and mGnRH induced ovulation and produced viable eggs. The GnRH study was performed in August at the end of the reproductive season, and this may account for one of the fish (mGnRHa) failing to ovulate.

ACKNOWLEDGMENTS

The project was supported by the State of Florida's Marine Resources Conservation Trust Fund.

LITERATURE CITED

Chapman, P. 1982. Final Report for Sportfish Introductions Project. Study 1: Artificial Culture of Snook. Florida Game and Fresh Water Fish Commission, Tallahassee, Florida.

Neidig, C. L., D. P. Skapura, H. G. Grier and C. W. Dennis. [2000]. Techniques for spawning common snook: broodstock handling, oocyte staging, and egg quality. *J. North Amer. Aquacult.* (In press).

Powell, J.F.F., Y. Zohar, A. Elizur, C. Park, W.H. Fisher, A.G. Craig, J.E. Rivier, D.A. Lovejoy, and N.M. Sherwood. 1994. Three forms of gonadotropin-releasing hormone characterized from the brain of one species. Neurobiology. *Proc. Nat. Acad. Sci., USA* **91**:12081-12085.

Sherwood, N.M., H.J. Grier, C. Warby, J. Peute, and R.G. Taylor. 1993. Gonadotropin-releasing hormones, including a novel form, in snook *Centropomus undecimalis*, in comparison with forms in black sea bass ,*Centropristis striata. Reg. Peptides* **46**:523-534.

Taylor, R.G., H.J. Grier, and J.A. Whittington. 1998. Spawning rhythms of common snook in Florida. *J. Fish Biol.* **53**:502-520.

Zohar, Y., A. Goren, M. Tosky, G. Pagelson, D. Leibovitz, and Y. Koch. 1989. The bioactivity of gonadotropin-releasing hormones and its regulation in the gilthead seabream, *Sparus aurata*: *In vivo* and *in vitro* studies. *Fish Physiol. Biochem.* 7:59-67.

Recreational SCUBA Diving Activity in the U.S. Caribbean

GRACIELA GARCIA-MOLINER, WALTER R. KEITHLY, JR.
and IRIS N. OLIVERAS
Caribbean Fishery Management Council
268 Muñoz Rivera Avenue, Suite 1108
San Juan, Puerto Rico 00918-2577

ABSTRACT

SCUBA diving activity has been increasing in the U.S. Caribbean without any statistics on the number of people involved as users of the marine resources. SCUBA diving schools, stores and centers were identified from the telephone book, the tourism guides and diving magazines. A telephone survey, conducted by the Caribbean Fishery Management Council during 1998 - 1999, aimed at (1) identifying these centers to establish a database; (2) gathering information on the (a) number of divers using the resource; (b) areas visited; (c) number of visits per area; and (d) activities carried out by the divers at these sites.

Preliminary results indicate that there are over 100 diving operations in the area. Of these, most are involved in taking divers sightseeing. All levels of expertise are reported, but the beginner and intermediate divers predominate. Diving activities take place at depths between 10 and 20 meters (U.S.Virgin Islands) and 10 - 33 meters (Puerto Rico) over coral reef areas. The preliminary results of the survey show that on average, there are two dive trips per day, five days per week, which include nine divers per trip. As a first approximation, there could potentially be over 300,000 divers per year in the U.S. Caribbean. There were over 90 named diving locations which were reported as visited over 10 times per week. Further analyses of the data will include the number of areas frequented by divers (chartered activities) and the number of dives to each specific reef or wreck. This information, along with descriptions of changes in the species diversity and condition of the reef, could help in determining the effect of divers on reefs.

KEY WORDS: divers, recreational diving, SCUBA

INTRODUCTION

Management of marine resources is usually exercised over commercial fishing activities. This is so because most of the data available to determine the status of fish stocks and the impacts from fishing on these stocks are from the commercial fishing sector. Recreational activities which have direct and indirect impacts over marine resources include boating, cruising, anchoring of cruising and fishing vessels, fishing, snorkeling and SCUBA diving, among others. Most of the information available from the recreational fishing activities comes from big game fishing (marlin) either from tournaments or special projects in both Puerto Rico and the U.S. Virgin Islands. The other source of information is the noncontinuous creel surveys targeting the shore fishing activities (e.g., Mateo 1998) or telephone surveys (e.g., Jennings 1992) which allow for a general estimate of the per cent of the population who fish.

Recreational SCUBA divers impact marine resources, directly by touching and removing coral (habitat) and indirectly by removing fish. No information is available documenting the changes in the SCUBA diving activity in the U.S. Caribbean. The 'sport' became popular in the 1970s and its popularity has continued to increase ever since. There is no documented history of the development of the recreational SCUBA diving industry in the U.S. Caribbean. An additional factor of importance is the increasing number of visitors to the U.S. Caribbean that has also resulted in an increasing number of people diving.

In order to establish a baseline database on the recreational diving activities, and lacking a comprehensive list of SCUBA divers, centers or operators, this study focused on a telephone survey of the diving operations in the U.S. Caribbean during 1998 - 1999. This survey besides identifying and documenting these operations aimed at:

i) estimating the number of divers using marine resources
ii) describing the activities carried out by these divers (e.g., spearfishing, photography)
iii) identifying the areas used, and
iv) estimating the number of visits to each area.

METHODS

SCUBA schools, dive centers or shops, and dive operations were identified from the local yellow pages for Puerto Rico and the U.S. Virgin Islands (St. Thomas, St. John, and St. Croix). Local tourist guides (e.g., Que Pasa (PUERTO RICO), May - June 1999; St. Thomas This Week, February 9-16 1998), newspapers and specialized magazines were searched for information on SCUBA operations. The owners/managers of the Centers also cooperated by providing us with the names and phone numbers of other operators. The local government offices were also contacted (e.g., P.R. Tourism Co., Puerto Rico Department of Natural and Environmental Resources, U. S. Virgin Islands Department of Planning and Natural Resources, etc.) for information on SCUBA diving activities.

The survey conducted included questions aimed at identifying the greatest number of diving operations in the area and the number of years in business, the port of origin of these operations, the sites (reefs) most visited and the frequency of dives on these reefs, the degree of expertise of the divers visiting the reefs and the depths of the dives, the activities allowed by the operators (e.g., photography, collection, harvest), the number of divers and number of dives per trip, as well as the number of trips per week. The answers of the respondents were taken at face value.

The telephone interviews were conducted by the same person and each lasted about 20 minutes unless the respondent provided additional comments or requested information. Each individual store/center was tried at least five times but most interviews were done on the first try with only one refusal. Calls were made during the day only; between 7:00 and 8:00 a.m. and 5:00 and 7:00 p.m. because most diving trips take place between 9:00 a.m. and noon and 1:00 and 4:00 p.m. every day. Night diving was an on-request activity.

RESULTS

The number of diving operations identified in the U.S. Caribbean totaled 104, 26 in the U.S.V.I. and 78 in Puerto Rico, 13 of which have gone out of business (Table 1). The overall effective response rate based on 91 diving operations was 73%, much better than expected. The highest number of diving operations was found in San Juan, Puerto Rico (14) and St. Thomas (13), where most of the tourist populations are found.

Table 1. Number of dive operations identified in the U.S. Caribbean and effective response rate by areas.

	Coast / Island	Number of dive shops	Closed Operations	Answered	Effective Response (%)
Puerto Rico	North	37	6	20	0.65
	East	20	3	12	0.71
	South	10	1	6	0.67
	West	11	0	8	0.73
Total PR		78	10	46	0.68
USVI	St. Thomas	13	3	9	0.90
	St. John	5	0	3	0.60
	St. Croix	8	0	7	0.88
Total USVI		26	3	20	0.87
Total		104	13	66	0.73

Figure 1 shows the cumulative increase in the number of dive shops in the area since 1968 and the range in the number of years in business between two and 30. The activities allowed by the dive operators include sight-seeing, photography, videotaping, spear fishing, hand harvest (lobster), trolling, wreck exploration and some participate in scientific studies (e.g., fish census and measuring coral growth). In the U.S.V.I., 79% of the diving operations are exclusively of a passive nature, not allowing any removal of fish or coral. In Puerto Rico, 51% of the businesses do not allow fishing while 12% did not specify the activities allowed, and 37% allow harvesting. Among the species being harvested are spiny lobster, queen conch, hogfish and snappers.

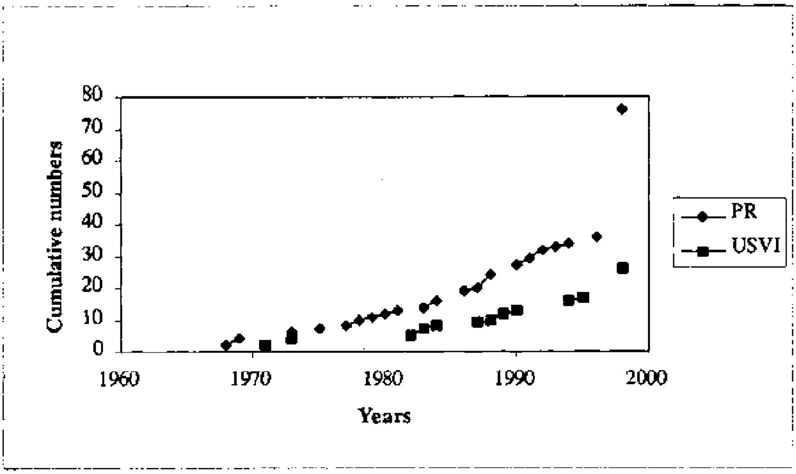


Figure 1. Cumulative increase in the number of dive operations in the U.S. Caribbean

Diving depths were reported to be shallower in the USVI (5- 27 m) than in Puerto Rico (10 - 33 m) with most diving taking place at an average depth of 16 and 20 m in the USVI and Puerto Rico, respectively. The expertise or experience of the divers reflects the differences in depth with most divers in the USVI being beginners and intermediate divers being reported from Puerto Rico.

Most diving operations have one diving vessel, five businesses rent the diving platforms or have no need for one because most diving is done from shore, while six businesses own between two and four boats. Most dive shops are not restricted to their immediate sea area but are agile in moving boats to other coastal areas. In Puerto Rico for example, shops in the north coast concentrated their diving efforts on the east (Palominos and Palominitos) and west coasts (Mona and Desecheo Islands) where there is more coral reef development than in the narrow and high energy north coast.

On average, the number of divers per trip in the USVI is nine while in Puerto Rico is thirteen; dive operators make three dives per day in the USVI while only two in Puerto Rico; and go out five days per week in the USVI and three days per week in Puerto Rico. However, the individual answers were used to estimate the maximum number of divers each center could potentially take out on an annual basis. This was calculated from the number of divers per trip, the number of dives and trips per day and the number of days per week when operators take divers out. It is the number of dives rather than the number of divers that is estimated because it is not known how many are repeat dives versus different divers. Assuming that there is no seasonality effect, meaning that the number of days per week when dive trips take place is the same throughout the year and, that they operate 52 weeks per year, the maximum number of potential dives is 169,156 dives per year in the USVI and 196,664 dives per year in Puerto Rico. The total number of potential dives, only from dive shop operations in the U.S. Caribbean is 365,820 dives per year.

Each respondent identified between one and 14 sites frequently visited, for diving or snorkeling, and in some cases specified the frequency of their visits to each site (e.g., number of visits per week). Based on this information, in the USVI, the diving areas around St. Thomas receive 80% of the total annual potential dives, while St. Croix receives only 13% and St. John 7%. The most visited reef areas around Puerto Rico are the Islands of Desecheo and Mona on the west coast, the Island of Culebra, the keys of La Cordillera such as Palominos and Palominitos and the Island of Vieques on the east coast. These sites receive visits from more than 25% of the dive operations established in Puerto Rico.

Table 2 shows a sample of the most visited areas in the U.S. Virgin Islands and the potential number of dives per year that each reef or site receives. Cow and Calf might be the most frequented site in the USVI.

Table 2: Potential number of divers visiting these sample reef sites in the U.S.V.I.

Island	Reef Area	Divers per year	Potential maximum number of divers per Island
St. Thomas	Buck Island	5,720	
	Cow and Calf	11,648	
	Congo Cay	2,080	
			134,316
St. John	Cow and Calf	832	
	Congo Cay	832	
			11,232
Total	Cow and Calf	12,480	
	Congo Cay	2,912	145,548
St. Croix	Wapa Gardens	884	
	Salt River National Park	936	
	Cane Bay	416	
	North shore	1,768	23,608
Total USVI			169,156

Dive operations allowing fishing account for 37% (Puerto Rico) and 21% (USVI) of the total number of respondents. The number of divers actually fishing could be estimated from the responses given, that is, the maximum number of potential divers harvesting fish if all divers in a party/trip were fishing. Table 3 shows the values estimated for the number of divers harvesting lobster and spearfishing. In the USVI 3%, and 30% in Pureto Rico of the total number of divers could be harvesting fish. If each potential diver is successful in harvesting one lobster or spearfishing one fish, recreational SCUBA divers could potentially be removing 63,440 fish per year. Anecdotal information indicates that one dive shop could sell 22,360 spear guns per year.

The last column in Table 3 is an attempt to estimate the potential impact of divers on corals. Impact being broadly defined as ranging from touching the coral with their hands to re-suspending sediment by flipping their fins. The estimate is obtained by multiplying the number of potential divers (365,820) by two hands and two feet. The potential impact is then 1,463,280 units of impact per year.

Table 3. Maximum number of potential divers fishing and impacting coral in the U.S. Caribbean.

	Divers	Lobster	Spear fishing	Total Fishing	Per Cent	Impacts on Corals
USVI	169,156	3,432	1,248	4,680	0.03	676,624
PR	196,664	3,640	55,120	58,760	0.30	786,656
TOTAL	365,820	7,072	56,368	63,440	0.17	1,463,280

DISCUSSION

Impacts from recreational activities are not well documented in the U.S. Caribbean. A detailed description of recreational fishing effort, harvest, species targeted and fishing practices is not available. The present trend of increasing recreational SCUBA diving activities, including harvesting of commercially important species, shows no indication of leveling off.

A database has been established for diving operations and other diving sectors have been identified (e.g., clubs, classes, certified instructors, private diving). A description of the activities carried out by divers was presented and an estimate of the potential impact calculated. Areas impacted by these activities have been identified but their condition needs to be assessed.

The number of visitors to Puerto Rico between July 1997 and June 1998 was recorded at 4,670,800 of which 3,396,115 spent at least one night on the Island (Puerto Rico Tourism Company, 1998). The Tourism Company estimated 20, 000 divers in 1997. This survey estimates 196,664 divers using marine resources annually, a number that could represent 4% of the total visitors reported to Puerto Rico and, ten times more divers than previously thought. The dive operators in the tourist guides account for only 30% of the operations on the Island.

In the U.S. Virgin Islands, the number of visitors totaled 2,138,900 in 1998 (B.M. Melendez, Government Development Bank, Bureau of Economic Research, United States Virgin Islands). The number of potential divers in the U.S.V.I. was estimated at 169,156 which could account for 8% of the total visitors. The U.S.V.I. dive operators might be targeting the tourist industry, most specifically, the cruise ship tourist. Therefore, the areas around St. Thomas could be receiving more of a direct impact from divers than St. John and St. Croix.

The survey does not include information on the more local diving industry, that is, it does not include the local divers who own boats (over 40,000 recreational vessels registered in the U.S. Caribbean) and equipment and who dive on weekends, on local dive clubs, and other such groups. It does not include information on the certification SCUBA classes offered on average three times a year (beginners, open water and advanced). These groups need to be incorporated into the data gathering process.

This survey does not include information on the costs of running diving operations, nor does it incorporate the economic benefits derived by the local governments from the lure of diving in the tropical coral reef areas. A cost-benefit analysis of the recreational SCUBA diving industry would be welcomed. As a first step it is worth mentioning that in Puerto Rico, the new Fishing Law requires that recreational fishers obtain a license for fishing. This license carries a fee and registration, information that could be helpful in obtaining information about how many divers are actually fishing in Puerto Rico.

Safety is always an issue to consider, and there is only one nonmilitary and operating Hyperbaric chamber in Puerto Rico, which began offering services in 1997. There were twenty four reported diving accidents between August 1997 and May 1998, three of which were recreational SCUBA divers (Holstein and Soto 1998). These divers, two who were certified, had been diving at depths between 27 and 38 meters and were divers not associated with the dive operations. Operational costs of the treatments were not available from the report. The report does not include information on diving accidents which resulted in death prior to arrival at the facilities.

A description of the activities carried out by 66 out of 91 identified dive shops operators in the U.S. Caribbean is finally available. The next step is to set up monitoring at the areas frequented by divers to determine direct and indirect impacts. Historical scientific data exist from some of these areas and before and after comparisons should be attempted. Some of the respondents are already cooperating with monitoring efforts through private non-profit organizations or universities.

The Magnuson-Stevens Act as amended, requires that essential fish habitat be defined and both fishing and non-fishing impacts on habitat be described. It also requires that fishing communities or communities affected by fishing activities be identified and described. Most of the information available is from the commercial fishing sector. Some information is available from the recreational fishing sector but mostly from shore fishing or boat fishing activities. At the time of the Council amending the fishery management plans, nothing was available on the recreational SCUBA diving sector. Questions regarding the number of recreational divers, their distribution and activity patterns, and their overall impact of fisheries and essential fish habitat are still unanswered. The experience of the divers would greatly influence the impact they could have on the coral and surrounding areas. This basic information is also needed to determine what mitigating measures are necessary if there are adverse impacts from inexperience divers, for example. Among these mitigating measures are the placement of mooring buoys and the establishment of restricted fishing or diving areas, among others.

Finally, among the comments received from the respondents the most common ones were the requests for education and orientation, as well as more contact from government regulating agencies and the Council.

LITERATURE CITED

Holstein, D.J. and O. Soto. [1998]. Operational statistics. Medical Center, Hyperbaric Medicine. Administración de Servicios Médicos de Puerto Rico, Centro Médico de Puerto Rico. Unpubl. MS. 46 pp.

Jennings, C.A. 1992. Survey of non-charter boat recreational fishing in the U. S. Virgin Islands. *Bulletin of Marine Science* 50(2):342-351.

Mateo, I. [1998]. Activity and harvest patterns in U.S. Virgin Islands recreational fisheries (Job 4). Annual Performance Report U.S. Fish and Wildlife Service Sport Fish Restoration Program, U.S. Virgin Islands. Unpubl. MS. 19 pp.

Puerto Rico Tourism Company. 1998. *Selected Statistics of the Tourism Industry in Puerto Rico*, 1997 - 1998 Edition. Office of Statistics. San Juan, Puerto Rico.

A Longitudinal Perspective on the Social and Economic Characteristics of the U.S. Gulf of Mexico Charter and Party Boat Industry

ROBERT B. DITTON[1], STEPHEN G SUTTON[1], STEPHEN M. HOLLAND[2],
JOHN R. STOLL[3] and J. WALTER MILON[4]
[1]*Department of Wildlife and Fisheries Science*
Texas A&M University
College Station, TX 77843-2258 USA
[2]*Department of Recreation, Parks and Tourism*
University of Florida
Gainesville, FL 32611-8208 USA
[3]*Public and Environmental Affairs Department*
University of Wisconsin- Green Bay
Green Bay, WI 54311-7001 USA
[4]*Department of Food and Resource Economics*
University of Florida
Gainesville, FL 32611-0240 USA

ABSTRACT

Three hundred and five charter boat operators were randomly selected and interviewed in the five-state U.S. Gulf of Mexico study area (including the Florida Keys); this constituted 24% of the estimated population (1,275) of charter boats in the region. Interviews were also completed with operators of 43 of the 92 party boats (47%) in the same region. Personal interviews were completed between March and September 1998. Interviews lasted from 30 - 60 minutes and covered a range of topics including: demographics, previous experience, community tenure; boat description and recreational fishing products offered; species dependence; financial operations and economic impact; and their opinions on problems facing the industry and fisheries management in the U.S. Gulf of Mexico. This study sought to replicate two previous studies conducted in 1987 by Ditton et al. (1989) and Holland et al (1989). The paper provides a longitudinal perspective on the charter and party boat industry in the U.S. Gulf of Mexico with regard to increases or decreases in boat fleet size; total annual number of trips made, average number of anglers per trip, and boat-trip base fees in standardized U.S. Dollars. Discussion will focus on understanding the resultant trends, charter and party boat definitional problems, methodological challenges, and usefulness of the data in support of fisheries management.

KEY WORDS: Charter boats, head boats, recreational fisheries

INTRODUCTION

Charter boats are typically defined as vessels for hire that carry six or fewer passengers in addition to the crew, whereas party boats (or head boats as they are referred to in Florida) are for-hire vessels that carry more than six passengers (up to 150). We found charter boats with a capacity of up to 25 and cases where some party boats are operated as charter boats to reduce insurance costs. One of the best discriminators of boat type is the base fee charged per trip and the number of passengers included in the base fee. In our study, charter boats are defined as for-hire vessels operating primarily in Federal or EEZ waters with a base fee charged on a group basis. Party boats also operate in the EEZ on a for-hire basis but with a base fee charged on a per-person basis (hence the term head boat). Accordingly, vessels classified as charter boats are smaller, carry six or fewer passengers, and operate only when the vessel has been chartered. Party boats tend to be larger, carry more passengers, operate on a schedule, require a minimum number of passengers in order to make a trip, and will carry as many passengers as possible to maximize income.

In addition to the aforementioned definitional problems, there is also confusion as to whether the for-hire fleet is part of the recreational or commercial fishery. On the one hand, charter and party boat operators may be viewed as a part of the recreational fishery because they provide a unique opportunity for anglers to fish offshore for recreation and food. Alternatively, these boat operators may be viewed as a part of the commercial fishery because they operate a commercial service through which they earn a living from fishing activity (albeit the fishing activity of others). The Gulf of Mexico Fishery Management Council, for example, considered charter and party boat operators a part of the commercial fishery despite the fact their customers must comply with recreational bag limits—"Those fishermen earning their livelihood from the fishery were considered commercial users" (Gulf of Mexico Fishery Management Council 1989:41). In an environment with increasingly restrictive management regulations, this means uncertainty and difficulty for operators trying to make a living providing recreational anglers with offshore fishing opportunities. Classification is further complicated because some operators also use their boats for commercial fishing purposes when not carrying recreational customers. An alternative perspective would be to view the for-hire fleet and its operators as a separate sector unto themselves with ties to both the recreational and commercial fishing sectors.

Charter and party boats are an essential component of the marine recreational fishery because they provide access to offshore fishing for those without boats. In other words, they allow anglers to negotiate one of the most critical leisure constraints, a lack of sufficient discretionary income. Without for-hire vessels, the demographics of offshore anglers would be considerably different in that only those individuals capable of owning offshore vessels and their personal friends would be able to fish in offshore waters. Likewise, in the Gulf and Caribbean region, they provide an important fishing tourism service for visiting anglers who would otherwise lack offshore fishing access. Without a for-hire fleet, anglers would choose to fish elsewhere with local economic impact implications. Federal fishery managers need to do everything possible to maintain for-hire boats and the opportunities they provide from an equity or fairness perspective as well as from a local fishing tourism development perspective.

Nothing stays the same in fisheries; the charter and party boat fleet appears to be growing along with the population and tourism but is in a constant state of flux. For example, studies of the Texas charter boat industry revealed that about 50% of the charter businesses on the Texas coast dropped out or exited the business in two previous five year periods for an average of 10% per year (Ditton and Loomis 1985, Ditton and Vize 1987). New operators and boats entered the industry on a regular basis while others exited. Two possible reasons for this turnover noted by Ditton and Vize (1987) include increasing operating costs for fuel and insurance and regulatory effects of fisheries management. Accordingly, with so much change over time in the industry, it is difficult for fisheries managers to monitor what is happening in the charter and party boat fishery, i.e., harvest, number of trips taken per year, number of anglers carried offshore, etc. The various Gulf of Mexico Fishery Management Council permits that must be held by charter and party boat operators only reveal an intention to fish for the permitted species; little more insight is available without a social survey of the population of charter and party boat operators. Previously, cross-sectional surveys of for-hire boat operators were completed in Texas (Woods 1977, Ditton et al. 1978) and Florida (Browder 1981). These studies provided a foundation for two subsequent social science research efforts focused on charter and party boat operators- one in Florida and the other focusing on the other four Gulf states (Ditton et al. 1989; Holland et al. 1989, Ditton et al. 1992, Gill et al. 1993, Holland et al. 1992). In 1998, we sought to replicate this perspective with another Gulf-wide effort focusing on the industry operating along the coast of Texas, Louisiana, Mississippi, and Alabama (henceforth referred to as the western and central Gulf) (Sutton et al. 1999) and the west coast of Florida including the Florida Keys (Holland et al 1999). Every effort was made to maintain the consistency of questions across the two cross-sectional studies; some new questions were added, however, to understand current management concerns such as the state level of economic impacts associated with the for-hire fleet.

The objectives of this paper are to (1) present an in-depth review of the study methods used to assist others trying to better understand their for-hire fleets and (2) present a preliminary trends perspective on charter and party operators as well as fleet size, number of trips made, and number of anglers carried offshore. To access the complete study results (Sutton et al. 1999, Holland et al. 1999), please consult the web (http://lutra.tamu.edu/rbd/charter.htm).

METHODS

Field personnel between March and September 1998 interviewed a sample of charter and party boat operators in Texas, Louisiana, Mississippi, Alabama, and Florida. Personal interviews were chosen as the data collection technique because a large amount of in-depth information was required on a range of topics. This approach allowed for standardized prompts, probing, and explanations of questions as they were presented to respondents; this resulted in a deeper and more meaningful understanding of problems and attitudes. Most previous studies of charter and party boat operators and their businesses have utilized personal interviews rather than mail questionnaires.

A list of charter and party boats operating in the five-state study areawas compiled carefully from lists obtained from various sources (Gulf States Marine Fisheries Commission; National Marine Fisheries ServiceResearch Lab in Beaufort, North Carolina; and Ditton et al. (1995). None of these lists were used without carefully checking to ensure each listing fit our criteria as an offshore charter or party boat given the information contained in the list. We identified additional operators not included in these lists through various contacts with chambers of commerce and visitors' bureaus and use of Internet sites. Privately used charter and party boats were not included in the sample frame because they could not be consistently located for each study. The inability to locate non-public boats would have made replicating a complete listing of for-hire boats in the study area impossible, resulting in a biased sample frame. Follow-upefforts revealed that some boats included in our sample had new operators, or could not be located. Operators with boats no longer in business were deleted from the list of boats in each state. Also, information provided by boat operators upon contact made it necessary to re-classify some charter boats to party boats and vice versa as per the operational definitions we used.

Charter boats were divided into the regions (Table 1) where they were operated. Approximately 20% of the estimated population of charter boats in each region were sampled randomly using the SAS computer package (SAS 1996). All party boats in the western and central Gulf were included due to their limited number; approximately 33% were sampled on the West Coast of Florida including the Florida Keys. Operators of sampled boats were contacted either by mail or telephone to explain the study and establish a date and time for an interview. Replacements to the charter boat sample were sometimes necessary to maintain the required number of interviews in each region. Wherever possible, replacements were randomly selected from those remaining in the original list using the same procedures used to draw the original sample.

Table 1. Port communities in the study area by region.

State	Region	Port Communities Included
Texas	1	Galveston, Freeport
	2	Port O'Connor, Port Aransas, Aransas Pass, Rockport, Corpus Christi
	3	South Padre Island, Port Isabel, Port Mansfield
Louisiana	4	Grand Isle, Venice, Port Fourchon, Cocodrie, Empire, Cameron City, Intercoastal City, Houma, New Orleans, Grand Isle
Mississip	5	Biloxi, Gulfport, Long Beach, Ocean Springs
Alabama	6	Orange Beach, Dauphin Island, Fairhope, Mobile, Gulf
Florida	7	Panhandle: Pensacola, Destin, Panama City
	8	West Peninsula: Tampa/ St. Petersburg/ Clearwater, Naples, Ft. Myers, Sarasota
	9	Keys: Key West, Marathon, Islamorada

The interview schedule contained 59 questions and took from 30- 60 minutes to complete. The instrument was based largely on the one used by Ditton et al. (1989) and Holland et al. (1989) with some additions and revisions made in consultation with NMFS personnel. Extensive pre-testing was not necessary because most questions had been tested and used previously. The interview schedule consisted of seven sections including the captain's background, boat description, fishing patterns and activities, operating policies, financial operations, opinions on management regulations, and captain's demographics. All respondents were informed of the purpose of the study and the measures taken to maintain the confidentiality of their responses. The respective versions of the interview schedule and informed consent form used were approved by the Institutional Review Boards at Texas A&M University and the University of Florida.

Taking into account the adjustments made during field interviewing and the difficulties involved in identifying the population of charter boats, we estimate there were 845 charter boats operating on the Gulf coast of Florida (including all of the Florida Keys) and 430 charter boats operating in the remaining four Gulf states (Texas, Louisiana, Mississippi, and Alabama) (Table 2). We included all of the boats from the Florida Keys here because we were unable to separate Gulf and Atlantic-side charter boat operations. Interviews were completed with 218 (26%) and 87 (20%) of the estimated population of charter boats in Florida and the four other Gulf States, respectively. Of the 23 party boats identified in the western and central Gulf, we sought to interview all boat operators. Interviews were completed with 21 (91%) party boat operators in this area. One party boat business owner with two party boats refused to allow his captains to be interviewed. In Florida, 53 and 16 head boats (the term used in Florida) were identified on the Florida Gulf coast and in the Florida Keys, respectively. Interviews were completed with 17 (32%) and 5 (31%), respectively.

Table 2. Interview completion rates for each region of the study area.

	Charter Boats			Party Boats/ Headboats		
Region	Estimated Population	Sample Size	%	Estimated Population	Sample Size	%
1	80	15	19	5	3	60
2	70	14	20	9	9	100
3	35	8	23	4	4	100
4	50	12	22	0	0	–
5	85	16	19	1	1	100
6	110	22	20	4	4	100
7	183	50	27	18	1	6
8	432	105	24	35	16	46
9	230	63	27	16	5	31
Total	1,275	305	24	92	43	47

RESULTS

Overall, there were few differences observed in charter and party boat captains' personal characteristics (years of education, marital status, number of people in their household, and income) between the two studies (1987 and 1997). The average age of Florida charter boat captains increased from 42 in 1987 to 46 in 1997; there was a similar aging pattern for head boat captains in Florida. In the western and central Gulf, the percentage of household income derived from the charter business has decreased substantially since 1987. Here, the percentage of household income derived from charter fishing was 69% in 1987 but only

50% in 1997. The pattern was reversed along the West Coast of Florida where percentage of household income from the charter boat business increased from 60% in 1987 to 75% in 1997. In the western and central Gulf, the percentage of household income derived from the party boat business increased from 88% in 1987 to 98% in 1997. In Florida, the percent of household income from their head boat business increased from 85% in 1987 to 93% in 1997; 92% indicated that 100% of household income was from head boats.

The overall population of charter and party boats in the U.S. Gulf of Mexico has increased in size over the past ten years. Since 1987, the number of charter boats in the western and central Gulf has more than doubled (from 210 to 430). The largest percentage increases in charter boats were in Mississippi (300%), Alabama (189%), and Texas (65%) with virtually no growth in charter boats in Louisiana. On the Gulf coast of Florida (excluding the Florida Keys), the number of charter boats has increased by 16% (from 530 to 615). The Panhandle charter boat population decreased 8% from 198 to 183; while the Gulf Peninsula charter boat population increased 30% from 332 to 432. For the Florida Keys, the number of charter boats increased by 17% (from 206 to 241). Growth in the party boat or head boat industry has been less dramatic. On the Gulf coast of Florida (excluding the Florida Keys), the number of head boats has increased by 20% (from 44 to 53) over the past ten years. This is roughly the same percentage growth as seen for charter boats. For the Florida Keys, the number of head boats has decreased by 12% over the same time period (from 18 to 16). Since 1987, the number of party boats in the western and central Gulf has also decreased (from 26 to 23).

Whereas the number of charter boats has increased 104% in the western and central Gulf over the period from 1987 to 1997, the annual number of passenger trips has increased 235% (from 95,000 to 318,716). For the Florida Gulf coast (including the Florida Keys), where the number of charter boats increased 16%, the number of passenger trips has increased from 472,897 to 589,410 (25%). Despite decreasing numbers of party boats in the western and central Gulf and fewer head boats in the Florida Keys, the annual number of passenger trips has increased from 37,148 to 117,990 (217%) between 1987 and 1997 in the western and central Gulf and from 320,587 to 614,888 (92%) for the Florida Gulf Coast (including the Florida Keys) during the same time period.

The increases in annual numbers of passenger trips taken offshore cannot be attributed to increased trip frequency because the average annual number of trips taken (in the Western Gulf) has not increased substantially between 1987 and 1997. Charter boats operating in 1997 in the western and central Gulf made 109 trips on average with a mean of 6.8 passengers per trip (Sutton et al. 1999); in 1987, they made 103 trips on average with an estimated mean of 4 passengers per trip (Holland et al. 1992). Party boats operating in 1997 in the same area made 135 trips on average with a mean of 38 passengers per trip (Sutton et al. 1999); in 1987, they made 131 trips on average with an estimated mean of 19 passengers per trip (Ditton et al. 1992). A different pattern is revealed in Florida where there has a 14% decrease in charter boat trips and a 42% increase in head boat trips between 1987 and 1997. Charter boats operating in 1997 off the Gulf coast of Florida made 134 trips on average with an average of 5.1 passengers per trip, and those operating in the Keys made 171 trips on average with an average of 4.3 passengers per trip; in 1987, charter boats operating from the Gulf and Keys (combined) made 166 trips on average and average of passengers per trip was not asked but an estimated mean of 4 passengers was used (Holland et al. 1989). Party boats operating in 1997 off the Florida Gulf made 280 trips on average with an average of 25.4 passengers per trip and those operating in 1997 in the Florida Keys made 676 trips on average with an average of 22.0 passengers per trip; in 1987, they made 280 trips on average. For Florida Gulf charter boats, there has been about an 8% increase in the proportion of full-day, compared to half-day trips since 1987, while for Florida Keys' charter boats, there has been about a 16% increase in the proportion of full-day, compared to half-day trips since 1987. The most impressive change since 1987, is a more than three-fold increase in the number of half-day trips by head boats and about a 45% reduction in the number of full day trips by head boats, with most of the growth of half-day trips occurring in the Florida Keys. Weather and the length of the vacation season are generally seen as constraints to the number of trips that can be taken on an annual basis. Since the Florida Keys has the longest warm weather season in the U.S. Gulf, this is a possible explanation for the increasing average number of head boat trips. Capital improvements would allow operators to take an increasing number of customers offshore without increasing the number of trips taken on an annual basis.

In the western Gulf, where the average number of trips per boat has remained much the same, our preliminary results indicate charter and party boat operators are using pricing to be able to remain in what has become a very competitive business. Since 1987, charter boat operators have increased their average trip fees by about 40% (in standardized 1987 U.S. Dollars) in the western and central Gulf of Mexico; they have decreased their charter fees by 11% for the Florida Gulf coast (including the Florida Keys). In the former case, they have probably done this to cover the higher costs of operating their larger boats and have made it more palatable to customers by generally increasing the number of passengers included in the base fee. In the case of Florida, the large number of boats provides a competitive barrier to increasing prices. Party boat base fees on average for half- and full day trips in the western and central Gulf are 40% lower than 1987 prices (Ditton et al. 1989); this loss of revenue has been offset by the three-fold increase in passenger trips since party boat fees are charged on a per-head basis. Head boat fees on the Florida Gulf coast (including the Florida Keys) have increased 18% and the additional number of half-day trips has increased the revenue of Florida headboats.

DISCUSSION

An integrated perspective on marine fisheries management including social and economic understanding is still relatively new for many of us. This just wasn't the way prospective fisheries managers were trained in the past. But as our colleagues have come to appreciate the broader definition of the term "fishery", one that includes fish stocks, harvesters, the overall business infrastructure of the fisheries business (whether it be recreational or commercial), and all others with a stake in fishery resources, they begin to see the need for an integrated perspective. If managers are to manage fisheries today, they will require more data and understanding than they have at present because their proposed actions to restore or maintain fish populations are likely to affect a variety of relevant stakeholders. Without the support of the latter for any proposed management actions, it is likely nothing will be implemented. This is because fisheries management takes place in a political environment like all other types of decision making in a democracy. Thus, it is imperative that managers have up-to-date social and economic understandings of their various stakeholder groups and how they are likely to respond to proposed regulations and rulemaking and why if they are to anticipate problems, overcome opposition, and build constituent support. Unfortunately, due to a lack of constituency support for and confidence in the fisheries management process, there have been insufficient funds and little overall investment to date in human dimensions research in order to obtain such needed perspectives. But this may be changing as managers conclude that relying on the best available biological research doesn't necessarily guarantee needed management actions are implemented.

The charter and party boat industry in the U.S. Gulf of Mexico may be an exception to some extent. Not only are harvest levels and number of trips routinely monitored, but studies have been completed to better understand the social and economic structure of the for-hire industry as well as to profile the customers who use charter and party boat services and their experience and management preferences (Ditton et al. 1978, Roehl et al. 1993). Managers have also apparently seen the need for a longitudinal perspective on this group of boat operators and their business activities instead of relying on "snapshot" perspectives afforded by a study completed nearly ten years ago. This replication has yielded understandings of operators and their businesses not available for most other recreational or commercial fisheries issues. Having the types of insights provided by human dimensions research is one thing; using them effectively to propose responsive management directions in the Gulf of Mexico is another. It remains to be seen whether management personnel with primarily biological backgrounds can make wise use of human dimensions data and understandings in support of their statutory responsibilities for fisheries management. While charter and party boat operators have been involved in the fisheries management process as members of various committees over the years, the industry to date has typically played an opposition role to government management proposals rather than one of finding positions to support. Hopefully, human dimensions research results will lead to new management proposals that the industry will be able to support. We have collected a wealth of data on the Gulf of Mexico charter and party boat fishery and only present preliminary results in this paper; much more will be disseminated in the future from this recently completed project.

It was not surprising that individuals with similar social and economic characteristics continue to be involved in this industry. Increases in fleet size and number of anglers taken offshore, on the other hand, are probably due to increases in population and tourism activity (an overall increase in demand for offshore recreational fishing) Gulfwide and in the western Gulf, greater boat size and efficiency. The question remains whether or not this type of growth can continue or even be sustained at present levels given the status of the fish stocks upon which the charter and party boat industry depend and more restrictive harvest regulations. That the number of vessels has increased in this increasingly restrictive management environment is surprising. Since the Gulf of Mexico Fishery Management Council first implemented management plans for coastal pelagic and reef fish fisheries, nearly all Gulf of Mexico species targeted by the for-hire fleet have come under increased management with greater harvest restrictions and uncertainty from year to year on bag and size limits. A recent example of this uncertainty is the current 18 inch length limit for red snapper imposed this past June; whereas all anglers were impacted to some degree by the sharp increase in minimum size from 14 inches, the prospect of zero catch has particularly hurt the business of party boat operators who target primarily snapper and grouper in the western Gulf of Mexico. For only if boat operators are able to provide their current customers with fishing trips that meets their experience preferences will they be able to maintain their annual number of trips offshore. To remain in business, they will need some stability in fishing rules and regulations so as to maintain their current customers or they will need to find new customers willing to accept constantly changing size and bag limits, lower catches (or perhaps even catch and release fishing-only with appropriate target species).

The longitudinal study design helped us to understand the extent of change in parameters of interest for the charter and party boat fleet, for example, between two points in time. However, it did not provide a complete record of the change that has occurred. There has likely been extensive turnover among charter fishing operators between 1987 and 1997 based on previous research on this topic (Ditton and Loomis 1985, Ditton and Vize 1987). Because managers are interested mainly in net numbers of boats in the fleet, the number of boats targeting particular fishery resources, and overall harvest by this user segment, this turnover may have little meaning to them. But this turnover may be an indicator that these small-business operators are being impacted by government regulations seeking to maintain fish populations as well as the uncertainty of a distant and often unresponsive regulatory process. Fishery managers in the U.S. are required by law to carefully evaluate and take into account regulatory impacts on businesses and organizations. One of the best ways for accomplishing this is to devote much more effort to maintaining an updated database of charter and party boat operators in the Gulf of Mexico than is currently the case. Besides name and address, basic social and economic descriptive information could be collected to provide a current-status look at this group of fishery stakeholders. The key would be the development and maintenance of an up-to-date database of boat operators. With an understanding of the number of years they have been operating their boats, for example, management staff would have a useful means for tracking operator turnover, possible regulatory impacts, and the implications for service delivery. What has happened in the charter and party boat industry in the intervening years should be just as important to managers as our study results for 1987 and 1997.

ACKNOWLEDGMENTS

This research was supported by the MARFIN Program, National Marine Fisheries Service through research grants to Texas A&M University (Grant Number NA77FF0551) and the University of Florida (Grant Number NA77FF0553)

REFERENCES

Browder, J.A., J.C. Davis, and E. Sullivan. 1981. Paying-passenger recreational fisheries of the Florida Gulf Coast and Keys. *Marine Fisheries Review* 43(8):12-20.

Ditton, R.B., L.D. Finkelstein, and J. Wilemon. 1995. Use of offshore artificial reefs by Texas charter fishing and diving boats. Technical Report #HD-604. Department of Wildlife and Fisheries Sciences, Texas A&M University, College Station. 38 pp.

Ditton, R.B., S.M. Holland, and D.A. Gill. 1992. The U.S. Gulf of Mexico party boat industry: activity centers, species targeted, and fisheries management opinions. *Marine Fisheries Review* **47**(1):43- 47.

Ditton, R.B., R.N. Jarmen, and S.A. Woods. 1978. An analysis of the charter boat fishing industry on the Texas Gulf Coast. *Marine Fisheries Review* **40**(8):1-7.

Ditton, R.B. and D.K. Loomis. 1985. Examining business turnover in the Texas charter boat industry: 1975- 1980. *Marine Fisheries Review* **47**(1):43-47.

Ditton, R.B., T.J. Mertens, and M.P. Schwartz. 1978. Characteristics, participation, and motivations of Texas charter boat fishermen. *Marine Fisheries Review* **40**(8):1-7.

Ditton, R.B., J.R. Stoll, and D.A. Gill. 1989. The social structure and economics of the charter and party boat fishing fleets in Texas, Louisiana, Mississippi, and Alabama. Final Report to U.S. Department of Commerce, NOAA, National Marine Fisheries Service, Contract 5617000, St. Petersburg, FL. 259 pp.

Ditton, R.B. and J.D. Vize. 1987. Business turnover in the Texas charter fishing industry, 1975 - 1985. *Marine Fisheries Review* **49**(2):162-165.

Gill, D.A., R.B. Ditton, and S.M. Holland. 1993. Charter and party boat operators in the U.S. Gulf of Mexico: A social structure perspective. *Marine Fisheries Review* **55**(3)16- 20.

Gulf of Mexico Fishery Management Council. 1989. Amendment Number 1 to the Reef Fish Fishery Management Plan. Gulf of Mexico Fishery Management Council, Tampa, FL.

Holland, S.M., R.B. Ditton, and D.A. Gill. 1992. The U.S. Gulf of Mexico charter boat industry: activity centers, species targeted, and fisheries management opinions. *Marine Fisheries Review* **54**(2):21- 27.

Holland, S.M., A.J. Fedler, and J.W. Milon. 1999. The operations and economics of the charter and head boat fleets of the eastern Gulf of Mexico and South Atlantic coasts. Report to U.S. Department of Commerce, NOAA, National Marine Fisheries Service, Grant Number NA77FF0553, St. Petersburg, FL.

Holland, S.M. and J.W. Milon. 1989. The structure and economics of the charter and party boat fishing fleet of the Gulf coast of Florida. Final Report to U.S. Department of Commerce, NOAA, National Marine Fisheries Service, Contract NA87WC-H-06141, St. Petersburg, FL. 278 pp.

Roehl, W.S., R.B. Ditton, S.M. Holland, and R.R. Perdue. 1993. Developing new tourism products: sport fishing in the southeast United States. *Tourism Management* (August):278-288.

Sutton, S.G., R.B. Ditton, J.R. Stoll, and J.W. Milon. 1999. A cross-sectional study and longitudinal perspective on the social and economic characteristics of the charter and party boat fishing industry of Alabama, Mississippi, Louisiana, and Texas. Human Dimensions of Recreational Fisheries Research Laboratory #HD-612. Department of Wildlife and Fisheries Sciences, Texas A&M University, College Station, TX. 198 pp.

Woods, S.A. Woods. 1977. A business analysis of the party boat industry on the Texas Gulf Coast. Master of Science Thesis, Department of Management, Texas A&M University College Station TX. 103 pp.

Assessment of the Recreational Sport Fisheries of Florida Bay and Adjacent Waters from 1985-1998

THOMAS SCHMIDT, GABRIEL DELGADO, and MARIO ALVARADO
South Florida Natural Resources Center
Everglades National Park
40001 State Road 9336
Homestead, FL 33034

ABSTRACT

A stock assessment based on recreational catch and catch rate (CPUE) from 1985 - 1998 was conducted on four of the most popular gamefish in Florida Bay, Everglades National Park: snook (*Centropomus undecimalis*), gray snapper (*Lutjanus griseus*), spotted seatrout (*Cynoscion nebulosus*), and red drum (*Sciaenops ocellatus*). The responses of catch to fishing effort and CPUE to environmental factors (rainfall, water level, and salinity) were determined.

Snook catch rates have shown a cyclical trend every four years. The peaks may reflect recruitment of juveniles that were released in prior years because of size restrictions. Recruitment may also be enhanced by increased rainfall/runoff as there was a weak correlation between water levels recorded three years before and CPUE from 1985 - 1998. Although, no statistical significance was found, this trend suggests that periods of low salinity lead to increases in abundance.

During the 1990s, gray snapper CPUE and estimated total harvest have dropped lower than anytime during the previous record and may be due to regulations imposed on the fishery in 1988 and 1990. A significant relationship was found between CPUE and mean annual salinities in northern Florida Bay suggesting that periods of high salinity may lead to increases in abundance. Mean annual rainfall and water levels were significantly inversely related to CPUE indicating that low rainfall and water levels lead to increases in abundance.

Spotted seatrout CPUE has increased recently; yet, harvest rates have been holding steady since 1990. The lack of increase in harvest associated with the increase in catch may be due to size restrictions. A significant negative correlation was found between CPUE and water levels from the previous year.

Red drum CPUE has been stable since 1989 when present bag limits were imposed. No statistically significant relationships were found between CPUE and any of the environmental variables.

For each species, annual estimated total catch was highly correlated with the estimated total effort. This suggests that current catches do not greatly impact Florida Bay stocks and that additional increases may be possible.

KEY WORDS: Everglades National Park, Florida Bay, gray snapper, recreational fisheries, red drum, snook, spotted seatrout

INTRODUCTION

Fishing activity and harvest of sportfish from Everglades National Park (ENP) have been monitored nearly continuously since 1958. This project represents one of the longest ongoing recreational fisheries monitoring programs in the world. The mainland shoreline of ENP extends from the Florida Keys to Everglades City on Florida's west coast. Tabb et al (1962) have described the animal and plant communities of the park's waters and identified distinct ecological zones which vary in their topographical, hydrological, and biotic characteristics. These zones were utilized to delineate fishing areas used in this and other ENP fishery investigations since 1960 (Figure 1). As in Tilmant (1989) and Rutherford et al. (1989bc), we refer to fishing Areas 1 - 5 as Florida Bay and adjacent waters.

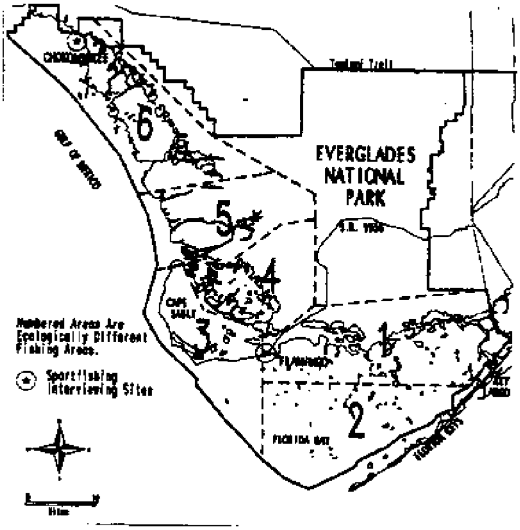


Figure 1. Ecologically different fishing areas in Everglades National Park

In the 1970s, an overall decline in the recreational harvest of gamefish was observed in the Florida Bay portion of ENP. This decline was only partially explained by a decrease in fishing effort, suggesting that population changes were reflective of long-term cyclic trends or possibly, climatic and environmental conditions. In 1985 commercial fishing was eliminated in ENP; presently only recreational guided and non-guided (sport) anglers are permitted to fish within park waters. From the mid 1980s to the early 1990s, further concerns over declining catches and catch rates (CPUE) of gamefish species in Florida Bay were noted (Boesch 1993). This prompted ENP to revisit the effects of fishing and environmental factors on the marine resources of the park. Previous studies

(Higman 1967, Davis 1980, Rutherford et al. 1989b, c) indicated that environmental factors may explain as much of the variability in fish abundance as does fishing pressure. For example, during 1989 - 1990, extreme environmental conditions (drops in water temperature and hypoxia, possibly related to seagrass die-off) resulted in a large fish kill in Florida Bay (Schmidt and Robblee 1994). Since sportfish catches are correlated with indices of freshwater runoff (ie. rainfall and water level), it was felt that the reduction of freshwater inflow to Florida Bay may have been a factor in the declining catch rates (Boesch 1993).

Environmental relationships and long-term population trends can only be accurately expressed on a species by species basis (Tilmant 1989). Therefore, a stock assessment based on sport fishermen (non-guided) catch and CPUE from 1985 - 1998 was conducted on four of the most popular gamefish in Florida Bay: snook (*Centropomus undecimalis*), gray snapper (*Lutjanus griseus*), spotted seatrout (*Cynoscion nebulosus*), and red drum (*Sciaenops ocellatus*). The responses of catch to fishing effort and CPUE to environmental factors (rainfall, water level, and salinity) were determined.

METHODS

Methods employed to obtain sport fishing and boating activity in ENP have been previously documented by Higman (1967), Davis and Thue (1979), Tilmant et al. (1986), and Tilmant (1989) and are briefly discussed below as they pertain to this study. Recreational anglers are interviewed at boat launch sites (Flamingo and Chokoloskee) upon completion of their trip every weekend (Figure 1). Data recorded includes area fished, fish kept, fish released, effort (angler-hours and trip-hours), species preference, and angler residence.

Flamingo is the greatest single access point to Florida Bay, used by 50 - 65% of the anglers. Daily estimates of the number of fishing boats operating in park waters were made by regressing the daily counts of empty trailers at Flamingo against a known number of boats fishing the same day. Aerial surveys were used to determine the correlation of boat trailers at the Flamingo launch ramps to the total number and distribution of boats within the park. A highly significant linear relationship was found between the number of trailers at Flamingo and the boats observed in park waters ($r = 0.84$, $N = 243$, $p < 0.01$). The percentage of boats actually fishing was determined from angler interviews.

The catches of the interviewed anglers are only samples of the total park harvest. The estimated total catch and harvest of each species for the non-guided (sport) fishery were calculated by dividing the recorded daily catch or harvest by the total number of boats interviewed to give the mean catch or harvest per trip. This figure was then multiplied by the estimated number of fishing boats in Florida Bay to obtain the estimated total catch or harvest. In obtaining the mean catch rate (CPUE) and harvest rate (HPUE) for a species, rates of individual trips were calculated after Malvestuto (1983). Only those anglers successfully catching a species were used to calculate the CPUE to avoid bias in the possible change in effort applicable to a species each year. Only those anglers harvesting a species were used to calculate the HPUE.

For the purposes of this study, catch rates were used as an index of relative abundance. Catch rates are directly related to environmental factors such as rainfall, and are generally, not directly affected by fishing regulations while harvest rates most certainly are. The catch rates for the four major species were correlated with rainfall, water level, and salinity from 1985 - 1998. SPSS 9.0 was used to determine all correlation statistics. Total annual rainfall from 1985-98 was compiled and averaged from five stations within or near ENP (Flamingo, Royal Palm, Everglades City, Tamiami Ranger Station, and Tavernier; Butternut Key replaced Tavernier in 1997 and 1998). Water level data from 1985 - 1998 was obtained from well P-37 in western Taylor Slough. Salinity data from 1985 - 1998 was obtained from three stations in northern Florida Bay (Butternut Key, Taylor River, and Trout Cove).

It is not sufficient to know if catch rates are declining to determine if stocks are overfished. If both catch and catch rate are in decline, then there is a need to assess the amount of effort being placed on the fishery. The estimated total catch and estimated total effort of the four major species were correlated to determine if fishing effort impacted the stock.

RESULTS and DISCUSSION

Description of the Fishery

Most of the anglers fishing out of Flamingo were south Florida residents (Dade County to Ft. Lauderdale, excluding local residents from Flamingo, Florida City, and the Florida Keys). Most of the anglers interviewed did not try to catch any particular kind of fish. Over 80 species of fish and shellfish have been reported in the recreational catches of ENP since 1972; however, snook and red drum were the most sought after fish, followed by spotted seatrout, gray snapper, and tarpon.

The contribution of individual species has experienced considerable variation over the years. The biggest changes in catch composition appeared to be a noticeable decline in the proportion of spotted seatrout catches during recent years, the erratic year to year variation in the frequency of gray snapper catches, and the increase (1989 - 1998) in the proportion of snook and red drum catches (Figure 2).

Spotted seatrout are the most frequently caught sportfish in Florida Bay. Seatrout are estuarine dependent and spend their entire life cycle within ENP (Rutherford et al. 1982). Tagging studies indicated little inter-estuary movement

(Iversen and Moffett 1962, Beaumarige 1969, Rago and Goodyear 1985). This suggests that there are two unit stocks of spotted seatrout in park waters, one in the Florida Bay-Whitewater Bay area (Areas 1 - 5) and one in the 10,000 Islands area (Area 6) (Tilmant 1989, Schirripa and Goodyear 1994).

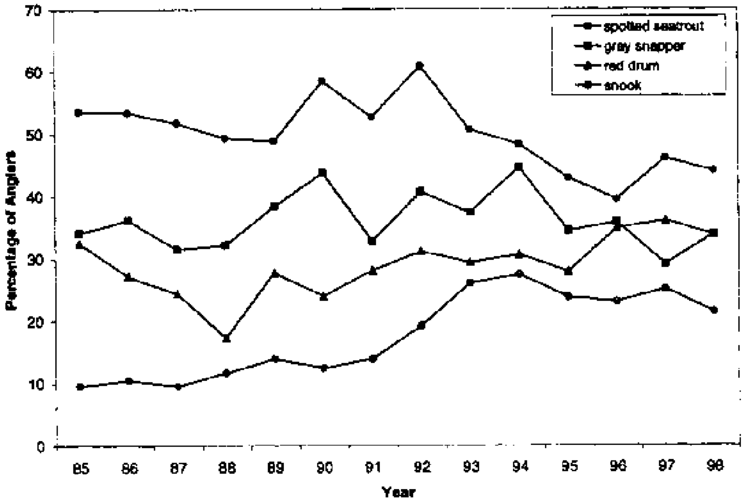


Figure 2. Percentage of anglers interviewed at Flamingo (Areas 1-5) catching spotted seatrout, gray snapper, red drum, and snook from 1985 - 1998

The second most commonly caught sportfish in Florida Bay is gray snapper. Snapper enter ENP as small juveniles using the area as a nursery, recruit to the fishery at age one, and grow up in park waters to age 3 - 4 (Rutherford et al. 1989a). Rutherford et al. (1989a) found that 3 - 4 year olds comprised as much as 87% of the fish harvested, with some up to age seven harvested, but few over age four. Snapper then start leaving the estuary to spawn and live on offshore reefs (Rutherford et al. 1983). Emigration is further supported by Schmidt et al. (1999); they found that harvested fish were larger in the outer regions of Florida Bay.

Red drum are a typical euryhaline species that use estuaries as nursery areas before migrating offshore to the more open waters of the Gulf between ages 3 - 5 (Yokel 1966). Therefore, the red drum fishery in ENP is directed towards late stage juvenile and early adult fish remaining in the park's estuaries (Tilmant et al. 1989). Tagging studies have shown little inter-bay movement of immature red drum in Florida (Ingle et al. 1962, Topp 1963).

Snook are a relatively non-migratory, estuarine dependent species. Snook will make localized movements between estuaries as juveniles and move to nearby offshore areas as adults for spawning. The fishery consists of snook aged 3 - 5 years old (Thue et al 1982, Muller and Murphy 1997). Two unit stocks have been identified in south Florida, one population on the Gulf coast and the other on the Atlantic coast with differences in size and age based on genetic properties (Tringali and Bert 1996). Florida Bay snook may comprise an immigrant adult population derived from one or both populations.

The overall trend in recreational fishing boats since 1973 shows high values in 1973 - 1975, with lows in 1979 - 1980, and a rebound in the mid-1980s (Figure 3). The decline during the late 1970s occurred because of increased gasoline costs and decreased fish abundance/catch rates; this combination of factors may have deterred anglers from visiting ENP (Tilmant et al. 1986). The decline in 1992 is attributed to the impacts of Hurricane Andrew; ENP was closed from September through December. There has been an increasing trend since 1992 with the highest number of fishing boats recorded during 1997 (28,927 fishing boats) followed by a slight decline in 1998 (Figure 3). The recreational fishing effort (angler-hours) has mirrored this trend as well (Figure 3).

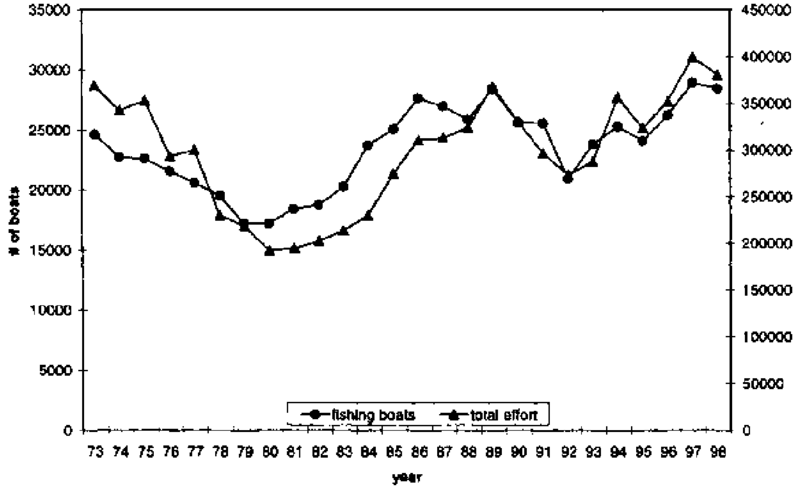


Figure 3. Estimated number of fishing boats and estimated total effort in Florida Bay (Areas 1 - 5), 1973 - 1998

Snook

Snook popularity has risen dramatically from 1985 - 1994. The percentage of boats catching snook in Florida Bay increased from 9% in 1985 to nearly 27% in 1994, but suffered a slight decline in recent years (Figure 2).

Harvest rates have been quite stable since 1985; however, catch rates have shown a cyclical trend every four years (Figure 4). Catch rate decreased from 1985-88, reaching a low of 0.1674 fish/angler-hour in 1988, only to increase to a high of 0.326 fish/angler-hour in 1992. Another low was reached in 1997 (0.217 fish/angler-hour); catch rate started to increase yet again in 1998 with a value of 0.229 fish/angler-hour. Muller and Murphy (1997) reported catch rates of similar magnitude and trends as those found in this study. The increases may reflect stock recruitment of small, juvenile snook, which were released in prior years because of size restrictions. The timing of the cycle seen in catch rate is probably partly due to the four years needed for juveniles to recruit to the fishery (Thue et al. 1982).

The declines in snook stock size from 1985 - 1988 may have been due to low rainfall and water levels in the upper marsh regions. There was a weak correlation between water levels recorded three years before and catch rates from 1985 - 1998 ($r = 0.591$, $N = 11$, $p > 0.05$). Although no statistically significant correlation was found, this suggests that a period of high rainfall/water level leads to an increase in the abundance of snook. Field studies on snook habitat have shown that the greatest number of juvenile snook are consistently found in shallow, well protected, back-water areas of estuaries that are influenced by freshwater runoff (Fore and Schmidt 1974, McMichael et al. 1987). Van Os et al. (1981) found that snook catch rate within the St. Lucie River estuary, was positively correlated with the length of river control discharges. They concluded that snook move into the freshwater discharges to take advantage of the augmented food supply.

The total estimated catch of snook from the sport fishery in Florida Bay increased from a low of 6,538 fish in 1986 to a high of 22,581 fish in 1994. However, snook catches have declined since 1994; 14,641 fish were caught in 1998. Despite the two fish per person bag limit and closed seasons during January, June, July, August, and December, sport fishermen harvest had not been reduced until 1998 (Figure 5). During 1985 - 1994, there was a 148% increase in catch, while harvest increased only 40%, suggesting that increased catches were due to young fish that were released in prior years. The increase in catch during 1991 - 1994 has been credited to the quick recovery of the stocks because of the regulations mentioned above; although, it should be noted that fishing effort doubled from 1991 - 1994.

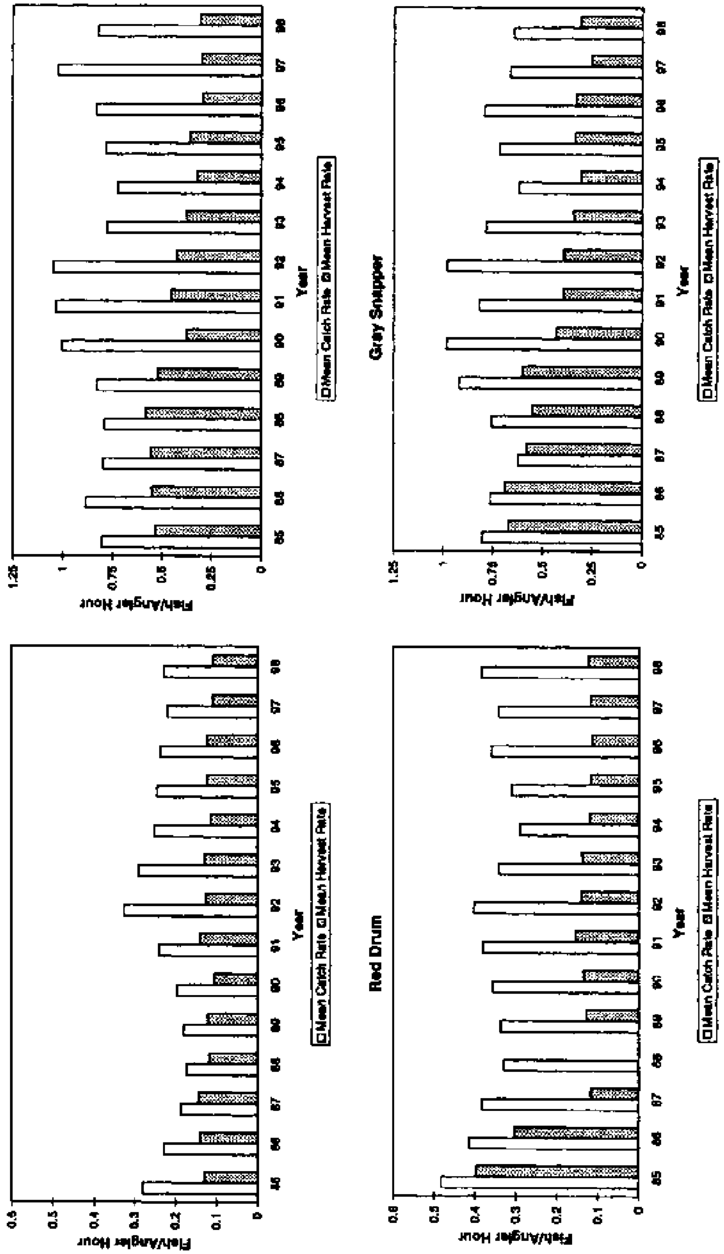


Figure 4. Recreational non-guided (sport) catch and harvest rates for the four major species of gamefish in Everglades National Park (Areas 1 - 5), 1985 - 1998

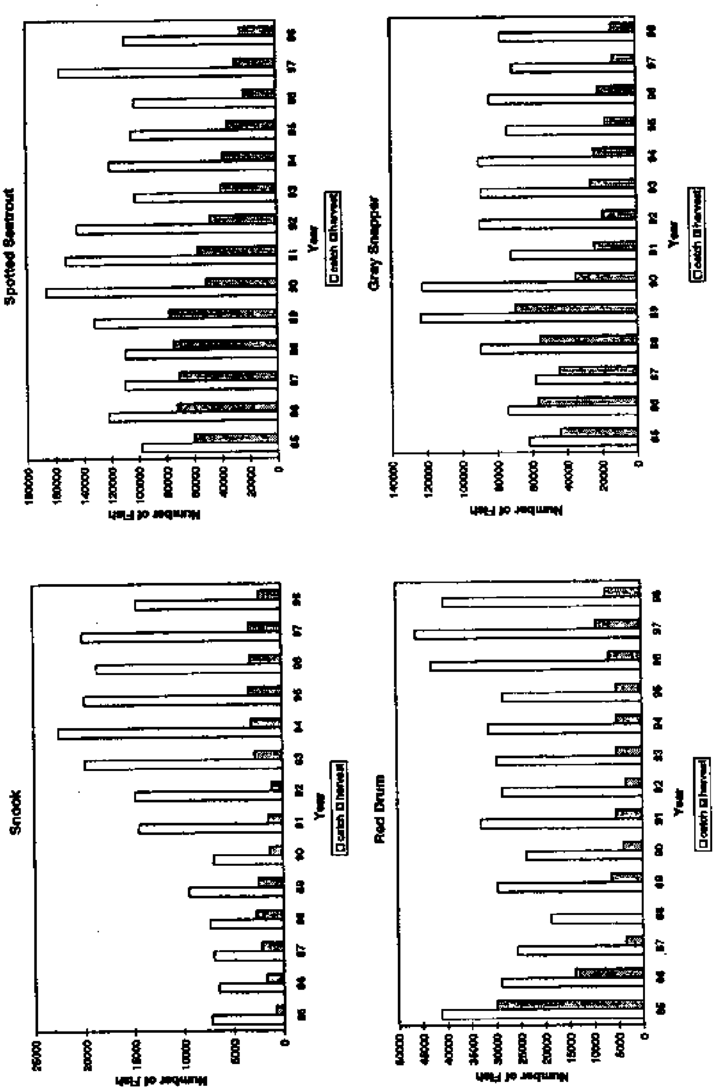


Figure 5. Estimated total catch and estimated total harvest of the four major species of gamefish by non-guided (sport) anglers in Everglades National Park (Areas 1 - 5), 1985 - 1998

Annual fishing effort of sport anglers catching snook in Florida Bay ranged from a low of 26,775 angler-hours in 1985 to a high of 107,825 angler-hours in 1997. The estimated total catch of snook for the sport fishery was highly correlated with the estimated total effort placed on the stock from 1985 - 1998 ($r = 0.901$, $N = 14$, $p < 0.0001$) (Figure 6). This suggests that current catches do not greatly impact the Florida Bay stock and that additional increases may be possible. Muller and Murphy (1997) also concluded that snook stocks in south Florida are in good condition. However, snook catches decreased dramatically in 1998 after five years (1993 - 1997) of good catches and all time high in effort in 1997. CPUE from 1993 - 1997 was decreasing, but catch remained very high because of the high fishing effort. During 1998, state regulations were revised to sustain a 40% spawning potential ratio (SPR) by increasing the minimum size to 26," while maintaining a maximum size of 34" and a two fish bag limit. The increase in minimum size may also be responsible for the decline in harvest in 1998.

Gray Snapper

The percentage of anglers reporting catches of gray snapper has fluctuated from year to year from 1985 - 1998 (Figure 2). The large decline seen in 1991 was probably due to new regulations imposed in 1990, which increased the minimum size to 10" with a bag limit of five fish per person. Snapper catch rate has shown no definitive trend; although, catch rates were highest during 1989 - 1990 (Figure 4). Snapper catch rate has declined the last two years, while harvest rates have shown steady declines since 1985 (Figure 4). During 1988 - 1990, the increase in catch rate, but not harvest rate was reflective of the new regulations.

A significant, positive relationship ($r = 0.601$, $N = 14$, $p < 0.03$) was found between catch rate and mean annual salinities found in northern Florida Bay, suggesting that periods of high salinity may lead to increased abundance of gray snapper. Average annual water levels recorded at P-37 were significantly inversely related to gray snapper catch rates during the same year ($r = 0.712$, $N = 14$, $p = 0.004$), indicating that during periods of reduced water levels in upper Taylor Slough, abundance of gray snapper increased. Rainfall was also inversely correlated with gray snapper catch rates ($r = 0.506$, $N = 14$, $p > 0.06$). Which leads to the theory that increases in gray snapper abundance during the period of 1988-90 may have been related to low yearly rainfall in the ENP area and periods of high salinities in Florida Bay. A series of low rainfall years from 1985 - 1990 resulted in hypersaline conditions in Florida Bay. Rutherford et al. (1983) reported larger fish in areas of higher salinity. Thus, if during low rainfall years (which cause high salinity conditions), sub-adult fish remain in Florida Bay longer, then gray snapper abundance should increase and the fish would become increasingly available to the angler (higher catch rates). During 1992 - 1995, water levels/rainfall increased, especially from Tropical Storm Gordon in November 1994, resulting in salinity reductions in northern Florida Bay with a notable decrease in gray snapper CPUE.

There was also a significant inverse relationship between catch rates and mean annual water level recorded one year before (r = 0.574, N = 13, p = 0.04), and between catch rates and mean annual rainfall recorded three years before (r = 0.778, N = 11, p = 0.005). These relationships are explained by the fact that higher salinities enhance the growth and survival of juvenile snapper which would recruit to the fishery in future years (Higman 1967, Rutherford et al. 1989c).

The estimated total catch and CPUE of gray snapper follow the same general trend; although, CPUE fluctuates a little more. The estimated catch in Florida Bay had been increasing from 1985, reaching highs in 1989 - 1990 following new size restrictions and highs in effort as well (Figure 5). The catch then dropped in 1991 and has remained relatively stable since (Figure 5). During the 1990s, the estimated total harvest for gray snapper has dropped lower than anytime during the previous record (Figure 5). This is probably due to the state regulations imposed on the fishery in 1990 when the minimum size was increased from 8" to 10" with a bag limit of five fish per person. These regulations may have prevented increases in snapper harvest because anglers shifted their efforts to red drum and seatrout instead; however, reduced stock size also limited harvest (Rutherford et al, 1989c).

Annual estimated effort for the gray snapper fishery ranged from a low of 96,311 angler-hours in 1985 to a high of 168,239 angler-hours in 1994. The yearly catches were lowest in 1985 (61,859) and 1987 (58,401) and highest in 1989 (123,707) and 1990 (122,327). Increased size limits in 1988 and 1990 and the imposition of new bag limits in 1990 may account for the high number of gray snapper caught and released during those years. The estimated total catch was linearly correlated with the estimated total effort from 1985-98 ($r=0.705$, $N=14$, $p<0.005$) suggesting that the maximum potential catch of the fishery in Florida Bay has not been reached (Figure 6).

Spotted Seatrout

The percentage of boats catching seatrout declined steadily from 1985-89, but increased sharply to a 14 year high of almost 65% in 1992 (Figure 2). Since then, the percentage of anglers catching seatrout declined to a low in 1996 of 39% (Figure 2). Seatrout were caught by 44% of the anglers in 1998. Fishing regulations may have affected angler strategy as the declining trend in seatrout is associated with increases in snook and red drum; fishermen may have switched their targeting preference to the latter two species when their stocks recovered.

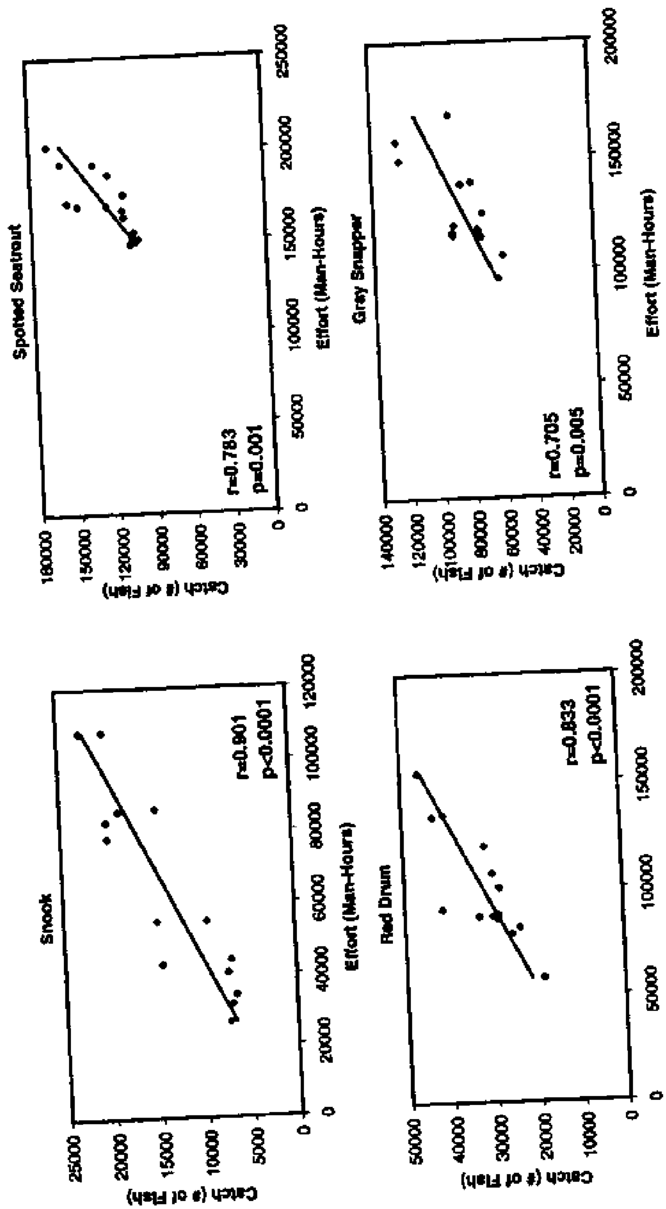


Figure 6. Correlation of total estimated catch and total estimated effort for the for major species of gamefish in Everglades National Park (Areas 1 - 5), 1985 - 1998

Seatrout catch rate had been holding steady from 1985-88, then increased from 1989 - 1992, but dropped in the two ensuing years (Figure 4). Since 1994, seatrout catch rate has been increasing in Florida Bay; however, there was a slight drop in 1998 to 0.8231 fish/angler-hour (Figure 4). Harvest rates have been decreasing slowly, but steadily over the period of record (Figure 4). The lack of increase in harvest rate associated with the increase in catch rate may be due to state regulations imposed on the fishery in 1989 which raised the minimum size from 12" to 14," and then for south Florida populations to 15" in 1996. These regulations were meant to reduce harvest to achieve a 35% SPR set by the Florida Marine Fisheries Commission (FMFC).

Seatrout catch rate and salinity seem to follow the same trend. As salinity increased to a high in 1990, seatrout catch rates increased and as salinities dropped in the proceeding years, catch rates also decreased; however, there was no statistically significant relationship between the two variables from 1985-98. When catch rates were correlated with annual water levels recorded at P-37 from the previous year, a significant inverse relationship was found ($r = 0.552$, $N = 13$, $p = 0.05$). In contrast, Thayer et al. (1998) reported increases in the abundance of larval and small juvenile seatrout in northern Florida Bay, suggesting that lowered salinities in this area may increase the survival of young-of-the-year fish. In laboratory studies, Taniguchi (1980) and Wakeman and Wolschlag (1977) found that survival and growth for larval and juvenile seatrout was greatest at 28ppt.

The estimated total catch and CPUE of seatrout follow the same general trend. Catch increased to a high in 1990 (effort also reached its peak in 1990), then declined till 1996 (Figure 5). There was a large spike in fish caught in 1997 due to increased effort (Figure 5), but may have also been due to the increase in minimum size to 15" in 1996. The estimated total harvest decreased dramatically from highs in 1988 - 1989 to a low in 1996 with a slight rebound during 1997-98 (Figure 5).

Total estimated effort for seatrout ranged from a high of 202,383 angler-hours in 1990 to a low of 147,882 angler-hours in 1995. The correlation of yearly effort with catch was linear and significant ($r = 0.783$, $N = 14$, $p < 0.001$) (Figure 6). There was no decrease in total catch with increasing effort, indicating that yearly fishing effort did not severely impact the fishery.

Red Drum

The percentage of boats catching red drum decreased dramatically from 33% in 1985 to 17% in 1988 when the fishery was closed due to overfishing (Figure 2). When harvest was reopened, the percentage of anglers catching the species increased steadily to a 14 year high in 1997 of 36% (Figure 2).

Our analysis indicates that red drum in Florida Bay have recovered since

1988. However, abundances remain considerably lower than during the period of 1979-85 (Tilmant et al. 1989). Catch rates have been increasing steadily since an all time low in 1994 to 0.3842 fish/angler-hour in 1998 (Figure 4). Since the fishery recovered faster than anticipated, the FMFC allowed year-round fishing in 1996, which may explain the higher catch rates in the late 1990s. Meanwhile, harvest rates in Florida Bay have remained quite stable since 1989 when bag limits of 1 fish per person were imposed (Figure 4). Increased size limits (12" to 18") and a closed season imposed on the fishery in 1985 accounted for the large declines in harvest rate after 1985 (Figure 4); however, the sharp decline during the early 1980s suggests overfishing and/or poor recruitment. If red drum populations were overexploited resulting in reduced recruitment during the late 1980s, the restrictive regulations mentioned above may have allowed the offshore stocks to rebuild, resulting in increased recruitment to the fishery beginning in 1989.

It is unknown if any environmental factors aided in the recovery as no statistically significant relationships were found between catch rate and any of the environmental variables (rainfall, water level, and salinity). However, Tilmant et al. (1989) found a positive correlation between the number of age 1 fish and rainfall from the previous year suggesting that below average rainfall in 1984 - 1985 may have been responsible for the poor recruitment during those years. In laboratory studies, Holt et al (1981) found that survival and growth for larval and juvenile red drum was greatest at 30‰.

The total estimated catch of red drum decreased steadily from 1985 - 1988, when the fishery was closed, following the same trend seen in CPUE (Figure 5). The total estimated catch has been recovering since, with a large jump in 1996 when year-round harvest was permitted, increasing the fishing effort (Figure 5). There was a 14-year high in 1997 with 45,979 fish caught. The estimated total harvest of red drum had been steadily increasing after 1992 until a slight drop in 1998 (Figure 5). The 9,227 fish kept in 1997 represents the highest value since the mid-1980s when the fishery collapsed (Figure 5).

The total estimated recreational fishing effort for red drum in Florida Bay ranged from a low of 58,093 angler-hours in 1988 to a high of 154,227 angler-hours in 1997, which represents about 2.5 times the fishing effort in 1988. A statistically significant linear relationship ($r = 0.833$, $N = 14$, $p < 0.0001$) was found between yearly effort from 1985-98 and the resultant catch, suggesting that the increase in fishing effort did not greatly impact the red drum stock (Figure 6).

ACKNOWLEDGEMENTS

We thank the ENP Flamingo Rangers who provided daily trailer counts for the catch and harvest estimations. We thank Joe O'Hop and Robert Muller at the Florida Marine Research Institute (St. Petersburg) for coordinating the 1997-98 data entry. We would also like to acknowledge George Schardt and Dewitt Smith of the South Florida Natural Resources Center (SFNRC-ENP) for providing the rainfall, water level, and salinity data, as well as SFNRC-ENP computer program managers Dave Buker and Darrell Tidwell.

LITERATURE CITED

Beaumariage, D.S. 1969. Returns from the Schlitz tagging program including a cumulative analysis of previous results. Florida Department of Natural Resources Tech. Ser. No. 59. 38 pp.

Boesch, D.F. 1993. Deterioration of the Florida Bay ecosystem: an evaluation of the scientific evidence. National Fish and Wildlife Foundation. Report to the Interagency Working Group on Florida Bay. Washington, D.C.

Davis, G.E. 1980. Changes in the Everglades National Park red drum and spotted seatrout fisheries, 1958-1978: Fishing pressure, environmental stress, or natural cycles? Proc. Colloquium on the Biology and Management of Red Drum and Seatrout. 81-87.

Davis, G.E. and E.B. Thue. 1979. *Fishery Data Management Handbook.* Everglades National Park, South Florida Reasearch Center. Homestead, FL. Report T-546. 80 pp.

Fore, P.L. and T.W. Schmidt. 1973. Biology of juvenile and adult snook, *Centropomus undecimalis*, in the Ten Thousand Islands. Ecosystem Analysis of the Big Cypress Swamp and Estuaries. EPA 904/9-74-002, Athens, GA.

Higman, J.B. 1967. Relationships between catch rates of sportfish and environmental conditions in Everglades National Park, Florida. *Proc. Gulf and Carib. Fish. Inst.* **19**:129-140.

Holt, J., R. Godbout, and C.R. Arnold. 1981. Effects of temperature and salinity on egg hatching and larval survival of red drum, *Sciaenops ocellatus. U.S. Natl. Mar. Fish. Serv. Fish. Bull.* **79**(3):569-573.

Ingle, R.M., R.E. Hutton, and R.W. Topp. 1962. Results of the tagging of saltwater fishes in Florida. Fl. Board Conser. Mar. Res. Lab. Tech. Ser. 38. 57pp.

Iversen, E.S. and A.W. Moffet. 1962. Estimation of abundance and mortality of a spotted seatrout population. *Transactions of the American Fisheries Society* **91**: 395-398.

Malvestuto, S.P. 1983. Sampling the recreational fishery. Pages 397-419 in: L.A. Nielsen and D.L. Johnson (eds.) *Fishery Techniques*. American Fisheries Society, Bethesda, MD.

McMichael, R.H., K.M. Peters, and G. Parsons. 1987. Early life history of common snook, *Centropomus undecimalis*, in Tampa Bay, Florida. Proc. Third Snook Symposium. 11 pp.

Muller, R.G. and M.D. Murphy. 1997. A generalized stock assessment of common snook, *Centropomus undecimalis*. FMRI/FDEP, St. Petersburg, FL. 24 pp.

Rago, P.J. and C.P. Goodyear. 1985. Estimation of mortality rates and population size from tagging studies in Everglades National Park, 1984-1985. U.S.F.W.S. Fishery Research Lab, Gainesville, FL. 34 pp.

Rutherford, E.S., T.W. Schmidt, and J.T. Tilmant. 1989a. Early life history of spotted seatrout (*Cynoscion nebulosus*) and gray snapper (*Lutjanus griseus*) in Florida Bay, Everglades National Park. *Bull. Mar. Sci.* 44(1):49-64.

Rutherford, E.S., J.T. Tilmant, E.B. Thue, and T.W. Schmidt. 1989b. Fishery harvest and population dynamics of spotted seatrout, *Cynoscion nebulosus*, in Florida Bay and adjacent waters. *Bull. Mar. Sci.* 44(1):108-125.

Rutherford, E.S., J.T. Tilmant, E.B. Thue, and T.W. Schmidt. 1989c. Fishery harvest and population dynamics of gray snapper, *Lutjanus griseus*, in Florida Bay and adjacent waters. *Bull. Mar. Sci.* 44(1):139-154.

Rutherford, E.S., E.B. Thue, and D.G. Buker. 1983. Population structure,food habits, and spawning activity of gray snapper, *Lutjanus griseus*, in Everglades National Park. Report 83/02. 41 pp.

Rutherford, E.S., E.B. Thue, and D.G. Buker. 1982. Population characteristics, food habits and spawning activity of spotted seatrout, *Cynoscion nebulosus*, in Everglades National Park. Report T-668. 48 pp.

Schirripa, M.J. and C.P. Goodyear. 1994. Simulation modeling of conservation standards for spotted seatrout (*Cynoscion nebulosus*) in Everglades National Park. Bull. Mar. Sci. 54(3):1019-1035.

Schmidt, T.W., G.A. Delgado, and J. Kalafarski. 1999. Annual Fisheries Report. Everglades National Park, SFNRC, Homestead, FL. 39p.

Schmidt, T.W. and M.B. Robblee. 1994. Causes of fish kills in the Flamingo area of Everglades National Park. Bull. Mar. Sci. 54(3):1083.

Tabb, D.C., D.L. Dubrow, and R.B. Manning. 1962. *The ecology of northern Florida Bay and adjacent estuaries*. Florida State Board of Conservation Tech. Ser. No. 39. 81 p.

Taniguchi, A.K. 1980. Effects of salinity, temperature, and food abundance upon survival of spotted seatrout eggs and larvae. Proc. Colloquium on the Biology and Management of Red Drum and Seatrout. Gulf States Marine Fisheries Commission. 16 pp.

Thayer, G.W., A.B. Powell, and D.E. Hoss. 1998. Response of larval, juvenile, and small adult fishes to changes in environmental conditions in Florida Bay: a decadal comparison. Proc. Florida Bay Science Conference, Miami, FL.

Thue, E.B., E.S. Rutherford, and D.G. Buker. 1982. Age, growth, and mortality of the common snook, *Centropomus undecimalis*, in Everglades National Park, Florida. Report T-683. 32 pp.

Tilmant, J.T. 1989. A history and an overview of recent trends in the fisheries of Florida Bay. *Bull. Mar. Sci.* **44**(1):3-22.

Tilmant, J.T., E.S. Rutherford, and E.B. Thue. 1989. Fishery harvest and population dynamics of red drum (*Sciaenops ocellatus*) from Florida Bay and adjacent waters. *Bull. Mar. Sci.* **44**(1):126-138.

Tilmant, J.T., E.S. Rutherford, R.H. Dawson, and E.B. Thue. 1986. Impacts of gamefish harvest in Everglades National Park. *Proc. Conf. Sci. in Nat'l Parks* **6**:75-103.

Topp, R.W. 1963. The tagging of fishes of Florida, 1962 program. Fl. Board Conser. Mar. Res. Lab. Prof. Pap. Ser. 5. 76 pp.

Tringali, M.D. and T.R. Bert. 1996. The genetic stock structure of common snook (*Centropomus undecimalis*). *Canadian Journal of Fisheries and Aquatic Sciences* **53**:685-694.

Van Os, E., J.D. Carrol, and J. Dunn. 1980. Creel census and the effects of freshwater discharges on sportfishing catch rates in the St. Lucie Estuary, Martin County, FL.

Wakeman, J.M. and D.E. Wolschlag. 1977. Salinity stress and swimming performance of spotted seatrout. *Proc. Annual Conference of Southeast Association of Fish and Wildlife Agencies* **31**:357-361.

Yokel, B.J. 1966. A contribution to the biology and distribution of red drum, *Scianops ocellatus*. M.S. Thesis. University of Miami. Coral Gables, FL. 160 pp.

Mapping Benthic Habitats Using Side Scan Sonar

RICHARD S. APPELDOORN[1], JOSE A. RIVERA[2], and MARTHA C. PRADA[1]

[1]*Department of Marine Sciences*
University of Puerto Rico
Mayaguez, Puerto Rico 00681-9013
[2] *NOAA/National Marine Fisheries Service*
Miami Laboratoy-Biodiversity and Protected Resources Division
Cabo Rojo, Puerto Rico

ABSTRACT

Detailed, but large scale habitat mapping is necessary if fisheries researchers are to investigate habitat-species distributions on spatial scales relevant to fisheries management and marine conservation, and identify essential fish habitats and the qualities that underlie their importance. The latter will include both the immediate structural qualities of a given area and the linkages among surrounding habitats. Knowledge of the underlying structure of habitat function should allow simple rules to be developed for future identification and protection of critical habitats. Despite over 40 years of concentrated academic investigation in the area of La Parguera, Puerto Rico, including geological mapping, there is no comprehensive habitat map suitable for large-scale biological studies. Developing such a map is now feasible using remote sensing and geographic information system (GIS). We are using side scan sonar (SSS) to develop a habitat map extending from the shoreline to the edge of the insular platform and covering over 20 nautical miles square (nm^2). SSS is advantageous over airborne remote sensing in its greater depth range and greater resolution. The SSS associated navigation equipment (computer, software, DGPS) and electric winch are mounted on a 22-ft vessel. A 300 kHz transducer is towed over 100-m wide paths to collect bottom images. To create larger views, georeferenced mosaics are generated from individual images using GIS technology. To date we have identified broad areas of habitat and located unknown patch reefs in otherwise soft-sediment areas. At 300 kHz, metal-frame fish traps were not detected, although evidence of their effect on corals were. Although SSS technology is valuable, its success requires proper logistical set-up, plus expertise in computers and electronics.

KEY WORDS: Habitat mapping, active sensors, side scan sonar

INTRODUCTION

The spatial distribution of benthic habitats and their complexity are key factors affecting fish distribution and abundance (McClanahan 1994). This relationship is especially noted in coral reef ecosystems (Choat and Bellwood

1991, Appeldoorn et al. 1997, Friedlander and Parrish 1998), where fish live and move among many different habitats (Coreless et al. 1997, Roberts 1997). Basic information on the distribution and complexity of benthic habitats is often unavailable for the majority of reef areas due to their large expanse and because it requires expensive and sophisticated equipment to map them. Remote sensing techniques offer a viable option to fill this gap for large areas, determining not only the location and amount of the distinct benthic habitats, but also how these habitats are distributed and for allowing, in some way, determination of habitat connectivity. However, in tropical regions, and specifically in coastal waters, the use of traditional optic sensors is very restricted because of rapid light absorption by the water column and phytoplankton, and the presence of colored dissolved organic matter, even in coral reef systems with relatively clear water. Active sensors, such as SSS, can solve this problem, offering a high resolution depiction of the bottom regardless of water transparency and depth.

Side scan sonar, first used in 1963, is now commonly employ in searching for shipwrecks, elaboration for nautical charts, and for seabed imaging worldwide (Fish and Carr 1990, Mazel no date). An example of extensive SSS use is the Gloria Project, which for more than 25 years was utilized to map geological features on the sea floor in all major oceans, including the U.S. Exclusive Economic Zone (U.S. Geological Survey 1987). However, Gloria is not a compact SSS. Few scientific works have made use of SSS in shallow waters. For example Siljestron et al. (1996) successfully mapped seagrass meadows of *Posidonia* and *Cymodocea* in Mediterranean waters. In Australia, Harris and Davis (1989) mapped submerged reefs using a 100-kHz SSS. SSS is also used in fish quantification (Pitched et al. 1996). In Puerto Rico, a section of the northeast shelf was mapped using SSS to determine distribution of sediment types (Rodriguez et al. 1998). Recently, work mapping benthic habitats was conducted in estuarine lagoons around San Juan Bay (Rivera, 1999). With the information collected from that project a detailed map of benthic habitats is being generated and will be available for multiple research purposes, especially fisheries research.

We are currently conducting a SSS survey of the insular shelf off La Parguera, Puerto Rico. The survey covers a 3-Nm wide section extending from the shoreline (Playita Rosada Isla west to Isla Cueva) out to the edge of the insular shelf. Despite over 40 years of concentrated academic investigation in the area of La Parguera, Puerto Rico, including geological mapping, there is no comprehensive habitat map suitable for large-scale biological studies. Our goal is to produce a map detailing the abundance and spatial distribution of benthic habitats. This includes both the immediate structural qualities of a given area and the linkages among surrounding habitats. Such information can be used to improve fish abundance surveys through habitat stratification, to identify and quantify potential fish habitats, and to study the relationship between habitat and their functional role in fisheries production. Knowledge of the relationship between habitat structure and function should allow simple rules to be developed for future identification and protection of critical habitats.

The purpose of this paper is to present how we have adapted methods and technology of SSS surveying, so that these can be applied to marine and estuarine coastal tropical regions. We illustrate this with preliminary results from the La Parguera survey.

METHODS

The study area is located off La Parguera (17°58'N, 67°02'W) in southwestern Puerto Rico (Figure 1). Collection of SSS imagery was done over a period of 40 days during 1998 and 1999.

A 22-ft fiberglass Privateer boat with a bow cabin and a four-cylinder inboard engine and stern drive was used to deploy the SSS and navigation systems. The boat is equipped with four gel cell 12V-103 amp batteries arrayed into two battery banks. Each bank is connected in series to supply 24V-DC. The 24V supply is connected to a Trace R/V inverter that outputs 120V AC power. Due to the creation of electromagnetic interference on SSS image, this inverter was exchanged with a Statpower Prosine^{tm} 1000 inverter. The 120V power is supplied to the data acquisition computer and to a portable navigation computer when needed. However, 12V is supplied directly from one battery to the Differential Global Positioning System (DGPS) signal and to the 1000 nit liquid crystal display (LCD) connected to the SSS imagery acquisition computer. The 1000 nit LCD monitor increased imagery display visibility in daylight conditions and reduced power consumption. The battery banks are selected utilizing a battery bank selector switch. During daylight hours, two 12V-75watt, 4.5 amp solar panels connected in series and a charge controller provide recharging power to both battery banks. When the boat is ashore the Trace inverter, once connected to a 120V source, can also recharge both battery banks overnight.

Two additional lead acid 12V-103 amp batteries are connected into a 12V-DC bank. This bank provides the starting power for the boat engine and for the electrical winch, depth sounder, cabin lights, bilge pump and a cigarette plug adapter used to supply power to the navigation computer when needed as backup. The 12V bank also has a battery selector switch. These batteries are independent from the 24-DC battery bank. Once running, the engine alternator/battery charger provides recharging power for these batteries. The engine battery bank is independent of the SSS acquisition battery bank in order to avoid the possibility of introducing electromagnetic interference to the SSS imagery from the engine.

A 300-kHz Marine Sonic Technology Ltd. transducer (commonly known as the "fish") was towed over a 100-m wide swath. The fish was kept 3–5 m off the bottom on average to avoid collision with common bottom features such as patch reefs, hard bottom ledges, etc., that usually protrude 2-3 m off the bottom. Where the bottom topography was highly irregular, the fish was raised further from the bottom (7-8 m). Keeping the fish 3-5 m off the bottom also ensures the needed angle of view for the tranducer to reach its 50-m range on each channel side.

A NMEA 0183 navigation serial data output string with differential corrections is provided from a Magellan DGPS both to the navigation system software (Hypack V8.2) and to the SSS image acquisition software (Marine Sonics Technology Ltd., Sea Scan PC V5.18). This provides exact geo-referencing for the SSS images upon recording to hard disk. The offset between the DGPS antenna location and the fish location is not corrected for during real time data acquisition. This correction is usually accounted for in the post-processing of SSS imagery.

In Sea Scan PC, there are options to adjust the gain, to make corrections for the offset, to make annotations and to record depth information. Some filters, and analysis of the images, like estimations of areas and heights are also possible using the reviewing software, Sea Scan PC Review. Sea Scan PC automatically creates consecutive images of 1,000 by 512 lines that result in a image file of 1.2 megabytes in size with its respective navigation information, but the software does not offer the possibility to merge images into larger mosaics.

Prior to starting the survey, a detailed planning session was conducted using Hypack survey planning software in order to define the study area and the determine the transect lines to be followed with the SSS. As a result, 19 navigation files (Figure 1) were created. Each file covers a total of 1 Nm and includes 24 transect lines, 75 m apart. With this design, we have 12.5 m of overlap on both sonar channels (left and right), keeping a constant range of 50-m on each side. Vessel speed ranged between 2.8 – 3.8 knots, but much slower speed was required when depths were more than 21 m in order to reduce cable drag and lower the fish. A continuous log of the position was recorded in the navigation computer.

Available post-processing image filtering algorithms in Sea Scan PC Review were applied to improve the quality of the single images and eliminate electromagnetic interference when present. Commonly applied filters are spike (to eliminate random points), plus (to add brightness), minus (to reduce brightness) or expand (to enhance contrast).

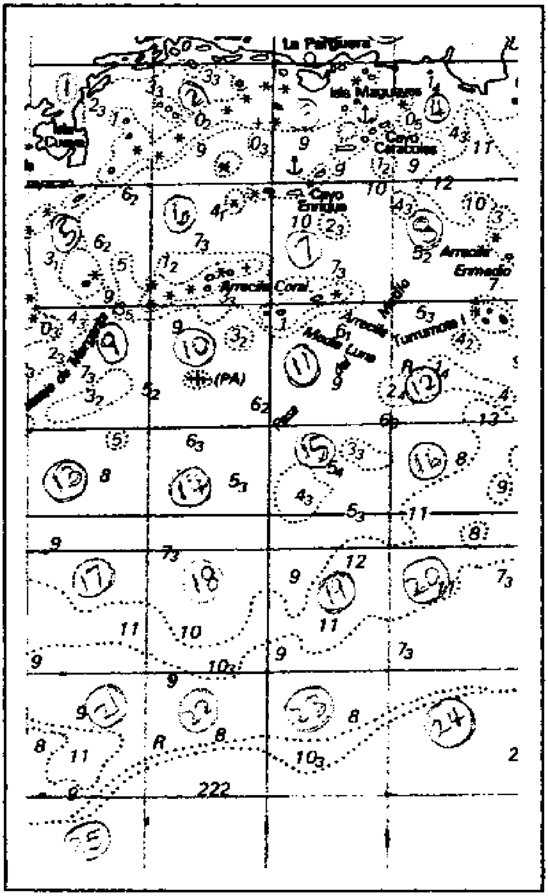


Figure 1. Area surveyed with SSS off La Parguera, southwest coast of Puerto Rico. Area extends from shore to shelf edge. Each square has 1 nm². Right most column was not included in the survey.

Further processing takes single images and merges these to form large mosaics. This is done using Caris V4.3.1, a GIS software from Universal, Ltd. This program has a special routine in its sonar module for processing Marine Sonic Technology (MST) images. The process starts by converting files from MST format into Caris format. Then, left and right channel images are joined by removing the water column and producing a geometrically correct bottom image. Mosaics were usually made by placing the brightest image on top when adjacent transects overlapped. The generated mosaic is then exported to the Caris - GIS module to make the initial visual interpretation of different sonar echo return signatures and to define polygons with similar echo return texture. These polygons are then related to habitat types based on interpreter experience and ground truthing information.

Ground truthing was used to check the accuracy of the interpretation and to revise the delimitation of the different polygons on the mosaic. Ground truthing is done using a tethered black and white underwater video camera in shallow or deep-water areas covered with sand, algae or grass. However, extensive ground truthing is also done by diving to get a general perception of the area and mosaic generation and reach areas in which the underwater video was not able.

The processing phase requires an additional computer with large memory (512 MB RAM) and hard disk storage capacity (>60 GB) due to the large amount of data collected and produced. For example, one file covering 1 Nm^2 and containing around 250 megabytes of raw imagery results in 1 gigabyte of processed information.

RESULTS

To date, 364 Nm of SSS transect lines have been surveyed out of a total of 442 Nm, representing 82% of the total study area. Of the 19 mosaics to be made, 63% have been processed.

At this moment, we have obtained around 4000 single images. Two examples are shown in Figures 2 and 3, where different types of bottom are clearly identifiable. Strong reflections produced by reef platforms, patch reefs or big isolated coral heads are easy to identify. Mixed coral-gorgonian areas are also possible to identify with more detailed observation. On sandy bottoms it is possible to observe ripple marks when present. Different densities of soft algae or the presence of silty-sand are also possible to detect. Reviewing bottom images at this scale has already revealed interesting features. We have routinely found uncharted patch reefs of a variety of sizes located in otherwise soft-bottom areas. The most interesting observation was the occurrence of deeper water (18 m) coral patches surrounded by halos of dense *Halophila decipiens* mixed with algae, primarily *Caulerpa racemosa* var. *macrophysa* and some *Udotea* sp. and *Dyctiota* sp. (Figure 2). Such halos have not been previously reported in the ecological literature. Figure 3 is from an area of high fish trapping activity and shows a coral reef area believed impacted by a fish trap or anchor drag. With such high quality definition, we found SSS technology useful for providing high resolution images of different types of benthic marine habitats. It also gave us information on structure and real size of those features. One disappointment has been our inability to distinguish the numerous fish traps in the area with the 300-kHz transducer. However, on occasion it was possible to observe the line attached to the trap. We believe using different sonar frequency can help identify spot fish traps more efficiently.

The large area views obtained from mosaics (Figures 4 and 5) complement the detailed single-image information with a view of the spatial distribution patterns among benthic habitats, which will allow us to establish the linkages between them. Our factor interfering with the interpretation of the bottom images at this scale is the presence of the discrete parallel lines along the transect path due to surface reflection. These lines are located in variable positions over the bottom image and cannot be eliminated by available filters.

At present, we are defining all the possible identifiable features and will join them into a classification system of benthic habitats that can be applied to similar environments. As an example, we made gross estimations on percentage coverage of general features for some mosaics (Table 1). From this we can observe that on the La Parguera shelf, sand-algae is the most abundant habitat, covering 38.38% of the area analyzed. This percentage increases to 50.55% if we include the silty-sand and algal plains. Sometimes, fleshy algae form dense patches with a distinct dark pattern on SSS image, especially observed in intermediate depths. This type of habitat covered around 4.4% of the total area analyzed. In deeper waters (20 - 24 m), sand channels between extensive areas of hard bottom can be found close to the shelf edge (Figure 5).

Hard bottom habitat is second in abundance with 17.49%, mostly located close to shelf edge (Figure 5). Widely dispersed corals cover a total of 9.97% of the area (Figure 4). Overall percentage of coral habitats may increase after we acquire SSS images from the numerous emergent reefs present in the area. Hard bottom and corals are two related habitats that need to be delimited carefully based on detailed observations and intense ground truthing, in order to increase the accuracy of interpretation of the SSS imagery. Seagrass habitat (*Thalassia, Syringodium* and *Halophila*) is present in shallow and deep waters but they have lower percent cover (14.73%).

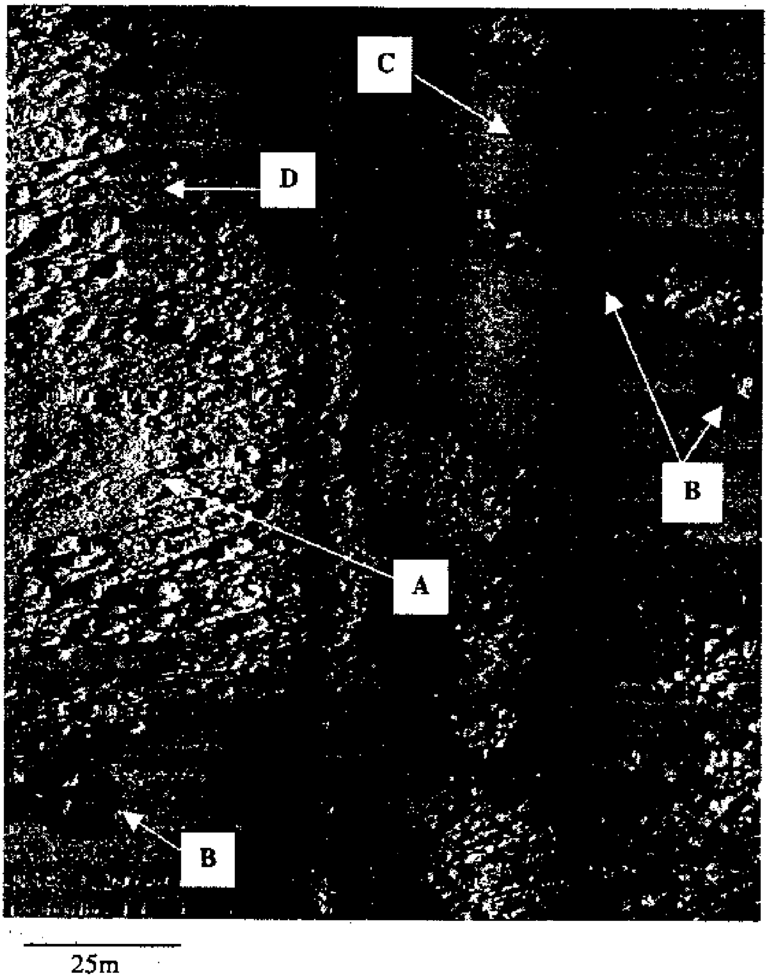


Figure 2. Edited single SSS image from 17°56'N, 67°03'W with water column removed. Surface echo has been edited out for easy interpretation. A. Strong rugose reflection shows coral reef habitat. B. Patch surrouned by halo of dense *halophila decipiens*. C. Smooth sand and algae plain. D. Sand ripple ridges.

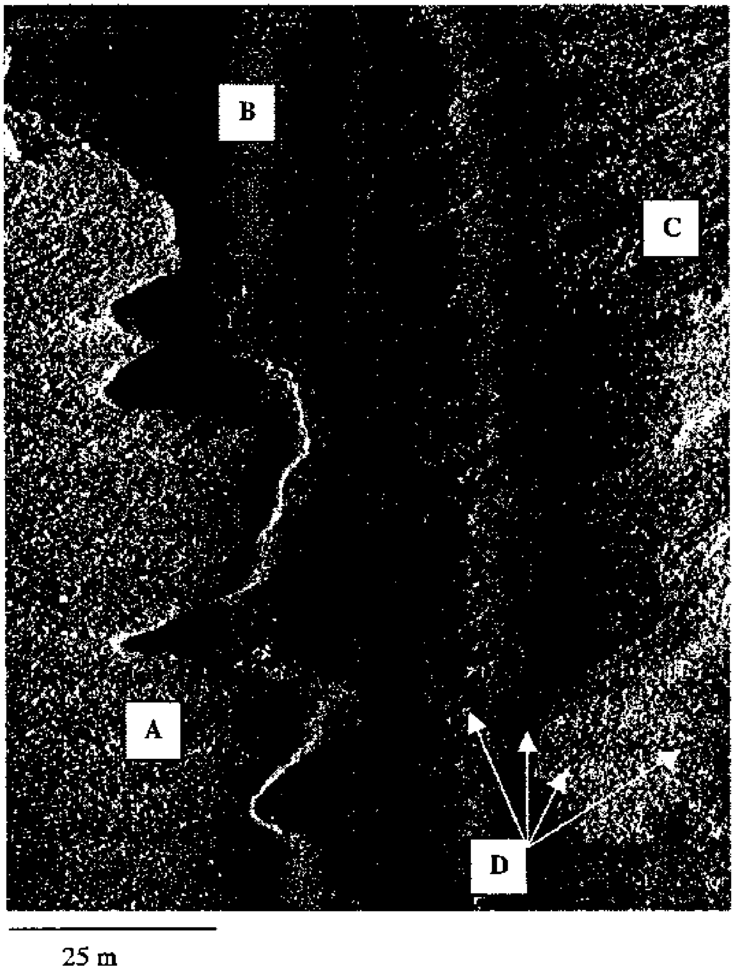


Figure 3. Edited single SSS image from 17°53'N, 67°04'W with water column removed. Surface return echo has been edited out for easy interpretation. A. Isolated coral colonies over hard bottom. This platform has 5 m height or relief. B. Sand and algae with small isolated corals. C. Coral reef formation. D. Coral reef area believed impacted by fish trap or anchor drag. Notice linear patterns on edge and at rieght of center. This area is known fish trap setting area.

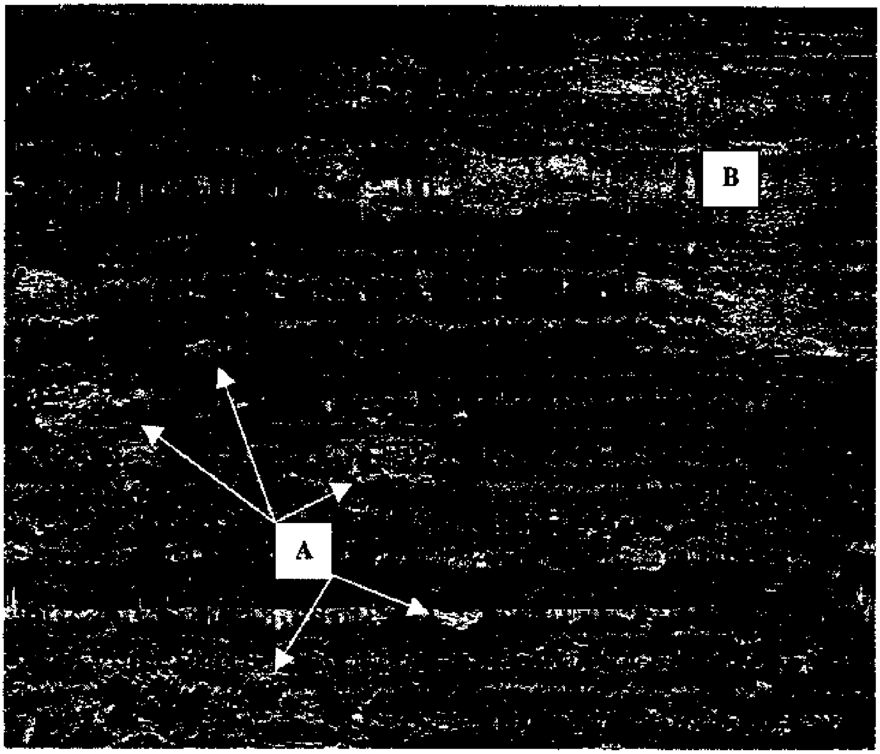


Figure 4. Section of a mosaic No. 10 (center position 17°55'N, 67°03'W) in 18 - 20 m depth. A. Uncharted patch reefs in a variety of sizes on sandy bottom. B. Coral reef area.

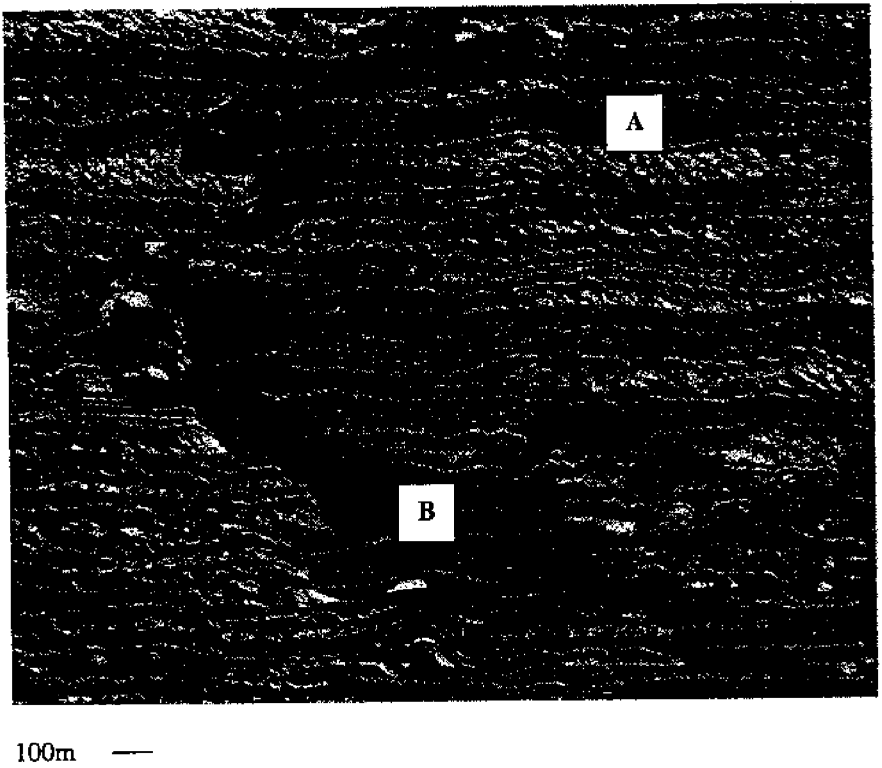


Figure 5. Sub-section of mosaic No. 22 (Center position 17°52'N, 67°03'W) in 20 - 24 m depth. A. Isolated coral and gorgonians on hard bottom. B. Sand channel.

CONCLUSIONS

Side scan sonar can be used successfully to describe and quantify marine habitats. Our preliminary results, show sand-algal plains as the dominant habitat, followed by hard bottom and corals.

Detailed information on habitat types using SSS technology allowed us to find abundant and uncharted coral reef patches over sand bottom. We also found unreported occurrence of patch reefs surrounded by halos of dense *Halophila decipiens*, which offer possibilities for new ecological research.

Despite our inability to detect fish traps, we were able to determine alterations on hard bottom recorded as a linear pattern on SSS images. These linear patterns are also observed in shallow seagrass areas where antropogenic effects such as anchor drags or prop scars are common.

Methods presented here involve a combination of different technologies, instruments, and software and require expertise in electrical and electronic techniques. It is very important to have trained personnel to facilitate efficient use and repair survey equipment.

Large area mosaics provide the basic information to establish patterns of spatial distribution of benthic habitats and how they interconnect. However, improvements in the processing techniques for building mosaics are needed to enhance the amount of information obtained and facilitate subsequent analysis of bottom features.

Considering that SSS image resolution is a function of frequency and range, fish traps probably can be detected using a higher frequency (e.g. 600 or 1,200-kHz) transducer. However, if the trap is built with plastic or other material with low sound reflectance, it will still be difficult to detect mesh. Furthermore, higher frequency transducers must be positioned closer to the bottom, which may reduce range of coverage (path width) and risk impact with bottom features.

ACKNOWLEDGEMENTS

This work was supported by the U.S. National Marine Fisheries Service, Habitat Conservation Division, the Caribbean Fisheries Management Council, the University of Puerto Rico, Sea Grant College Program (R/FA-06-3-94 and MPRD-14) and by Colciencias (Colombian Institute for the Development of Science and Technology).

LITERATURE CITED

Appeldoorn, R. S., C. Recksiek, R. Hill, F. Pagan, G. Dennis. 1997. Marine protected areas and reef fish movements: the role of habitat in controlling ontogenetic migration. *Proc. 8th Int. Coral Reef Symp.* **2**:1917-1922.

Corless, M., B. Hatcher, W. Hunte, S. Scott. 1997. Assessing the potential for Marine Reserves to adjacent fished areas in the Soufriere Marine Management Area, St. Lucia. *Proc.Gulf Carib. Fish Inst.* **47**:71-97.

Choat, J.H., D.R. Bellwood. 1991. Reef fishes: Their history and Evolution. Pages 39-66 in: P.F. Sale (ed.) *The Ecology of Fishes on Coral Reefs.* Academic Press, Inc. San Diego, CA.

Fish, J.P., A. Carr. 1990. ***Sound water images, a guide to the generation and interpretation of the Side Scan Sonar data.*** Lower Cape Publishing. Orleans, MA. 189 pp.

Friedlander, A.M., J.D. Parrish. 1998. Habitat characteristics affecting fish assemblages on a Hawaiian coral reef. *J. Exp. Mar. Biol. Ecol.* **224**:1-30.

Harris, P.T., P.J. Davis. 1989. Submerge reefs and terraces on the shelf edge of the Great Barrier Reef, Australia. Morphology, occurrence and implications for the evolution. *Coral Reefs* **8**:87-98.

McClanahan, T.R. 1994. Kenyan coral reef lagoon fish: effects of fishing, substrate complexity and sea urchins. *Coral Reefs* **13**:231-241.

Mazel, C. No date. ***Side Scan Sonar, record interpretation. Training manual.*** Klein Associates, INC. Salem, New Hampshire. 120 pp.

Miller, R., F. Dwan, C. Cheng. 1991. Digital pre-processing techniques for Gloria II Images. *Geomarine Letters* **11**:23-31.

Roberts, C. 1997. Connectivity and Management of Caribbean Reefs. *Science* **278**(21):1454-1457.

Pitcher T., O. Arve Misund, A. Ferno, B. Totland, V. Melle. 1996. Adaptive behaivour of herring schools in the Norwegian Sea as reveled by high-resolution sonar. *ICES J. Mar. Sci.* **53**:449-452.

Rodriguez, R., W. Schwab, M. Carlo, R. Thieler. 1998. Marine Geologic Map of Northeastern Insular shelf. USGS/UPR/Duke University. *Geologic investigations series* I-2612.

Siljestron, P.A., A. Moreno, J. Rey. 1996. Mapping seagrass meadows of ***Posidonia*** sp. and ***Cymodocea*** sp. in Mediterranean waters. ***J. Photogram. Remote Sens.*** **51**:308-315.

U.S. Geological Survey-EEZ-SCAN 85 scientific staff. 1987. Atlas of the U.S. Exclusive Economic Zone, Gulf of Mexico and Eastern Caribbean Areas. ***U.S. Geological Survey Miscellaneous Investigations Series I-1864-A.*** 104 pp.

Temporal Change of a Coral Reef Community in the South Mexican Caribbean

J. RODRIGO GARZA-PEREZ and JESUS E. ARIAS-GONZALEZ
Lab. de Ecologia en Ecosistemas Arrecifales Coralinos
CINVESTAV-IPN, Unidad Merida.
Carr. Antigua a Progreso Km 6 A.P.73 Cordemex
C.P.97130. Merida, Yucatan, Mexico

ABSTRACT

The coral reefs in the South Mexican Caribbean are considered to be in a healthy condition, but they are about to suffer a dramatic increase in the anthropogenic pressure on their resources. The cause of the increase in the use of resources is the rapid development of the zone, as the new "ecotouristic" attraction of Quintana Roo.

The health of the fore reef coral community at Mahahual Quintana Roo, in Mexico, was assessed using a videotransect method. Videotransects were filmed for the months of: May 1997, July 1998, and November 1998, after hurricane "Mitch" and a massive bleaching event. The community analysis was classified by its 'Morph-Functional Groups' values of cover, dominance, diversity, and eveness. MFG's have been shown to be an accurate descriptor of coral communities. The change of the structure of the coral reef community due to anthropogenic pressure could be stated from year to year, as well as the consequences of the stochastic events which occur months later.

This study is part of a comparative analysis of several coral reef communities in Quintana Roo, in order to gather information about the state of the resources and to suggest management strategies.

KEY WORDS: Coral communities, reef health assessment, videotransects

INTRODUCTION

Coral reef communities are important providers of food, cover and substrata for all the organisms associated to them, including economically important species, such as lobster, conch, snappers and groupers. They also preserve the coastline and all the ecosystems associated to it by suppressing energy from waves and ocean currents.

The natural and artificial processes influencing the coral reefs define their communities structure. Depending on the intensity and duration, natural pressures can change the topography of the reef by breaking and tearing down colonies, in the case of hurricanes (Rogers 1993, Connell 1997). In the case of severe spreading of coral diseases and/or bleaching, they can produce shifts in the dominant species (Aronson et al. 1998). Artificial processes are proven to change the community structure as well. As anthropogenic impacts due to urban development, agricultural runoffs, black water discharges, overfishing and tourism (Talge 1990, Hawkings and Roberts 1993, 1997, McClanahan 1995, Jennings and Polunin 1996, Arias-Gonzalez 1998).

The study area is located in the southern coast of the state of Quintana Roo in Mexico, one of the best preserved places along the Mesomerican Reef. However, this coast is one targetted for tourism development, namely: "The Mayan Coast". The importance of assessing the state of condition of the coral reef communities, relies on linking the reef's health state with the natural and anthropogenic pressures affecting the ecosystem. This would help to implement the management of the reef resources and to preserve the coral community structure. The present study is part of a multi-disciplinary research program to characterize the coral reefs of the Mexican Caribbean and assess their health, which has a critical importance for the knowledge of the resources and their correct management. It will also provide a baseline for future monitoring efforts and environmental impact assessments.

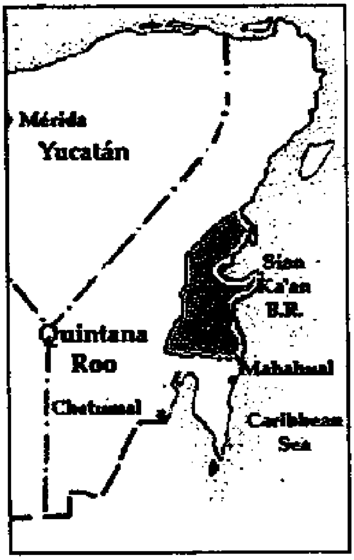


Figure 1. Mahahual localization

METHODS

The study area is located in Mahahual, which is a strategic place (Figure 1) due to its proximity with Sian Ka'an Biosphere Reserve, Chetumal the capital city of the state, and the tourist zone of Belize. This location has one of the better-developed reefs in the area. Mahahual's biological richness and diversity makes it a perfect place to monitor and assess the change in coral communities, due to natural and anthropogenic pressures as fisheries and tourism. Mahahual is a small fishermen village with no more than 300 inhabitants; the human activities are artisanal fisheries and SCUBA diving year-round, especially during spring break and summer when the population can reach the 600 - 800 with tourists. There is an important number of fishing vessels, but they mainly focus their activities in Banco Chinchorro. Mahahual has an extensive but shallow reef lagoon, limited by the well-developed reef crest. The first butress zone in the fore reef starts approximately at a depth of 7 m, and ends approximately at 16 - 20 m followed by a sand bar of variable width and the slope reef staring at a depth of 20 - 25 m.

The videotransect method proposed by Aronson and Swanson (1997) was considered to meet the requirements for assessing the condition of the coral reef community of Mahahual. This method was developed for the Caribbean spur and groove biotope, also known as buttress zone. The 10 m depth, in the fore reefs first step, which corresponds to the same reef habitat, was chosen for this study. The study site is located approximately at 18°42'42"N, 87°42'12"W, and three videos were taken on the months of May 1997, July 1998 and November 1998. The sampling method consisted in videotaping 10 random 25 x 0.4m transects of the reef substratum, for a total sampled area of 100 m^2. The transects were laid on the transversal axis of the spurs in a range of approximately 500 m. In this study we used a Hi-8 Sony Handicam (CCDTR4000) and a StingRay video housing equipped with a wide angle lens. The homogeneity of the recorded image area was preserved with a weighted tape of 40 cm to keep the camera at a constant height over the substratum.

The recorded video was displayed in a high resolution monitor and fifty non-overlapping still images were reviewed out of each transect, 10 random dots transparent sheets (of 10 dots each) in a rotating pattern were used to sample the still images, in order to obtain a 5,000 dot matrix. The data to collect was the point identification of the morph-functional groups (MFG's) present on the videos. This MFG's were used successfully to assess coral reef communities by Oxley and Osborne (1997) in the GBR, Australia and by Maldonado-Gasca (1998), Caceres-G. Canton (1998) and Membrillo-Venegas (1999) in reefs throughout the Gulf of Mexico and the Mexican Caribbean. The method has enough statistical power according to studies carried out by Aronson and Swanson (1997) in Florida, and Carleton and Done (1995) in the GBR, Australia, to detect small variations in the sampled communities.

The point identification data were transformed to percent cover for each MFG, in order to obtain the McNaughton Community Dominance Index. The percent cover values were transformed logarithmically to increase their normality and homogeneity, then, used to obtain Simpson and Shannon-Weiner Diversity Indexes, to perform a one-way ANOVA and an *a posteriori* Newman-Keuls Multiple Comparisons Test, in order to separate the three samples.

RESULTS

The Mahahual coral reef community was classified through the analysis of videotransects utilizing thirty-five MFG's - nine related to scleractinean corals, two related to hidrocorals, eight to octocorals, five to sponges, three to algae, and five related to bleach conditions. In addtion, we included four *Inert Substrata Types* (IST's) (Table 1). The classification of the MFG's present in the study area was an adaptation based on the criteria followed by Humann (1992, 1993) for Florida, Caribbean and Bahamas and by Osborne and Oxley (1997) for the Indo-Pacific. Species included in each MFG are not unique or exclusive, they could vary upon genetic criteria or grow morphs, in this study the considered species are the probably observed and/or identified in field, or in the videotransects. Scientific names were taken from Humann identification manuals (1992, 1993).

For May '97 the 5 dominant groups with a cover value of 69.06%, were IST Rubble, Encrusting Coral, IST Rock, Brown Algae and Leaf Coral. In this sample the two least groups were: Pillar and Flower-cup Corals.

In July 98 the 5 dominant groups accounted for 62.94% of the total cover and they were: IST Rock, Encrusting Coral, IST Rubble, Brown Algae and Feather Octocoral. The least dominant groups in this sample were Column Sponge and Flower-cup Coral.

For November '98, dominant MFG's accounted for the 73.36% of total cover, and they were: IST Rubble, Encrusting Coral, IST Rock, IST Dead Coral and Brown Algae. The least dominant groups in this sample were Tube Sponge and Fleshy Coral.

For a general characterization of the coral reef community of Mahahual at each sampling time, a Main Group cover analysis was made. The most important Main Groups were considered to be Scleractinean Corals, Algae, Octocorals, Rubble and Dead Coral, because we thought that those Groups could give an accurate general view of the state of condition of the coral community (Table 3). For May '97 the community had the highest scleractinean coral cover of all samples, 31.4%, presented an octocoral cover of 11.98%, an algae cover of 7.52%, a rubble cover 23.12% and a dead coral cover of 3.4%, it also showed a good proportion of calcareous pavement (17.66%) counted in IST Rock.

Table 1. Correspondance of Main Groups to MFG and to Genera/Species

Category /Main Group	Morpho-Functional Group	Genera and/or Species
Scleractinean Coral	Branching Coral	*Acropora palmata, A. cervicornis, A. prolifera*
	Finger coral	*Porites branneri, P. porits,* Genus *Madracis*
	Encrusting- Boulder coral	*Montastraea annularis, Montastraea cavernosa, Montastraea franksi, Montastraea faveolata, Solenastrea bouroni, Siderastrea siderea, S. radians, Agaricia agaricites, Agaricia humilis, Solenastrea hyades, Dichocoenia stokesii, Favia fragum, Porites asteroides, Madracis pharensis, Stephanocoenia intersepta.*
	Brain Coral	*Diploria strigosa, D. clivosa, D. labrynthiformis, Colpophillya nathans, Isophyllia sinuosa, Isophyllastrea rigida, Meandrina meandrites, Manicina areolata*
	Leaf Coral	*Agaricia tenuifolia, Agaricia agaricites* forma *danai*
	Fleshy Coral	Genus *Mycetophyllia*
	Disc-plate Coral	Genus *Scollymia*
	Flower-cup Coral	*Eusmilia fastigiata*
	Pillar Coral	*Dendrogyra cylindricus*
Octocoral	Finger Octocoral	*Briareum asbestinum*
	Branching Coral	Genus *Pseudoplexaura,* Genus *Eunicea, Plexaurella nutans, Muricea muricata*
	Feather Octocoral	Genus *Pseudopterogorgia.*
	Candelabrum Octocoral	*Eunicea mammosa, Plexaura homomalla*
	Fan Octocoral	*Gorgonia flabellum, G. ventalina.*

Table 1 continued.

Category /Main Group	Morpho-Functional Group	Genera and/or Species
	Bushy Octocoral	*Murica pinnata, Muricea elongata, Muricea laxa, Murceopsis flavida, Pterogorgia guadalupensis, P. anceps*
	Encrusting Octocoral	*Erytropodium caribeorum, Briareum asbestinum* (encrusting form)
	Broadfan Octocoral	*Plexaura flexuosa.*
Sponges	Encrusting sponge	*Cliona langae, Anthosigmella varians, Calix podatypa*
	Column Sponge	Genus *Aplysina*
	Tube Sponge	*Pseudocerata crassa, Agelas conifera, Callyspongia vaginalis, C. plicifera.*
	Vase Sponge	*Niphates digitalis, Cribochalina vasculum*
	Ball Sponge	*Speciospongia vesparium, Ircina strobilina, I. felix*
Algae	Brown Algae	Genus *Dyctiota*, Genus *Padina, Lobophora variegata, Stypopodium zonale*, Genus *Turbinaria*
	Green algae	Genus *Halimeda*
	Blue-green Algae	Phylum *Cyanophyta*, "fuzzy" brownish-red algae
Hidrocorals	Branching Hydrocoral	*Millepora alcicornis*
	Leaf Hydrocoral	*Millepora complanata*
Inert Substrata	Sand	100% loose sand ocasionally gravel and/or small macroalgae
	Rock	Calcareous stone pavement, sometimes with light algal turfs, and/or, standing old dead colonies (+1 yr.) w/ or w/out algal turfs of small canopy, w/ or w/out encrusting red alage of *Porolithon pachydermum* type.

Table 1 continued.

Category /Main Group	Morpho-Functional Group	Genera and/or Species
	Dead Coral	Complete colonies, big separated fragments of colonies, or tissue surfice recently dead (6 months +or-, less than a year), generally covered with light microalgae film.
	Rubble	Old dead coral rubble, may have a light macroalgae cover
Other MFG's	Bleached Coral	Alive scleractinean Coral (any group) 100% bleached
	Partially Bleached Coral	Encrusting scleractinen coral bleached in patches or presenting lighter coloration.
	Healthy Encrusting Coral	Encrusting coral without apparent gamage or diseases

The McNaughton Community Dominance Index (order 5) was used to define the dominant groups; the order 5 was chosen because the first 5 dominant groups comprise more than the 60% of the total cover. (See Table 2).

In July 1998, the scleractinean cover decreased slightly to 28.96%, and dead coral cover presented a small increase to 3.18%, in relation to May 1997. The values of cover of the most important Main Groups got homogenized, octocorals, algae, rubble and rock cover values were in the 14% to 17% range.

Mahahual coral reef community in November 1998, showed a major increase of the rubble cover to 29%, a decrease in scleractinean coral cover to 25.24%, and the rest of the Main Groups suffered a decrease, algae and octocorals had a cover of 10.58% each, rock had 10.64% of cover and dead coral had a substantial increase reaching 8.2% of cover.

Table 2. Mc Naughton Community Dominance Index, for samples , for Mahahual site.

MFG's	May 1997	MFG's	July 1998	MFG/s	Nov. 1998
rubble	**23.12**	rock	**16.78**	rubble	**29**
encrust.	**18.76**	encrust.	**15.86**	encrust.	**19.12**
rock	**17.66**	rubble	**14.76**	rock	**10.64**
brown algae	**4.5**	brown algae	**8.1**	dead coral	**8.2**
leaf coral	**4.02**	feather	**7.44**	brown algae	**6.4**
Order 5 Ac%.	**68.06**	Order 5 Ac%.	**62.94**	Order 5	**73.36**
finger coral	3.98	leaf coral	4.7	feather	5.04
dead coral	3.4	sand	4.68	sand	4.34
bushy	2.98	finger coral	4.5	green algae	3.92
feather	2.88	green algae	3.96	brain coral	3.56
sand	2.62	dead coral	3.18	bushy	2.1
branched	2.42	bluegreen	3.12	leaf coral	1.3
branched	2.28	branched	2.32	encrust.	1.04
brain coral	2.2	bushy	2.26	branched	0.88
finger	1.88	brain coral	1.98	finger	0.78
bluegreen	1.8	branched	1.3	fan	0.78
green algae	1.22	fan octocoral	0.96	finger coral	0.78
encrust.	1.12	finger	0.82	encrust.	0.72
fan octocoral	1.06	candlestick	0.74	branched	0.46
candlestick	0.58	encrust.	0.56	bluegreen	0.26
leaf hidroc .	0.54	encrust.	0.5	candlestick	0.22
branch.	0.36	pillar coral	0.48	branch.	0.2
encrust.	0.14	branch.	0.44	leaf hidroc .	0.12
tube sponge	0.14	leaf hidroc .	0.32	broadfan	0.06
fleshy coral	0.08	fleshy coral	0.12	column	0.04
ball sponge	0.06	column	0.1	tube	0.02
broadfan oct.	0.04	flowercup	0.02	fleshy coral	0.02
vase sponge	0.04	broadfan oct.	0	vase	0
column	0.04	vase sponge	0	ball sponge	0
disc-plate	0.04	tube sponge	0	pillar coral	0
pillar coral	0.02	ball sponge	0	disc-plate	0
flowercup	0.02	disc-plate	0	flowercup	0

Table 3. Cover values of Main Groups, May 1997, July 1998 and November 1998.

Main Groups	May 1997	July 1998	November 1998
Sclaractinean Corals	31.4	28.96	25.24
Octocorals	11.98	15.1	10.58
Hidrocorals	0.9	0.76	0.32
Algae	7.52	15.18	10.58
Sponges	1.4	0.6	1.1
Rubble	23.12	14.76	29.0
Rock	17.66	16.78	10.64
Dead Coral	3.4	3.18	8.2
Sand	2.62	4.68	4.34

Shannon-Weiner Diversity Index values (Table 4) had a relative low variation between samples (0.224 bits/ind.) July 1998 sample showed the highest diversity and equity values (3.766 and 0.801, respectively). A comparison between diversity indexes values for the three samples gave the same result. For the Simpson Diversity Index it was found a slight difference between samples (2.952 bits/ind.) and July 1998 obtained the highest value of diversity (11.11 bits/ind.).

Once characterized, important differences were observed between the three samples for the Mahahual coral reef community. For their comparative analysis these differences were statistically verified and samples were classified.

Table 4. Diversity Indexes. 1 - D = Simpson Index (bits/ind.), E = Equity, H '= Shannon-Weiner Index (bits/ind.), S = MFG's Richness.

Mahahual	S	1-D	H'	E
May 1997	31	8.158	3.542	0.715
July 1998	26	11.11	3.766	0.801
November 1998	30	8.464	3.621	0.738

A one-way ANOVA using the logarithmically transformed cover data of the three samples showed a F value of 1.06, smaller than the F critic value of 3.08, which meant that there was at least one difference between the three samples of the coral reef community of Mahahual. The *a posteriori* Multiple Comparison Test (Newman-Keuls) showed that exist a significant difference between May

1997 and November 1998 samples. But contrasts between May 1997 - July 1998 and July 1998 - November 1998 samples showed no significant difference.

DISCUSSION

The coral reef community of Mahahual for the months of May 1997, July 1998 and November 1998, showed differences between those months. The significant difference was found for the months of May and November. Values of Diversity and Equity for the three months were high and represented a relatively good general state. July 1998 had the highest diversity and equity values, we thought because it presents homogeneous values, in contrast with the other two samples. This gave July 1998 a numerically better state of condition, but ecologically it does not have the same meaning.

From one year to another (1997 - 1998) a decrease in live coral cover and an increase in octocoral and algae cover were observed. In this period the dead coral cover values remained low. A dramatic change was noted on November 1998, after the bleaching event and the hurricane "Mitch", when scleractinean coral, octocorals, algae and rock covers decreased. At the same time rubble and dead coral covers increased. The increase in rubble cover was possibly due to the strong water movement in the shallow parts of the reef, during "Mitch" in October 1998. That could have cast down the coral rubble and small colonies from the barrier and shallow parts to the deeper parts of the reef, as the first step and the sand channel beneath. Rubble in November 1998 was constituted mainly by *Acropora cervicornis* broken colonies. Aproximately ten years ago, there was a sudden dieoff of *Acropora* genus species throughout the Caribbean (Aronson 1998, Greenstein 1998). *Acropora palmata* dead standing colonies are considered, by observations in field, to be much more resistant to physical stress than *A. cervicornis*, and it was considered that dead colonies of *A. cervicornis* were susceptible to transport by wave action. This movement of great volumes of rubble also affected the scleractinean coral cover by burying small colonies.

The dead coral cover increase was probably due to the death of colonies by stress associated to the bleaching event. It was estimated that for November 1998 scleractinean corals had been exposed to extreme conditions at least two months since September, when the "El Niño" high SST's started to affect the region. That is when the first reports of high water temperature and loose of pigmentation of coral colonies by local people started to arise.

The percentage of scleractinean cover affected by bleaching was 57.92%, that is 14.62% of total cover. The main genera and species affected in decreasing order were: *Montastraea spp., Agaricia tenuifolia and Diploria labrynthiformis*. An important decrease in cover of the MFG Leaf Coral was noted. It was considered to be the first group to be affected by bleaching, because in November '98 videotransect and field observations, there were just a few colonies with remaining living tissue, and a major part of the dead coral cover was given by standing dead colonies of *Agaricia* genus (leaf morph).

Aronson *et al.* (1998) stated in a study done in Belize, a shift in the dominant species, from the genus *Acropora* to the genus *Agaricia*, after dominant species were decimated by white-band disease. This change took place in a 10-15 years lapse, originating a difference in the reefscape and topographic complexity (among the most noticeable changes that were observed), but changes to associated communities are unknown.

In Mahahual's coral reef community we found something alike, a structural change caused by stochastic phenomena. Table 4 indicates a gradual decrease in cover of MFG Branching Coral (that accounts for the genus *Acropora*), from values of 2.28% to 1.3% and finally to 0.46% in November 1998. MFG Leaf Coral increased from 4.02% in May 1997 sample to 4.7% in July 1998. After being affected by bleaching in November 1998, its cover had decreased to 1.3%. At this point we expected leaf coral cover to keep decreasing, because the recorded value was mostly from small living tissue patches in heavily damaged colonies (mechanically and by bleaching).

The reef fishes community was affected, decreasing radically in number of species and recorded individuals by visual census for November 1998 (Pers. Obs. and Pers. Comm. Nuñez-Lara). This change was probably due to the decrease in water quality (increase in temperature and decrease in dissolved oxygen) in the area caused by "El Niño", and the wave and currents conditions present before, during and after "Mitch"

Another change, that went almost unnoticed, but considered important to the reef structure composition and its function, is the increase to more than twice its former cover value, of green algae of the genus *Halimeda*, from 1.22% in May 1997, to 3.96% July 1998, and brown algae from 4.5% in May 1997 to 8.1% in July 1998. This change can not be attributed to stochastic events, such as hurricanes or bleaching, because the green algae cover increased even before them.

Studies in this zone (Arias-Gonzalez 1998, Nuñez-Lara 1998) indicate that the coral reef fishes communities' present changes due to human activities, as fishing. In McClanahan (1995) the model of fishing impacts over reef structure and processes shows some cascade responses of the system to depletion of certain fish guilds. Relating those results to our study, the increase in hermatypic algae, could probably be considered as a structural response of the system to balance its carbonate fixation budget. As the scleractinean cover decrease and the fish populations are impacted by local fishing, herbivore activity (grazing and foraging) decrease, algae cover tend to increase, having more space available and less competition.

It can be concluded that artificial impacts on the coral reef community between May 1997 - July 1998 are not significant, and natural impacts between

July 1998 - November 1998 are also not significant. Nevertheless, as May 1997 and November 1998 have significant differences, the impact due to human activities and natural pressures must have a synergetic effect over the coral reef, to produce the significant change. Community changes due to natural impacts, as the bleaching event and the hurricane, could not be assessed effectively. Probably, because the survey was done too close to these events, (two weeks after the hurricane and the continuing bleaching) when the system still had not presented a structural change.

LITERATURE CITED

Arias-González, J.E. 1998. Geomorphological trophic models of semi-protected and unprotected reef ecosystems in the south of the mexican Caribbean. *Jour. of Fish. Biol.* **53**:236-255.

Aronson, R.B., and D.W. Swanson. 1997. Video surveys of coral reefs: uni and multivariate applications. Pages 1923-1926 in: *Proc. 8th Int. Coral Reef Sym*. Panama. 1997.

Aronson, R.B., W.F. Precht, and I.G. Macintyre. 1998. Extrinsic control of species replacement on a holocene reef in Belize: the role of coral disease. *Coral Reefs* **17**:223-230.

Cáceres-G.Cantón, C.E. 1998. Caracterización del Arrecife de Boca Paila, ubicado en la Reserva de la Biósfera de Sian Ka'an, Quintana Roo, México. Tésis de Licenciatura F.M.V.Z.-U.A.D.Y.México.

Carleton, J.H., and T.J. Done 1995. Quantitative video sampling of coral reef benthos: large-scale application. *Coral Reefs* **14**:35-46

Connell, J.H. 1997. Disaturbance and recorvery of coral assemblages. Pages 9-22 in: *Proc 8th Int. Coral Reef Sym.* Panamá 1997.

Greenstein, B.J., H.A. Curran, and J.M. Pandolfi. 1998. Shifting ecological baselines and the demise of *Acropora cervicornis* in the western Noth Atlantic and Caribbean Province: a Pleistocene perspective. *Coral Reefs* **17**:249-261.

Hawkins J.P., and C.M. Roberts. 1993. Effects of recreational scuba diving on coral reefs: trampling on reef-flat communities. *Jour. of App. Eco.* **30**(1):25-30

Hawkins J.P., and C.M. Roberts. 1997. Estimating the carrying capacity of coral reef for scuba diving. Pages 1923-1926 in: *Proc. 8th Int. Coral Reef Sym.* Panamá, 1997.

Humann, P. 1993. *Reef creature identification: Florida, Caribbean, Bahahmas.* New World Pub. Inc. Jacksonville, Florida. 2nd print. 320 pp.

Humann, P. 1992. *Reef coral identification: Florida, Caribbean, Bahamas.* New World Pub. Inc. Jacksonville, Florida. (1994) 2nd print. 284 pp.

Jennings, S., and N.V.C. Polunin. 1997. Impacts of predator depletion by fishing on the biomass and diversity of non-target reef fish communities. *Coral Reefs* **16**:71-82

Maldonado-Gazca, A. 1998. Estructura del paisaje arrecifal bentónico de Mahahual, Quintana Roo, México. Tesis de Maestría , CINVESTAV-I.P.N. México. 66 pp.

McClanahan, T.R. 1995. A coral reef ecosystem-fisheries model: impact of fishing intensity and catch selection on reef structure and processes. *Ecol. Model.* **80**:1-19.

Membrillo-Venegas, N. 1999. Descripción de los paisajes arrecifales en la laguna del Arrecife Alacranes, Yucatán, México. Tésis de Maestría, CINVESTAV, I.P.N. Unidad Mérida. Yucatán México 100 pp.

Nuñez-Lara, E., and E. Arias-González. 1998. Relationship between reef fish community structure and environmental variables of southern mexican Caribbean. *Jour. of Fish. Biol.* **53**:209-221.

Osborne, K., and W.G. Oxley. 1997. Sampling benthic communities using video transects. Pages 363-376 in: English S, Wilkinson C. and V. Baker (eds.) *Survey manual for tropical marine resources. 2nd Ed. Australian Institute of Marine Sciences*, Townsville Australia.

Rogers, C.S. 1993. Hurricanes and coral reefs; the intermediate disturbance hypothesis revisited. *Coral Reefs* **12**:127-137.

Talge, H. 1990. Impact of recreational divers on coral reefs in the Florida keys. *Proc.of the Am. Aca. of U.W. Sci. 10th An. Sci. Diving Sym.* 365-374 pp.

Seasonal Distribution of Gobiids in Waters Adjacent to Estuarine Marsh-edge Habitats: Assessing the Effects of Habitat Alteration

J. READ HENDON, MARK S. PETERSON, and BRUCE H. COMYNS
Gulf Coast Research Laboratory, Institute of Marine Sciences
The University of Southern Mississippi
P. O. Box 7000
Ocean Springs, MS 39566-7000 USA

ABSTRACT

In order to assess the effects of habitat alteration on local fish populations, collections were taken by beam plankton trawl (BPL) in waters adjacent to marsh-edge habitats of the Mississippi Gulf Coast. Monthly samples were taken over two years along natural *Juncus/Spartina* marsh-edge, natural beach-edge, and altered marsh-edge habitats in Back Bay, Biloxi, and Davis Bayou, Ocean Springs. Altered habitats consisted of bulkheads or riprap. Gobies were the most abundant fishes found in these habitats, with ***Gobiosoma bosc***, ***Gobiosoma robustum***, ***Gobionellus boleosoma***, ***Gobionellus shufeldti***, ***Microgobius gulosus***, ***Microgobius thalassinus***, and ***Evorthodus lyricus*** being collected during the study. ***Gobiosoma bosc*** constituted the bulk of both the gobies (98.2%) and the fishes in general (47.0%) collected, with postflexion larvae occurring from May to October. Seasonally, larval abundance peaked in fall for all species except for *E. lyricus* (May) and *G. bosc*, which peaked twice in year one (May, October) and once in year two (June). Relative abundance of all gobies was highest along natural marsh-edge habitats (75.1%), followed by natural beach (19.5%), and then altered marsh-edge (5.4%), and larval *G. bosc* were significantly more abundant in natural marsh habitats than in altered marsh (Hendon et al., 2000). The relatively high abundance of larval gobies in natural habitats suggests that these are important spawning and/or nursery areas for gobiids. Alteration of these landscapes may thus adversely affect larval abundance and distribution, as was evident in the Back Bay/Davis Bayou estuary.

KEY WORDS: Gobiidae, habitat alteration, *Juncus/Spartina* marsh edge

INTRODUCTION

The importance of marsh-edge habitat, the ecotone between intertidal marsh grass and adjacent open water within three meters of the water/shore edge (Baltz et al. 1993, Peterson and Turner 1994), to estuarine fish populations has been well-documented over the past few years, particularly in relation to early life-history stages (Rakocinski et al. 1992, Baltz et al. 1993, Peterson and Turner 1994, Peterson et al. 2000). Marsh-edge habitat is a critical nursery area for both residents, such as gobiids, and transients, such as sciaenids and penaeids

(Peterson and Turner 1994). This is true primarily because of the refuge the marsh grass provides and the high productivity of both the phytoplankton community (diatoms associated with marsh grass) and the microbial community, which makes detrital energy available to animal consumers through the food web. The marsh-edge is, however, highly vulnerable to anthropogenic influences, i.e. development, which may reduce its suitability as a nursery area for young fishes (Hoss and Thayer 1993).

Members of the family Gobiidae are common benthic fishes found worldwide within a wide array of estuarine and marine habitats (Dawson 1969). This taxa includes the most abundant fish species in Mississippi marsh-edge habitat (Peterson et al. 2000) and two of the three most abundant fishes in Louisiana marshes (Rakocinski et al. 1992, Baltz et al. 1993). Spatial distribution of gobies is controlled mainly by habitat attributes (Wilkins and Myers, 1992), such as salinity (Gill and Potter 1993), depth (Baltz et al. 1993), density of vegetation (Humphries and Potter 1993), and size of oyster shell openings (Crabtree and Middaugh 1982). Community dynamics of marsh-edge habitats may be greatly affected by gobiid fishes, as they are abundant predators on polychaetes and small crustaceans (Fitzhugh and Fleeger 1985) and are themselves an important forage fish for many species, such as seatrout, croakers, striped bass, and drums (Pearson 1929, Dawson 1966, Wass and Wright 1969).

Because gobies are a ubiquitous resident fish and can tolerate a wide range of physical conditions (Dawson 1969), their relative abundance among habitats may be a good indicator of the effects of anthropogenic influence on habitat quality. For example, larval *G. bosc* taken from this study were found to be significantly more abundant along natural marsh habitat than along altered marsh during the first year of sampling (Hendon et al. 2000). The physical and chemical characteristics of these habitat types did not differ statistically, so it is likely that the alteration of the habitat landscape played a significant role in determining larval distribution. Since a statistical treatment of the habitat association of *G. bosc* larvae from this study has already been reported (Hendon et al. 2000) and because little has been published on the larval goby fauna of this area, the objective of this study was to describe the spatial and temporal distribution of gobiids within waters adjacent to Mississippi marsh-edge habitats, focusing primarily on the larval stage.

MATERIALS AND METHODS

Thirteen sites from Back Bay (Biloxi, MS) to Davis Bayou (Ocean Springs, MS) were sampled for fishes between October 6, 1995, and September 10, 1997 (Fig. 1). Sites were classified as waters adjacent to natural *Juncus/Spartina* marsh-edge ("natural marsh"; 7 sites), waters adjacent to natural beach-edge ("natural beach"; 3 sites), or waters adjacent to altered *Juncus/Spartina* marsh-edge ("altered marsh"; 3 sites). Altered marsh included one stretch of riprap (Site 5) and two bulkheads (Sites 9 and 12). Natural and altered marsh sediments were primarily fine-grained sand and mud, with natural marsh sediments having four times the total organic carbon (% TOC) than altered marsh (Peterson et al. 2000). Natural beach sediments were mainly medium- to coarse-grained sand with a high % TOC.

Collections were taken during daylight with a Renfro (1963) beam plankton trawl (BPL) fitted with 794 μm mesh outer netting and a 505 μm sleeve and cod end. The net was pulled by hand for 46 m along the marsh edge unless low water levels caused by atmospheric forcing required sampling further from the edge. Samples were fixed in 10% formalin and later stored in 70% ethanol.

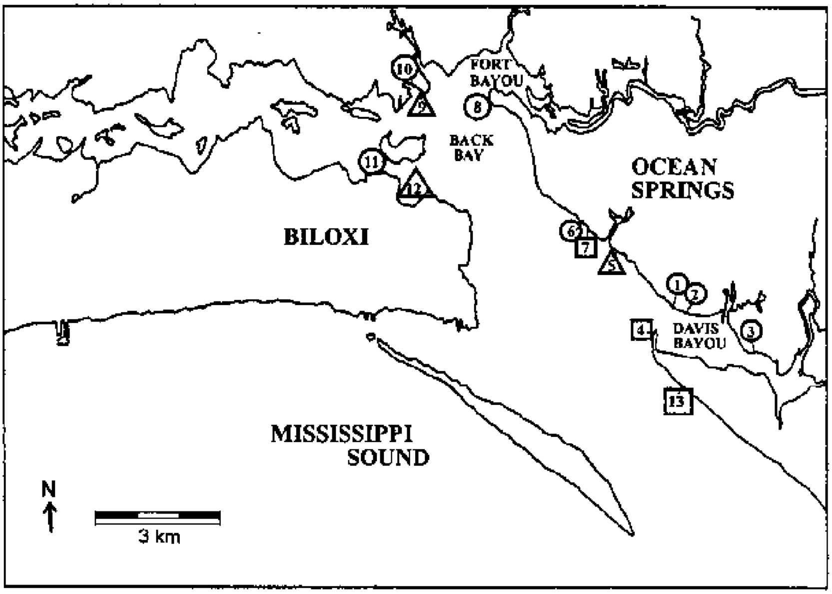


Figure 1. Sampling locations in Back Bay, Biloxi, and Davis Bayou, Ocean Springs. O= natural marsh; □= natural beach; Δ = altered marsh.

Physical parameters including water temperature (°C), salinity (°/$_{oo}$), dissolved oxygen (D.O., mg/L), turbidity (NTU), and water depth (cm) were measured at each site on each date. Wind speed (mph, converted to m/sec) was also measured at each site with a hand-held anonometer. Rainfall (cm) was obtained from the NOAA National Climate Data Center, Asheville, North Carolina, and tide height (cm) above mean low water was obtained in thirty minute intervals for each sampling date using MicroNautics, Inc., Tide.1 Program for the Back Bay area. Physical/chemical data were compared with non-parametric analyses among habitats and seasons and between years (Hendon 1998).

All fishes were sorted and archived, and gobiids were identified to the lowest possible taxonomic level, counted, and standard length (SL) was measured with dial calipers or an ocular micrometer. Small gobiids (<15 mm SL) were cleared and stained according to a revised protocol from Potthoff (1984). Identification to the species level was accomplished by counting dorsal and anal fin pterygiophores and comparing pigmentation patterns (Dawson, 1969). Specimens were then separated by life-history stage following the definitions in Leis and Rennis (1983). For unusually large collections, splits (3.9% of total samples) and subsamples (11.7%) were taken for enumeration and staging purposes, respectively. All splits and subsamples were adjusted accordingly before analysis. Because of low sample sizes for all species but *G. bosc*, abundance data given are general descriptive statistics ($\bar{x} \pm$ S.E. and ranges). A detailed statistical treatment of larval *G. bosc* distribution among habitat types is given in Hendon et al. (2000).

RESULTS

Habitat Characteristics

Habitats exhibited an overall significant difference in salinity [Kruskal-Wallis test (K-W); $\chi^2 = 10.20$; $n = 3$; $p = 0.006$], turbidity (K-W test; $\chi^2 = 13.57$; $n = 3$; $p = 0.001$), wind speed (K-W test; $\chi^2 = 7.77$; $n = 3$; $p = 0.021$), and depth (K-W test; $\chi^2 = 87.46$; $n = 3$; $p = 0.000$), while water temperature, dissolved oxygen, tide height, and rainfall were not significantly different ($p > 0.05$). Natural beach was the most variable of the habitats sampled. Salinity ranks were statistically higher at natural beach habitats than at natural marsh [Mann-Whitney *U*-test (M-W); $Z = -2.81$; $n = 2$; $p = 0.005$] and at altered marsh (M-W test; $Z = -2.84$; $n = 2$; $p = 0.005$), but no difference existed between ranks for natural marsh and altered marsh (M-W test; $Z = -0.73$; $n = 2$; $p = 0.466$). Mean turbidity ranks were also significantly higher in natural beach habitats than in natural marsh (M-W test; $Z = -3.11$; $n = 2$; $p = 0.002$) and altered marsh (M-W test; $Z = -3.38$; $n = 2$; $p = 0.001$), yet ranks for natural marsh and altered marsh did not differ (M-W test; $Z = -1.03$; $n = 2$; $p = 0.303$). Further, wind speeds were highest at natural beach habitats ($\bar{x} = 2.20$ m/s), followed by natural marsh ($\bar{x} = 1.65$) and then altered marsh ($\bar{x} = 1.48$), and natural beach waters were shallowest ($\bar{x} = 33.36$ cm), with natural marsh being intermediate ($\bar{x} = 45.22$) and altered marsh waters being deepest delineated by habitat type are given in Table 1.

All habitat parameters differed significantly among seasons ($p < 0.05$; Hendon 1998), and interannual differences existed between similar seasons for water temperature (higher in winter-1), salinity (higher in fall-1& summer-1), tide height (higher in winter-1 & spring-1), and rainfall (higher in summer-1 & spring-2).

Species Richness, Habitat Use, and Seasonality

During the two years of sampling, 26,348 gobies were collected with 24,213 (91.9%) developed enough to be identified to species. Only fish identified to species were included in these analyses. Seven gobiids were collected, with *G. bosc* constituting more than 98% of the goby abundance, and larvae comprising at least 50% of the abundance for each species (Table 2). At least 50% of the numbers of each species were collected in unaltered (natural) habitat.

Genus Gobiosoma — Two species in the genus *Gobiosoma* were collected over the two years: the naked goby, *G. bosc*, and the code goby, *G. robustum*. *Gobiosoma bosc* accounted for 23,780 (98.2%) of the specimens collected, with 23,667 (99.5%) of these being postflexion larvae. The naked goby occurred in 36.4% of the samples, with adults occurring throughout the year and juveniles present from August to December. Peaks in larval abundance occurred twice in year one, in May and in October, and only once in year two, in June, with larvae prevailing until October of both years (Figure 2). Mean abundance of *G. bosc* was highest in natural marsh habitat (165.35 ± 92.12; max = 9,808), followed natural beach ($\bar{x}$ = 127.45 ± 66.42; max = 2,511), and then altered marsh ($\bar{x}$ = 31.92 ± 14.45; max = 549), with larvae being significantly more abundant in natural marsh than altered marsh in year one (Hendon et al. 2000). Habitat measurements for *G. bosc* are given in Table 3.

The code goby, *G. robustum*, occurred in 5.1% of the samples taken, accounting for 246 of the gobies collected (Table 2). Only five adult *G. robustum* were collected, and juveniles (n = 37) were present only in October of year one. However, larvae were collected in relatively high numbers, occurring from May to October in year one, with peak abundance observed in August (Figure 2). Mean abundance was highest for this species in natural marsh (11.60 ± 3.38; max = 12), with altered marsh yielding the lowest mean abundance (1.60 ± 0.40). This species was collected only once along natural beach habitat (n = 6). The code goby occurred at the highest mean temperature of all species collected (Table 3).

Table 1. Habitat measurements for waters adjacent to natural marsh-edge, natural beach-edge, and altered marsh-edge. Data are given as mean ± standard error followed parenthetically by the range. Measurements for rainfall are not included in these statistics, as this was a regional phenomena that did not differ among habitats.

Variable	Natural Marsh-Edge	Natural Beach-Edge	Altered Marsh-Edge
Temperature (°C)	23.00±0.44 (09.7 - 32.3)	23.17±0.67 (10.7 - 32.0)	22.69±0.67 (11.1 - 32.3)
Salinity (°/$_{oo}$)	11.22±0.51 (0.5 - 24.0)	13.96±0.79 (1.5 - 24.5)	10.62±0.79 (0.5 - 24.9)
D.O. (mg/L)	8.04±0.17 (2.8 - 13.0)	8.67±0.21 (4.5 - 12.0)	8.15±0.24 (4.0 - 12.4)
Turbidity (NTU)	15.73±1.11 (2.5 - 101.5)	30.66±4.33 (3.4 - 230.0)	14.24±1.61 (2.1 - 86.5)
Depth (cm)	45.22±1.20 (10.0 - 90.0)	33.36±1.44 (10.0 - 75.0)	64.65±2.56 (1.0 - 105.0)
Wind Speed (m/sec)	1.65±0.11 (0.00 - 5.81)	2.20±0.21 (0.00 - 8.94)	1.48±0.16 (0.00 - 5.81)
Tide Height (cm)	32.12±1.52 (1.25 - 67.91)	32.58±2.44 (1.09 - 68.13)	32.21±2.44 (2.69 - 67.59)

Table 2. Descriptive statistics for the seven gobiid species collected within waters adjacent to the three marsh-edge habitat types over two years. Total number of gobiids identified to the species level = 24,213. NM = natural marsh habitat and NB = natural beach habitat. (No species was predominantly abundant in altered marsh habitat.)

Species	Total No.	% Number	% Larvae	Habitat Occurrence ($\bar{x}$ %)	Larvae Present
Gobiosoma bosc	23,780	98.21	99.5	NM (50.9%)	May - Oct
Gobiosoma robustum	246	1.02	82.9	NM (60.0%)	May Aug - Oct
Gobionellus boleosoma	79	0.33	63.3	NM (50.0%)	Mar Aug - Dec
Gobionellus shufeldti	4	< 0.01	50.0	NM (100%)	Sep, Oct
Microgobius gulosus	40	0.18	100.0	NM (50.0%)	Jun - Oct
Microgobius thalassinus	60	0.25	98.3	NB (60.0%)	May Sep - Oct
Evorthodus lyricus	4	< 0.01	100.0	NB (100%)	May

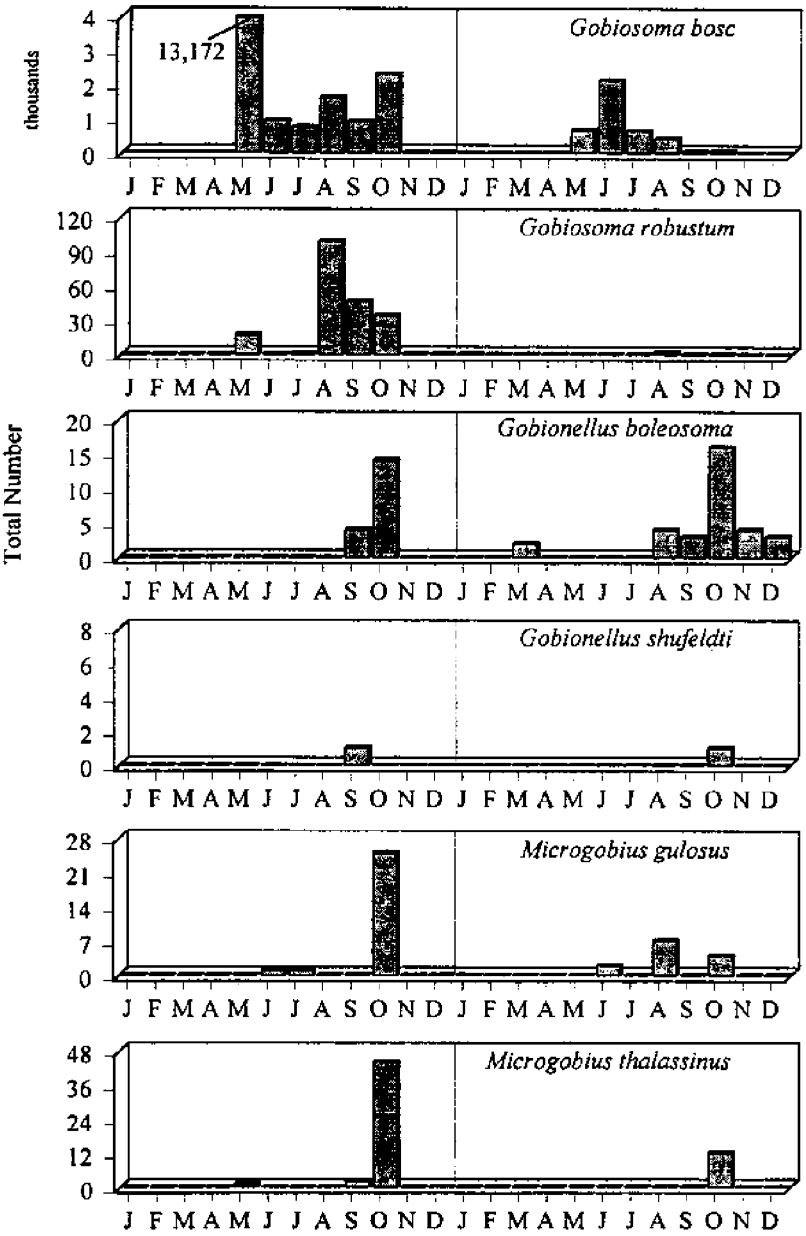


Figure 2. Total monthly abundance of postflexion larvae collected along Mississippi marsh-edge habitats over two years (habitats combined). Note that y-axis scale differs among graphs.

Table 3. Habitat measurements (mean ± S. E. followed parenthetically by the range) for each gobiid species collected (habitats combined). Habitat measurements for *E. lyricus* are given in the text.

Variable	***G. bosc***	***G. robustum***	***G. boleosoma***	***G. shufeldti***	***M. gulosus***	***M. thalassinus***
Temperature (^{o}C)	25.97 ± 0.34 (13.0-32.3)	27.14 ± 0.85 (16.2-32.3)	22.71 ± 0.85 (10.2-31.0)	24.28 ± 1.52 (21.5-28.6)	26.64 ± 0.69 (22.5-32.3)	25.60 ± 0.71 (20.4-29.7)
Salinity $(^{o}/_{oo})$	12.09 ± 0.47 (0.5-24.5)	13.33 ± 1.18 (0.8-21.0)	17.11 ± 0.97 (0.5-24.5)	18.23 ± 1.74 (15.1-22.2)	14.46 ± 1.70 (3.0-24.5)	16.19 ± 1.42 (3.6-24.4)
D.O. (mg/L)	7.40 ± 0.14 (2.8-11.5)	7.74 ± 0.42 (2.8-13.0)	7.64 ± 0.29 (2.9-11.7)	7.40 ± 0.31 (7.0-8.3)	7.70 ± 0.48 (4.3-10.6)	8.39 ± 0.40 (6.7-11.0)
Turbidity (NTU)	18.42 ± 1.24 (3.6-83.8)	17.52 ± 3.05 (4.3-64.6)	10.21 ± 0.91 (2.5-24.7)	9.68 ± 1.64 (6.8-13.0)	13.15 ± 1.98 (5.2-42.1)	17.47 ± 3.69 (5.2-47.8)
Depth (cm)	52.19 ± 1.43 (1.0-105.0)	56.38 ± 3.76 (20.0-90.0)	49.72 ± 2.89 (10.0-100.0)	45.00 ± 8.90 (30.0-65.0)	57.71 ± 5.24 (30.0-97.0)	60.62 ± 6.56 (30.0-97.0)
Wind Speed (m/s)	1.67 ± 0.10 (0.00-5.81)	1.86 ± 0.234 (0.00-4.47)	1.38 ± 0.23 (0.00-4.92)	2.01 ± 0.47 (0.89-3.13)	1.82 ± 0.35 (0.00-4.02)	2.13 ± 0.30 (0.45-3.58)
Tide Height (cm)	40.72 ± 1.33 (8.69-68.13)	40.96 ± 3.00 (13.83-65.63)	24.60 ± 2.57 (1.25-51.29)	22.90 ± 2.85 (15.44-28.54)	45.89 ± 2.94 (20.16-65.63)	38.07 ± 3.40 (15.07-59.71)
Rainfall (cm)	0.22 ± 0.04 (0.00-1.57)	0.50 ± 0.13 (0.00-1.57)	0.27 ± 0.07 (0.00-1.14)	0.25 ± 0.25 (0.00-0.99)	0.01 ± 0.01 (0.00-0.13)	0.16 ± 0.10 (0.00-0.99)

Genus Gobionellus — Two species in the genus *Gobionellus* were collected during this study: the darter goby, *G. boleosoma*, and the freshwater goby, *G. shufeldti*. The darter goby occurred in 7.5% of the samples taken and accounted for 79 of the gobies collected. Larvae dominated collections of this species, with abundance peaking in October of both years (Figure 2). Adults were sporadically collected throughout the year. Mean abundance of *G. boleosoma* was highest in natural marsh ($\bar{x}$ = 2.3 ± 0.54; max = 14), with collections in natural beach ($\bar{x}$ = 1.00 ± 0.001; max = 1) and altered marsh ($\bar{x}$ = 1.17 ± 0.17; max = 2) being uncommon. Habitat measurements for this species are given in Table 3.

Gobionellus shufeldti was less abundant than its congener, as one postflexion larva was collected in September of year one and one in October of year two (Figure 2). Two juveniles were collected in October of year two. This species was the only gobiid collected solely in natural marsh habitat. Contrary to its name, the freshwater goby occurred at the highest mean salinity of all gobiids (Table 3).

Genus Microgobius — The clown goby, ***M. gulosus***, was present in 3.4% of the samples, accounting for 40 of the gobies collected (Table 2). All ***M. gulosus*** collected were larvae taken from June to October, with peak abundance occurring in fall of both years (Figure 2). Mean abundance was highest in natural marsh habitat (3.13 ± 1.33; max = 12) followed by natural beach (¿=2.20±0.58; max=4) and altered marsh ($\bar{x}$ = 1.00 ± 0.00; max = 1). This species generally occurred in October when tide heights were high (Table 3).

The green goby, ***M. thalassinus***, occurred in 2.8% of the collections with 60 individuals collected (Table 2). Only one adult was collected during the study (December-1), and no juveniles were obtained. Larvae were present from May through October, with numbers peaking in October of both years (Figure 2). Abundance of the green goby was higher at natural beach ($\bar{x}$ = 8.50 ± 3.50; max = 12) than natural marsh ($\bar{x}$ = 4.33 ± 1.44; max = 11) or altered marsh ($\bar{x}$ = 2.00 ± 1.00; max = 3), and this species occurred mainly during the seasonally deep waters of October (Table 3).

Evorthodus lyricus — The lyre goby was collected only once during the study, in May of 1996 in natural beach habitat. The four larvae were collected at a depth of 75 cm in turbid (70.90 NTU), low-salinity (4.5 $^{o}/_{oo}$) waters 57.42 cm above mean low tide. Water temperature for this collection measured 27.9°C, and D.O. was 7.3 mg/L.

DISCUSSION

Gobies were an abundant resident taxa of the marsh-edge habitats of this estuary, particularly along natural marsh habitat. Although abundance was dominated by larval *G. bosc*, the collection of seven species in this family illustrates the diversity of these fishes in marsh-edge habitats. In a larger marsh-edge community study from which these data came (Peterson et al. 2000), gobiids accounted for 50.6% of all fishes collected. Several studies have examined the fish community within similar marsh habitats, but only a quarter of the studies reviewed showed gobiids constituting >13% of the total fish abundance, a feature probably due to sampling gear bias (Hendon 1998). Consequently, gobies are likely a more important component to marsh-edge community dynamics than several studies have exemplified, as the adults are both a resident predator and prey, and the larvae are an abundant seasonal source of food (Pearson 1929, Fitzhugh and Fleeger 1985). Three of the seven goby species were commonly collected, with the larvae of *G. bosc*, *G. robustum*, and *G. boleosoma* occurring in enough collections to suggest that natural marsh-edge serves as an important nursery habitat for the young of these species. Low and infrequent numbers of larval *G. shufeldti*, *M. gulosus*, *M. thalassinus*, and *E. lyricus* suggest that these species are relatively uncommon or may use other habitats during early developmental stages. The juvenile and adult stages of the gobiids collected are benthic and may likely be under-represented in this study, because the BPL would only effectively sample the pelagic environment. However, gobiid larvae are pelagic until about 15 mm SL (Shenker et al. 1983) and would consequently be fully vulnerable to our sampling gear.

The greatest numbers of larval *G. bosc* were collected in May and June of each year. By relating the mean SL of this stage of *G. bosc* ($\bar{x} = 6.63$ mm) with published growth rates of 5 to 6 mm per month for this species (Nero 1976, Conn and Bechler 1996), it may be ascertained that these fish are approximately one month old and that peak spawning occurred during April and May, respectively. A second, smaller peak in abundance of larval *G. bosc* occurred in October of year one, and consequently spawning of *G. bosc* lasted from April to September in this year. The same appears to be true for *G. robustum*. *Gobiosoma bosc* employed a bimodal spawning strategy in year one as in Conn and Bechler (1996), while only a single peak in larval abundance was evident in year two. These larvae were also much more abundant in year one than two. It is likely that the reproductive strategies of these two species were affected by the lower water temperatures of spring-2 (Dahlberg and Conyers 1973), and consequently year two abundances were lower. Assuming growth rates similar to *G. bosc*, the spawning season for each gobiid collected coincides with the available literature on goby reproduction (Dawson 1966, Fritzsche 1978, Allen and Barker 1990), except for *G. boleosoma* which spawned in the fall of this study but were reported to spawn in spring by Fritzsche (1978).

All gobiids collected were more abundant at unaltered habitats (namely natural marsh) than altered shorelines, and no larvae occurred in relatively high numbers at altered marsh habitats. It is therefore likely that natural *Juncus/Spartina* marsh-edge is a significant nursery habitat for gobiids in this estuary. The occurrence of *G. bosc*, *G. robustum*, *G. boleosoma*, *G. shufeldti*, and *M. gulosus* along natural marsh-edge habitats coincides with much of the current literature on these species (Peterson and Ross 1991, Rakocinski et al. 1992, Baltz et al. 1993, Peterson and Turner 1994). However, Peterson and Ross (1991) found juvenile and adult *G. boleosoma* to be the dominant gobiid along a Mississippi riverine-estuarine gradient, whereas larval *G. bosc* were numerically dominant in the present study.

In conclusion, gobies appear to be a much more important component to marsh-edge communities than previously thought, based on both their high abundance and their reported significance to the estuarine food web. The high abundance of these fishes along natural marsh relative to altered marsh suggests that unaltered *Juncus/Spartina* marsh is fundamental to the reproduction and/or recruitment of gobies, and thus these habitats may be viewed as source areas. Further, because gobies are eurytolerant and are abundantly distributed throughout the Gulf and Caribbean, yet appear to be adversely affected by the modification of natural marsh habitats, it is likely that alteration may be equally if not more detrimental to the reproductive success of less tolerant fishes, including several recreational and commercial species.

ACKNOWLEDGMENTS

Funding for this project was provided by the Mississippi Department of Marine Resources, Tidelands Trust Fund. We would like to thank C. F. Rakocinski for his assistance with revisions of the thesis from which these data were generated. Thanks also to G. Duff, G. Fulling, P. Bond, and C. Flowers, who assisted with sampling, fish identification, and/or statistical considerations.

LITERATURE CITED

Allen, D.M. and D.L. Barker. 1990. Interannual variation in larval fish recruitment to estuarine epibenthic habitats. *Marine Ecology Progress Series* **63**:113-125.

Baltz, D.M., C. Rakocinski, and J.W. Fleeger. 1993. Microhabitat use by marsh-edge fishes in a Louisiana estuary. *Environmental Biology of Fishes* **36**:109-126.

Conn, C. H. and D. L. Bechler. 1996. Reproductive strategies in a population of *Gobiosoma bosci* (Osteichthyes:Gobiidae) with slow and fast maturing individuals. *Gulf Research Reports* **9**:177-182.

Crabtree, R.E. and D.P. Middaugh. 1982. Oyster shell size and the selection of spawning sites by *Chasmodes bosquianus, Hypleurochilus geminatus, Hypsoblennius ionthas* (Pisces, Blennidae) and *Gobiosoma bosci* (Pisces, Gobiidae) in two South Carolina estuaries. *Estuaries* **5**:150-155.

Dahlberg, M.D. and J.C. Conyers. 1973. An ecological study of *Gobiosoma bosci* and *G. ginsburgi* (Pisces, Gobiidae) on the Georgia coast. *Fisheries Bulletin, U. S.* **71**:279-287.

Dawson, C.E. 1966. Studies on the gobies (Pisces: Gobiidae) of Mississippi Sound and adjacent waters. I. *Gobiosoma. The American Midland Naturalist* **76**:379-409.

Dawson, C.E. 1969. Studies of the gobies of Mississippi Sound and adjacent waters. II. An illustrated key to the gobioid fishes. *Publications of the Gulf Coast Research Laboratory Museum* 1:1-60.

Fitzhugh, G.R. and J.W. Fleeger. 1985. Goby (Pisces:Gobiidae) interactions with meiofauna and small macrofauna. *Bull. Mar. Sci.* **36**:436-444.

Fritzsche, R.A. 1978. Development of fishes of the mid-Atlantic Bight: an atlas of egg, larval, and juvenile stages. Vol. 5. U. S. Fish and Wildlife Service, FWS/OBS-78/12.

Gill, H.S. and I.C. Potter. 1993. Spatial segregation amongst goby species within an Australian estuary, with a comparison of the diets and salinity tolerance of the two most abundant species. *Marine Biology* **117**:515-526.

Hendon, J.R. 1998. Spatial and temporal distribution of gobiid fishes along natural and altered marsh-edge habitats. M.S. Thesis, The University of Southern Mississippi, Hattiesburg, Mississippi. 108 pp.

Hendon, J.R., M.S. Peterson, and B.H. Comyns. 2000. Spatio-temporal distribution of larval *Gobiosoma bosc* in waters adjacent to natural and altered marsh-edge habitats of Mississippi coastal waters. *Bull. Mar. Sci.* **66**(1): in press.

Hoss, D.E. and G.W. Thayer. 1993. The importance of habitat to the early life history of estuarine-dependent fish. *American Fisheries Society Symposium* **14**:147-158.

Humphries, P. and I.C. Potter. 1993. Relationship between the habitat and diet of three species of atherinids and three species of gobies in a temperate Australian estuary. *Marine Biology* **116**:193-204.

Leis, J.M. and D.S. Rennis. 1983. *The larvae of Indo-Pacific coral reef fishes.* New South Wales University Press, Sydney, Australia, and University of Hawaii Press. Honolulu, Hawaii.

Nero, L.L. 1976. The natural history of the naked goby *Gobiosoma bosci* (Perciformes: Gobiidae). M. S. Thesis, Old Dominion University, Norfolk, Virginia. 84 pp.

Pearson, J.C. 1929. Natural history and conservation of redfish and other commercial sciaenids on the Texas coast. *Bulletin of the United States Bureau of Fisheries* **44**:129-214.

Peterson, G.W. and R.E. Turner. 1994. The value of salt marsh edge vs interior as a habitat for fish and decapod crustaceans in a Louisiana tidal marsh. *Estuaries* **17**:235-262.

Peterson, M.S., B.H. Comyns, J.R. Hendon, P.J. Bond, and G.A. Duff. 2000. Habitat use by early life-history stages of fishes and crustaceans along a changing estuarine landscape: differences between natural and altered shoreline sites. *Wetlands Ecology and Management* (in press).

Peterson, M.S. and S.T. Ross. 1991. Dynamics of littoral fishes and decapods along a coastal river-estuarine gradient. *Estuarine, Coastal and Shelf Science* **33**:467-483.

Potthoff, T. 1984. Clearing and staining techniques. In: *Ontogeny and Systematics of Fishes.* The American Society of Ichthyologists and Herpetologists, Special Publication 1. Allen Press, Inc., Lawrence, KS.

Rakocinski, C.F., D.M. Baltz, and J.W. Fleeger. 1992. Correspondence between environmental gradients and the community structure of marsh-edge fishes in a Louisiana estuary. *Marine Ecology Progress Series* **80**:135-148.

Renfro, W.C. 1963. Small beam trawl for sampling postlarval shrimp. In:: Fishery Research, Biological Laboratory Galveston, Texas, Fiscal Year 1962. U.S. Bureau of Commercial Fisheries, Galveston.

Shenker, J.M., D.J. Hepner, P.E. Frere, L.E. Currence, and W.W. Wakefield. 1983. Upriver migration and abundance of naked goby (*Gobiosoma bosci*) larvae in the Patuxent River estuary, Maryland. *Estuaries* **6**:36-42.

Wass, M.L. and T.D. Wright. 1969. Coastal wetlands in Virginia, interim report. Special Report in Applied Marine Sciences and Oceanographic Engineering, Virginia Institute of Marine Science, Gloucester Point, Virginia. **10**:1-154.

Wilkins, H.K.A. and A.A. Myers. 1992. Microhabitat utilisation by an assemblage of temperate Gobiidae (Pisces: Teleostei). *Marine Ecology Progress Series* **90**:103-112.

Do Low-Salinity, Rock Jetty Habitats Serve as Nursery Areas for Presettlement Larval and Juvenile Reef Fish?

FRANK J. HERNANDEZ, JR.[1], RICHARD F. SHAW[1], JOSEPH S. COPE[2], JAMES G. DITTY[3] and TALAT FAROOQI[1]
[1]*Department of Oceanography and Coastal Sciences, Coastal Fisheries Institute,*
Louisiana State University
Baton Rouge, LA 70803 USA
[2]*Pfleger Institute of Environmental Research*
1400 N. Pacific St.,
Oceanside, CA 92054 USA
[3]*NOAA/NMFS, Fish Ecology Section*
4700 Ave. U
Galveston, TX 77551 USA

ABSTRACT

Previous investigations of artificial reef habitat in the northcentral Gulf of Mexico have identified the importance of offshore oil and gas platforms to economically-important reef fish. This study investigated the potential importance of another structurally-complex, hard-substrate habitat, a coastal rock jetty system, as a low-salinity, landward endmember of artificial reefs along a transect of three oil and gas platforms extending from the inner continental shelf to the shelf break. Quatrefoil light-traps and a bow-mounted plankton pushnet were used to collect fish along a pair of stone rubble jetties at Belle Pass near Fourchon, Louisiana during new moon and full moon phases from April to August, 1997. Light-traps and the pushnet collected 17,949 and 111,854 fish, respectively. Clupeiformes (engraulids and clupeids) comprised approximately 95% and 70% of the total light-trap and pushnet catch, respectively. Reef (or structure-dependent) fish, though not as abundant, included blennies, gobies, eleotrids, sparids, and lutjanids. The pushnet collected more individuals and taxa than did the light-trap. There were significantly higher pushnet densities and light-trap catch-per-unit-efforts (CPUEs) during new moon periods than full moon periods. This result may reflect a gear avoidance phenomenon where light-traps are more effective when not competing with the ambient light of a full moon and the pushnet is more effective under the darkness of new moon. An alternative hypothesis could also be related to lunar spawning and/or settlement periodicities. Significantly lower densities and CPUEs were observed at sampling stations located within the jetty walls versus stations located externally. This result may be related to possible differences in environmental parameters (turbidity, temperature, salinity, and dissolved oxygen) between inner (estuarine) and outer (coastal) sampling stations. Preliminary results indicate that the jetty may serve as a refuge area for presettlement reef fish in the absence of other structurally-complex habitat.

KEY WORDS: Rock jetty, nursery area, presettlement reef fish

INTRODUCTION

There are approximately 4,500 oil and gas platforms in the Gulf of Mexico, many of which serve as artificial reef habitat for economically-important reef fish (CDOP 1985, Gallaway 1981, Rooker et al. 1997). A debate persists, however, as to whether or not the fish associated with these structures and other artificial reefs are the result of increased fish production or simply aggregation (Bohnsack 1989, Pickering and Whitmarsh 1997, Seaman and Sprague 1992). Bohnsack et al. (1994) state that a major problem for managing reef resources is the incomplete understanding of the interactions between recruitment and habitat structure. In an effort to address questions concerning reef fish seasonality and across-shelf distribution, we have sampled extensively the early life history stages at several artificial reef habitats, including platforms located on the shelf slope (220 m depth), mid-shelf (61 m depth), inner shelf (22 m depth). This paper addresses preliminary results from the coastal endmember of this cross-shelf transect and another potentially important artificial habitat, a low-salinity, rock jetty system.

Many species of reef fish do not settle directly onto reefs but utilize other coastal habitats as nursery grounds prior to moving to offshore reefs. While habitats such as high-salinity seagrass beds are important to many reef related species (Connolly 1994), other structurally-complex habitats have been identified as nurseries (Ferrell and Bell 1991, Bennett 1989, Ross and Moser 1995). Seagrass beds are often the most common form of shelter available in certain settlement areas, but experimental evidence suggests that presettlement larvae of a number of different species select any complex structure at the time of settlement (Bell et al. 1987). Due to the overwhelming influence of the Mississippi River and its distributaries, Louisiana estuarine and coastal areas are generally low salinity, turbid, and lacking in seagrass beds and naturally-occurring hard substrate habitats (except for oyster reefs).

The objective of this study was to determine the composition and abundance of postlarval and juvenile fish inhabiting a coastal Louisiana jetty environment with particular emphasis on reef fish taxa. In addition, we were interested in comparing the light-trap and plankton pushnet methodologies in a low-salinity, turbid environment.

MATERIALS AND METHODS

The study was conducted along a pair of stone rubble jetties located at the terminus of Belle Pass, a major shipping channel near Fourchon, Louisiana, USA (29° 03.90' N, 90° 13.80' W). The jetties are approximately 91 m apart and run in a general north-south direction (Figure 1). The east jetty is approximately 335 m long and the west jetty is approximately 305 m long. Depths along the jetties ranged from 3 - 5 m. For sampling purposes, the sides of the jetties were labeled as East Exterior (EE), East Interior (EI), West Interior (WI), and West Exterior (WE). Four sampling stations, one on each side of each jetty, were located approximately at the jetty midpoints and were identified during sampling by distinct rock outcroppings.

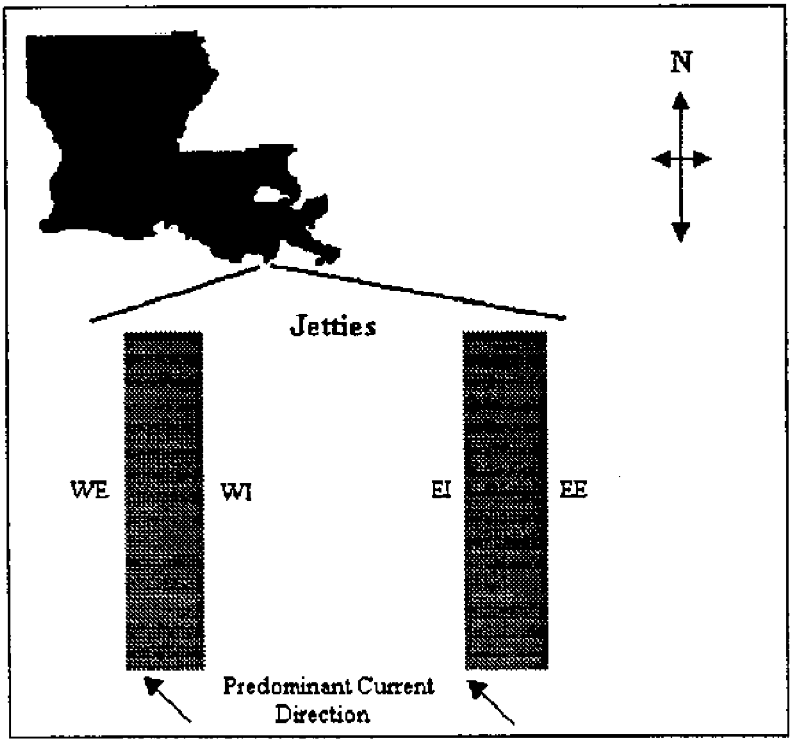


Figure 1. Schematic diagram of the study site with the sampling stations: west exterior (WE), west interior (WI), east interior (EI), and east exterior (EE).

A quatrefoil light-trap and a bow-mounted push plankton net were used to collect fish during two consecutive nights around new and full moon periods in 1997 from April through August, the estuarine recruitment period for a large number of our reef fish. New and full moon lunar phases were chosen because previous studies have indicated that reef fish exhibit lunar patterns in both spawning and recruitment (Robertson 1992, Kingsford and Finn 1997). Two sets of samples were taken each night. A set included a light-trap and a bow-mounted pushnet sample at each of the four stations. The order of stations sampled within each set was chosen using a random number table. Light-traps were equipped with a Brinkman Q Beam Starfire II halogen fishing light (12-v, 250 000 candlepower) and a submersible battery that was secured to the top of the light-trap with bungee cords. A 1000 μm mesh size was used for the bottom draining cod end. At each station, a buoyed mooring was used to suspend the light-trap approximately 1 m below the surface as close to the jetty as possible, which was usually between 0.5 - 2 m of the surface- exposed rock. The light-trap was allowed to fish for 10 minutes. Then, a bow-mounted pushnet with a 1000 μm mesh net mounted on a 1 m x 1 m frame was fished subsurfacely along the edge of the jetty for 3 - 5 minutes. A General Oceanics flowmeter (large rotor) was used to determine the volume of water filtered. All samples were fixed in 2 - 4% buffered formaldehyde. Subsurface and nearbottom salinity, temperature, dissolved oxygen, and turbidity were measured at each station using a DataSonde 3 Hydrolab and a DataSonde 3 Multiprobe Logger. Within 12 hours of collection all samples were rinsed and switched to an 80-90% ethanol solution. A projected 352 samples were to be collected [(5.5 months x 2 trips/month x 2 nights/trip x 2 sets/night x 4 jetty stations x (1 light-trap + 1 push trawl)] over the course of the study. However, not all stations were sampled equally due to adverse weather conditions and/or equipment failure. A total of 297 samples (148 light-trap samples and 149 push trawl samples) were collected and analyzed.

Samples were picked for all fish larvae, juveniles, and adults. Fish were measured to the nearest mm with an ocular micrometer (<10 mm) or a small ruler (>10 mm) and identified to the lowest possible taxonomic level. Preflexion larvae were measured to the end of the notochord (NL) and all postflexion larvae, juveniles, and adults were measured to the posterior end of the vertebral column (SL). Due to the very large numbers of clupeiform fishes (clupeids and engraulids), statistical analyses were performed on the data without all clupeiform fish, since these fish are seldom the taxa of interest in reef studies and their abundance tends to overwhelm the trends of other taxa (Choat et al. 1993). Light-trap samples were standardized to a catch-per-unit-effort (CPUE) of fish per minute. An ANOVA was run on the rank-transformed CPUEs for the effects of lunar phase and station location. Also customized comparisons were made using contrast statements in the SAS statistical package to test for differences between different combinations of the jetty stations, such as east stations vs. west and internal stations vs. external. Pushnet samples were standardized to the number of fish per 100 m^3 (density) and the same ANOVA design was run on the log-transformed mean densities for the pushnet data.

RESULTS

The light-trap and pushnet collected 17,949 fish and 111,854 fish, respectively. Catches by both gear types were dominated by clupeiform fishes (engraulids and clupeids) which comprised 95.26% of the light-trap total catch and 68.32% of the total pushnet catch. Only one non-clupeiform species, *Membras martinica* (Atherinidae), comprised over 1% of the total light-trap catch. Non-clupeiform taxa collected by the pushnet that comprised over 1% of the total catch include an unidentified gobiid (referred to as Goby Type I), *Cynoscion arenarius* (Sciaenidae), *Gobionellus oceanicus* (Gobiidae), *Citharichthys* sp. (Bothidae), *Symphurus* sp. (Soleidae), and *Microgobius* sp. (Gobiidae). Though not as common as other taxa, reef fish taxa were collected by both gear types (Table 1). The light-trap collected individuals from four reef fish families, while the pushnet collected fish from 12 reef fish families. Overall, the light-trap collected fish from 21 families with 38 taxa identifiable to at least the genus level. The pushnet collected fish from 41 families with 80 taxa identifiable to at least genus. The most dominant taxa were presettlement blenniids and gobiids. All taxa sampled with the light-trap were also collected by the pushnet but in greater numbers.

Mean light-trap CPUEs for each trip ranged from 0 – 1.48 fish per minute (Figure 2). The ANOVA on the ranks of mean CPUE values (minus clupeiforms) indicate significantly higher CPUEs occurred during new moon phases than full. There was also a significant difference between the different sampling stations, with significantly higher CPUEs occurring at the external (WE and EE) stations than the internal (WI and EI) stations (Figure 3).

Mean pushnet densities for each trip ranged from 18 – 288 fish/100 m^3 (Figure 4). ANOVA results on the log-transformed, pushnet mean densities also indicated that significantly more fish were collected during new moon phases than full. Significantly higher densities also occurred at the external stations vs. the internal stations (Figure 5).

Table 1. List of reef fish taxa and the total number of each collected with light-traps (LT) and pushnet (PN) during the course of the study.

Taxon	Total Number	
	LT	PN
Blenniidae		
Chasmodes sp.		1
Hypleurochilus bermudensis		1
Hypsoblennius hentz/ionthas	154	551
Scartella sp.	4	87
Unidentified	3	7
Eleotridae		
Bathygobius soporator		3
Dormitator maculatus	18	252
Unidentified	3	53
Ephippidae		
Chaetodipterus faber		13
Gobiidae		
Gobionellus oceanicus	16	2201
Gobiosoma bosc		11
Gobiosoma sp.	40	402
Microgobius sp.	7	1175
Type I	88	11890
Unidentified	4	231
Haemulidae		
Unidentified		1
Labridae		
Unidentified		3
Lutjanidae		
Lutjanus griseus	2	18
Lutjanus synagris		8
Unidentified	1	
Mullidae		
Unidentified		1
Rachycentridae		
Rachycentron canadum		2
Serranidae		
Mycteroperca sp.		1
Scaridae		
Sparisoma sp.		4
Sparidae		
Unidentified		9

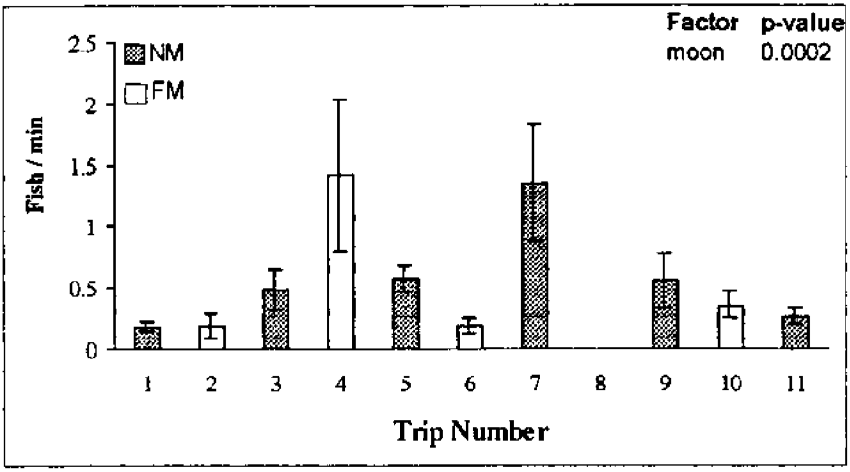


Figure 2. Mean light-trap CPUEs (minus clupeiforms) for each sampling trip. Open columns indicate trips during full moon phases and dark columns indicate trips during new moon phases. Error bars are standard errors of the mean.

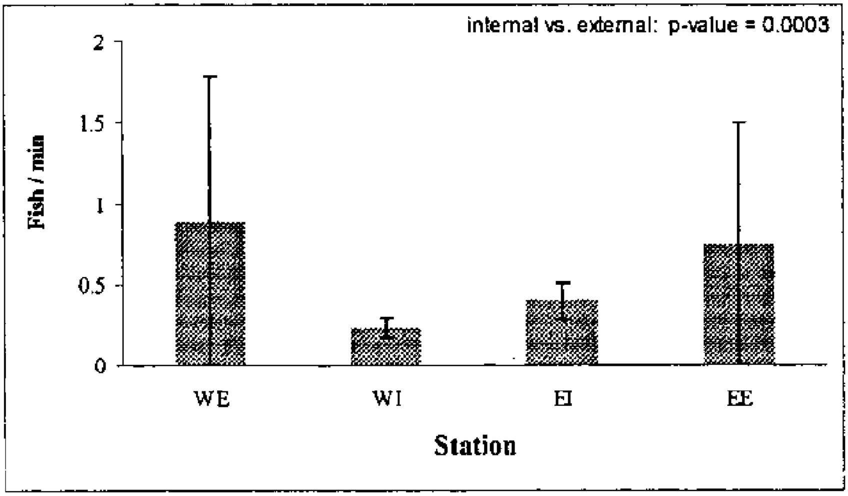


Figure 3. Mean light-trap CPUEs (minus clupeiforms) for each sampling station. Error bars are standard errors of the mean.

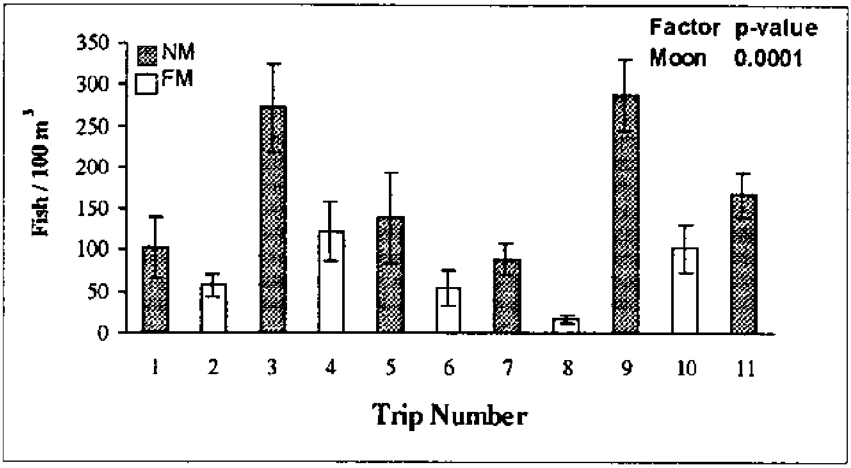


Figure 4. Mean pushnet densities (minus clupeiforms) for each sampling trip. Open columns indicate trips during full moon phases and dark columns indicate trips during new moon phases. Error bars are standard errors of the mean.

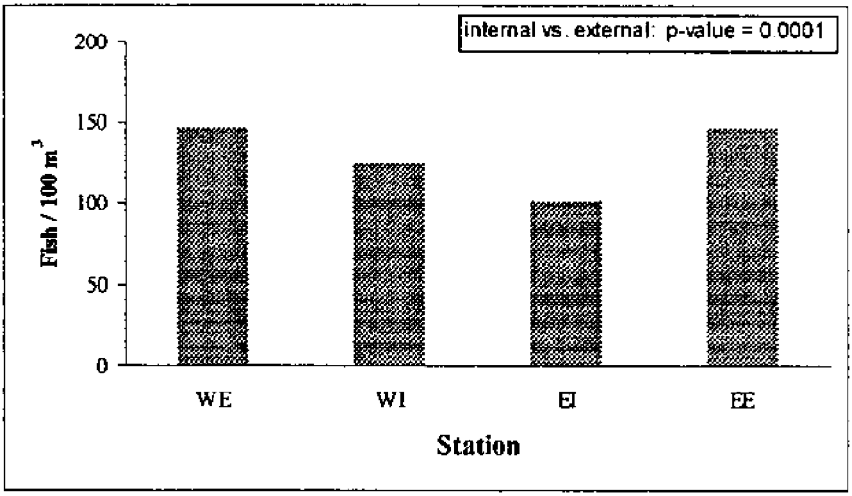


Figure 5. Mean pushnet densities (minus clupeiforms) for each sampling station. Error bars are standard errors of the mean.

DISCUSSION

An advantage to using light-traps and pushnets is that both gears have demonstrated the ability to sample larger larvae and juvenile fishes (Herke 1969, Kriete and Loesch 1980 Choat et al. 1993, Hernandez and Lindquist 1999). In this study, both gears sampled primarily postsettlement and juvenile individuals. Future analyses will look at the length-frequency distributions to statistically compare the sizes of the fish collected by each gear type. Also, both gears were easily deployed in this relatively structurally-complex environment. A potential disadvantage of the light-trap is that it tends to be taxon-specific, attracting only photopositive taxa (Doherty 1987, Choat et al. 1993). This can result in samples dominated by relatively few taxa (Doherty 1987, Brogan 1994 Choat et al. 1993). Our results are typical in this respect as clupeiform fishes comprised over 95% of total light-trap catch. The dominance of clupeiform fishes in the pushnet samples (near 70%) attests to the abundance of these species in the nearshore environment. The pushnet catches were larger and more diverse than the light-trap catches. Again, this is probably related to the taxon selectivity of the light-trap, and the volume of water sampled, although to date there is no accurate way to calculate the volume of water sampled by a light-trap.

The most common reef or structure-dependent fishes were gobiids, blenniids, and eleotrids (Table 1). These species, while present on offshore artificial reef sites such as oil and gas platforms, are also common in the nearshore and estuarine environment which probably explains their high abundance. While these taxa are small cryptic inhabitats in artificial reef environments and are not economically important, they may serve as model species in looking at cross-shelf distribution patterns of reef fish recruitment. For example, presettlement juveniles of *Hypsoblennius hentz/ionthas* were common not only at the jetty site but at our innershelf platform site, which was sampled during the same year. Once the identification of this species can be confirmed, it will be used in otolith analyses to determine pelagic larval durations for individuals collected at the jetty versus the innershelf platform. This, combined with information on local current patterns, can give us an idea of the effective transport radius for this species. Also, otolith-based growth estimates for these fish will be compared between the two locations to compare pelagic larval pelagic durations, settlement times, and any other advantages (with respect to accumulative mortality) that early settlement times at one habitat versus another may confer.

While other species of reef fish were less common, their presence indicates the jetty environment can serve as a nursery area. During their pelagic phase, mortality rates of reef fish larvae approach 100% (Leis 1991), and juveniles always appear to be rare. Louisiana coastal waters are dominated by mud and silt bottomes with little bathymetric relief and are devoid of typical reef fish nursery habitats such as natural reefs and seagrass beds. Therefore, the role of the artificial habitats such as jetties and breakwaters may become more important as islands of refuge for individuals that would otherwise be lost to unsuitable habitat and, therefore, mortality.

The larger mean light-trap CPUEs and pushnet densities during new moon phases may be a result of gear efficiency. Light-traps rely on illumination in the surrounding water mass to attract fish. Theoretically, the contrast in trap-generated illumination should be greater when there is less ambient light such as during a new moon phase as opposed to a full moon. Few studies utilizing light-aggregating devices have addressed this efficency issue within the framework of lunar periodicities in fish spawning, larval supply (transport) and settlement advantages. Gregory and Powles (1985) observed higher catches during new moon phases in a freshwater system but didn't report a statistical difference. Rooker et al. (1996) used a nightlight lift-net in nearshore habitats in Puerto Rico and looked at sampling abundances for all four lunar phases. They found that new moon abundances of larval fish were four times higher than the next most abundant phase (last quarter) during the summer months and suggested the ambient light intensities might have played a factor in gear efficiency.

Another possible reason for larger catches during new moon periods may be related to the lunar periodicities of some taxa with respect to spawning, recruitment, and settlement. Robertson (1991) noted that many larval biology hypotheses concerning lunar reproductive patterns pertain to propagule dispersal and predation rates and occur both at the beginning and end of the planktonic phase. Many reef fish, for example, time their spawning events with different lunar cycles (Thresher 1984). Previous studies have also documented higher rates of fish settlement during darker, new moon periods than full moon periods (Victor 1986, Rooker et al. 1996), presumably a response to decrease mortality from visual predators. These patterns of spawning and recruitment, in association with the local physical regime, result in variable larval supply and settlement patterns often with distinct lunar signals.

A similar gear efficiency effect may be occurring with the pushnet. Net avoidance is a well documented phenomenon and several studies have demonstrated increased net avoidance during the day as opposed to night (Brander and Thompson 1989, Suthers and Frank 1989, Leis 1991). While all of our pushnets were taken at night, there may have been higher incidences of net avoidance during brighter, full moon periods than darker, new moon periods.

Another factor that may affect the efficiency of both gear types is the turbidity of the water masses sampled. Higher turbidity should decrease the effectiveness of the light-trap and increase the effectiveness of the pushnet. Turbidity data was collected but has not been analyzed yet. Differences in turbidity may explain the observed differences between the internal stations versus the external stations (Figures 3 and 5). Though the tidal currents are not particularly strong in this location, there may be significantly higher mixing in the channel between the jetty walls resulting in higher turbidities than outside the jetty walls. Turbidity and other environmental parameters (salinity, temperature, dissolved oxygen) will be analyzed in the future to explain the differences in catches between the internal and external stations.

In addition to the environmental factors, the hydrology around the mouth of the inlet may aid in concentrating fish at the outer stations. Hydrodynamic models describing tidal inlet flow patterns often predict the formation of eddies upstream and downstream of inlet mouths (Carter 1988). There is a west-northwest net residual coastal flow along the Louisiana coast that is favorable for this type of eddy formation/setup. While many of these models predict the movement of passive particles, the mechanism may still be a valid explanation for the concentration of postlarval and juvenile fish at the outer stations of our sampling site.

LITERATURE CITED

Bell, J.D., M. Westoby, and A.S. Steffe. 1987. Fish larvae settling in seagrass: do they discriminate between beds of different leaf density? *J. Exp. Mar. Biol. Ecol.* **111**:133-144.

Bennett, B.A. 1989. The fish community of a moderately exposed beach on the Southwestern Cape coast of South Africa and an assessment of this habitat as a nursery for juvenile fish. *Estu., Coast., & Shelf Sci.* **28**:293-305.

Bohnsack, J.A. 1989. Are high densities of fishes at artificial reefs the result of habitat limitations or behavioral preference? *Bull. Mar. Sci.* **44**:631-645.

Bohnsack, J.A., D.E. Harper, D.B. McClellan, and M. Hulsbeck. 1994. Effects of reef size on colonization and assemblage structure of fishes at artificial reefs off southeastern Florida, U.S.A. *Bull. Mar. Sci.* **55**:796-823.

Brander, K. and A.B. Thompson. 1989. Diel differences in avoidance of three vertical profile sampling gears by herring larvae. *J. Plankton Res.* **11**:775–784.

Brogan, M.W. 1994. Two methods of sampling fish larvae over reefs: a comparison from the Gulf of California. *Mar. Biol.* **118**:33–44.

Carter, R.W.G. 1988. *Coastal Environments: An Introduction to the Physical, Ecological, and Cultural Systems of Coastlines.* Academic Press, London.

CDOP (Committee on Disposition of Offshore Platforms). 1985. *Disposal of offshore platforms.* Marine Board, Commission on Engineering and

Technical Systems, National Research Council, National Academy Press, Washington, D.C. 76 pp.

Choat, J.H., P.J. Doherty, B.A. Kerrigan, and J.M. Leis. 1993. A comparison of towed nets, purse seine, and light-aggregation devices for sampling larvae and pelagic juveniles of coral reef fishes. *Fish. Bull., U.S.* **91**(2):195 -209.

Connolly, R.M. 1994. A comparison of fish assemblages from seagrass and unvegetated areas of a southern Australian estuary. *Australian J. Mar. Freshwater Res.* **45**:1033-1044.

Doherty, P.J. 1987. Light-traps: selective but useful devices for quantifying the distributions and abundances of larval fishes. *Bull. Mar. Sci.* **41**(2):423–431.

Ferrell, D.J. and J.D. Bell. 1991. Differences among assemblages of fish associated with *Zostera capricorni* and bare sand over a large spatial scale. *Mar. Ecol. Prog. Ser.* **72**:15-24.

Gallaway, B.J. 1981. *An ecosystem analysis of oil and gas development on the Texas-Louisiana continental shelf.* U.S. Fish & Wildl. Serv., Office of Biological Services, Washington, D.C., FWS/OBS-81/27. 89 pp.

Gregory, R.S. and P.M. Powles. 1985. Chronology, distribution, and sizes of larval fish sampled by light traps in macrophytic Chemung lake. *Can. J. Zool.* **63**:2569–2577.

Herke, W.H. 1969. A boat-mounted surface push-trawl for sampling juveniles in tidal marshes. *Prog. Fish-Cult.* **31**:177-179.

Hernandez, F.H., Jr. and D.G. Lindquist. 1999. A comparison of two light-trap designs for sampling larval and presettlement juvenile fish above a reef in Onslow Bay, North Carolina. *Bull. Mar. Sci.* **64**(1):173–184.

Kingsford, M. and M. Finn. 1997. The influence of phase of the moon and physical processes on the input of presettlement fishes to coral reefs. *J. Fish. Biol.* **51**(Supp. A):176-205.

Kriete, W.H., Jr., and J.G. Loesch. 1980. Design and relative efficiency of a bow-mounted pushnet for sampling juvenile pelagic fishes. *Trans. Am. Fish. Soc.* **109**:649-652.

Leis, J.M. 1991. The pelagic stage of reef fishes: the larval biology of coral reef fishes. Pages 183–230 in: P.F. Sale (ed.) *The Ecology of Fishes on Coral Reefs.* Academic Press, San Diego.

Pickering, H. and D. Whitmarsh. 1997. Artificial reefs and fisheries exploitation: a review of the 'attraction versus production' debate, the influence of design and its significance for policy. *Fish. Res.* **31**:39-59.

Robertson, D.R. 1991. The Role of Adult Biology in the Timing of Spawning of Tropical Reef Fishes. Pages 356–386 in: P.F. Sale (ed.) *The Ecology of Fishes on Coral Reefs.* Academic Press, San Diego.

Robertson, D.R. 1992. Patterns of lunar settlement and early recruitment in Caribbean reef fishes at Panama. *Mar. Biol.* **114**:527-537.

Rooker, J.R., G.D. Dennis, and D. Goulet. 1996. Sampling larval fishes with a nightlight lift-net in tropical inshore waters. *Fish. Res.* **26**:1–15.

Rooker, J.R., Q.R. Dokken, C.V. Pattengill, and G.J. Holt. 1997. Fish assemblages on artificial and natural reefs in The Flower Garden Banks National Marine Sanctuary, USA. *Coral Reefs.* **16**:83-92.

Ross, S.W. and M.L. Moser. 1995. Life history of juvenile gag, *Mycteroperca microlepis*, in North Carolina estuaries. *Bull. Mar. Sci.* **56**:222-237.

Seaman, W., Jr. and L.M. Sprague (Eds.). 1992. *Artificial Habitats for Marine and Freshwater Fisheries*. Academic Press Inc., New York. 285 pp.

Suthers, I.M. and K.T. Frank. 1989. Inter-annual distributions of larval and pelagic juvenile cod (Gadus morhua) in southwestern Nova Scotia determined with two different gear types. *Can. J. Fish. Aquat. Sci.* **41**:1428–1435.

Thresher, R.E. 1984. *Reproduction in Reef Fishes*. T.F.H. Publications, Inc. 399 pp.

Victor, B.C. 1986. Larval settlement and juvenile mortality in a recruitment-limited coral reef fish population. *Ecol. Monogr.* **56**:145–160.

Effects of Refuge Size and Complexity on Recruitment and Fish Assemblage Formation on Small Artificial Reefs

ROBIN L. SHERMAN, D.S. GILLIAM and R.E. SPIELER
Nova Southeastern University Oceanographic Center
8000 North Ocean Drive
Dania Beach, FL, 33004 USA

ABSTRACT

There have been a number of natural and artificial reef studies examining possible correlation's between refuge size and complexity and the associated fish assemblages. Results of these studies have been contradictory indicating possible site dependent differences. We examined the role of refuge size and complexity in fish recruitment and the formation of associated fish assemblages, using artificial reef modules, at two different depths off Fort Lauderdale, Florida, USA. The 1 m^3 reef modules (Swiss Cheese reefs) were constructed of poured concrete with 12 tunnels running through the block, six tunnels in each direction perpendicular to each other. Twenty replicates were constructed of each of three different refuge configurations, 12 large tunnels (square opening, 15 cm per side), 12 small tunnels (7.5 cm per side) or six large and six small tunnels. Ten replicates of each reef design were deployed at each depth (7 m and 20 m) on sandy substrate. Significant differences were found for recruits (fishes 0 - 5 cm TL), total fish abundance, and species richness between refuge configurations as well as depths. Additionally, this study was designed, in part, to replicate work done previously by others, in the Caribbean examining the effects of refuge size on fish abundance and species richness. Differences were found between the studies. This may be due, in part, to differences in local fish species composition and population structure.

KEY WORDS: Refuge, site dependent differences, substrate

INTRODUCTION

Many aspects of artificial structure have been examined in efforts to identify ways to increase fish abundance and diversity, improve catch rates of targeted species, manipulate habitats, and restore damaged coral reefs (for references see: Bohnsack and Sutherland 1985, Bohnsack 1990, Bohnsack et al. 1991, Seaman 1997, Spieler et al. in press). A number of natural and artificial reef studies examined possible correlation's between refuge size and complexity and the associated fish assemblages (Molles 1978, Gascon and Miller 1982, Roberts and Ormond 1987, Brock and Norris 1989, Hixon and Beets 1989, Eklund 1996, Friedlander and Parrish 1998, Spieler 1998, Spieler in press, Authors unpublished). Results of these studies have been contradictory indicating possible site dependent differences. We examined the role of refuge size and complexity in fish recruitment and the formation of associated fish assemblages, using artificial reef modules, at two different depths off Fort Lauderdale, Florida.

MATERIALS AND METHODS

Experimental Design

This research focused on three central hypotheses:

i) the recruitment and aggregation of a diverse assemblage of fishes to artificial reefs can be effected by the size of the refuges available,

ii) diverse refuge sizes are superior to uniform refuge sizes for the recruitment and aggregation of a diverse assemblage of fishes to an artificial reef, and

iii) deployment site selection (i.e. depth) will effect the formation and maintenance of the fish assemblage on a small artificial reef.

Briefly, the experimental design consisted of comparing fish abundance, species richness, and species composition among three groups of reefs (10 Swiss Cheese reefs each) with either large holes only (Large refuge reefs), small holes only (Small refuge reefs), or large and small holes (Mixed refuge reefs), at two different depths. The hypotheses above allowed us to make three specific predictions that were statistically verifiable:

i) Artificial reefs with large refuges will have different fish abundance, species richness, and species composition than reefs with small refuges

ii) Reefs with mixed sized refuges will have greater fish abundance, and species richness than either large refuge reefs or small refuge reefs, and

iii) Reefs deployed at different depths will have different fish abundance, species richness, and species composition.

Construction and Deployment

The reefs (Swiss Cheese reefs) were constructed at the CSR Rinker Concrete Plant, Pembroke Pines, Florida using waste concrete (concrete remaining in trucks when returning from a job) poured into reusable molds made from pressure treated plywood. Tunnels were formed in the concrete using wooden dowels of either 3.5 cm square or 10 cm square wrapped in 2 cm thick Styrofoam. When assembled, this form contained approximately 1 m^3 of poured concrete with 12 tunnels (refuges) running through the block, six tunnels in each direction perpendicular to each other (Figure 1). Twenty replicates were constructed of each of three different refuge configurations, 12 large refuges (square opening, 15cm per side), 12 small refuges (7.5 cm per side) or six large and six small refuges. Ten replicates of each reef design were deployed at each of two depths (7 m and 20 m) on sandy substrate. The deep site was approximately one kilometer east of the shallow site. Thirty reefs were deployed at each site, in sandy substrate, with 35 meters between each reef.

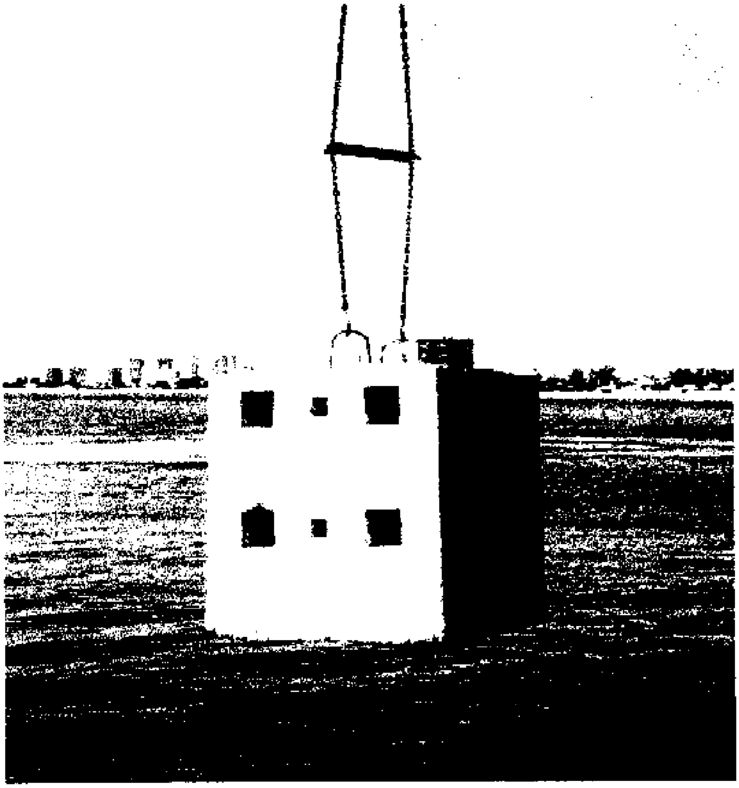


Figure 1. Mixed refuge size reef being deployed offshore Broward County, Florida, USA

Monitoring

In an effort to examine seasonal differences the reefs were monitored approximately quarterly (eight times) from October 1996 through January 1999. Divers, using SCUBA, counted, and recorded census data on slates marked on one edge with five size intervals: < 5, 5 - 10, 10 - 20, and 20+ cm to aid in length estimation. The reefs are small enough to allow for an accurate total count without sub-sampling. Species, numbers of fish per species, and estimated total length, by size class, of all fishes within 1 m of each reef (18 m^3, total volume including reef) were recorded. The size classes were also used to calculate fish biomass on the reefs. The mean total length (TL) for each size class was used in length-weight equations published by Bohnsack and Harper (1988). When a length-weight equation for an identified species was not available, the equation for a congeneric was used.

Data were analyzed with non-parametric analysis of variance techniques using Statistical Analysis Systems (SAS) software (SAS Institute Inc., Cary, NC, USA) (PROC GLM of ranked data Kruskal-Wallis k-sample test, and a Student-Newman-Keuls test between means).

RESULTS

Refuge Size

With the following exceptions, little difference was noted in the number of fishes (all species combined) among the three refuge sizes for any size class. At both the shallow and deep sites there were more 20+ cm fish and more total fish (all sizes combined) associated with large or mixed refuge reefs than small refuge reefs ($p < 0.05$, ANOVA/SNK) (Figure 2, 3). At the shallow site there were also more juvenile fishes (< 5 cm TL) on the large refuge reefs than those with small refuges ($p < 0.05$, ANOVA/SNK).

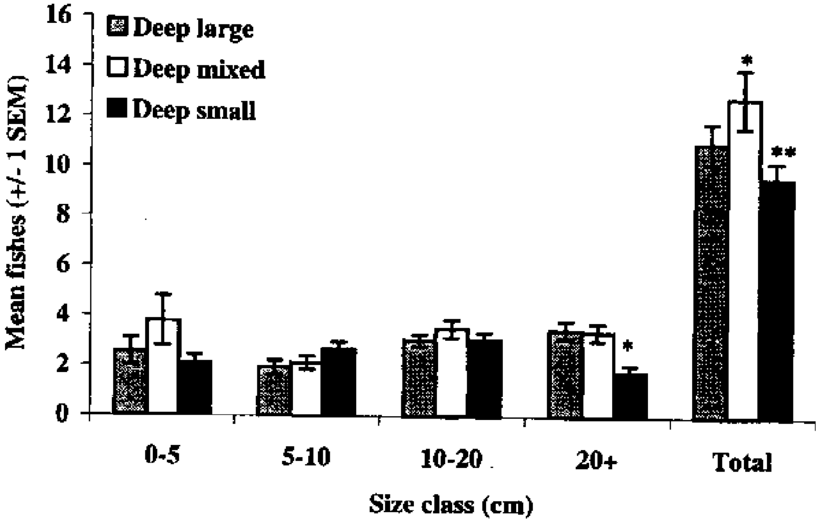


Figure 2. Mean number of total fishes (+/- 1 SEM) by size class on deep reefs with one of three treatments: large, small or mixed refuge sizes. Asterisked treatments are significantly different from other treatments within a size class ($p < 0.05$, SNK).

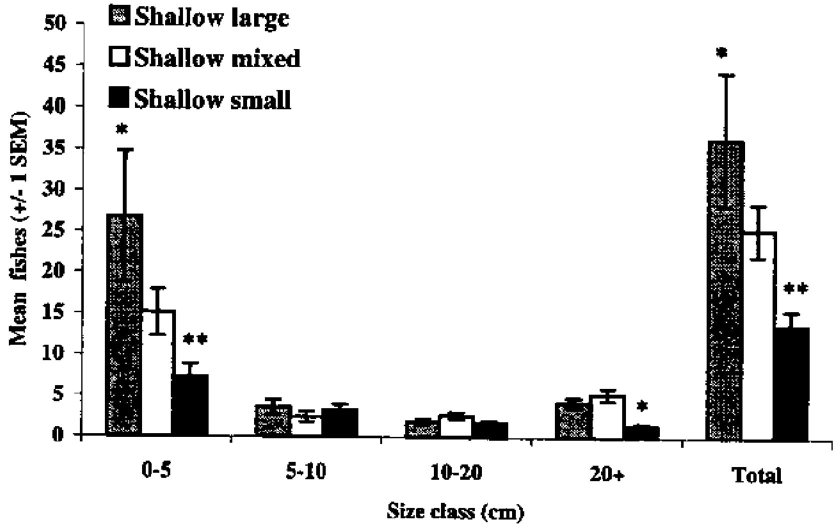


Figure 3. Mean number of total fishes (+/- 1 SEM) by size class on shallow reefs with one of three treatments: large, small or mixed refuge sizes. Asterisked treatments are significantly different from other treatments within a size class ($p < 0.05$, SNK).

Likewise, the number of species, by size class, did not differ much among the three treatments. Both shallow and deep reefs had more species in the 20+ cm size class on the large and mixed refuge reefs and at the shallow site, there were also more total species on the large and mixed refuge reefs ($p < 0.05$, ANOVA/SNK) (Figure 4, 5). There were more <5cm TL species on large and mixed refuge shallow reefs vs. deep. Large size class fishes (>10 cm), as well as total species (all sizes combined), were more abundant deep on large and mixed refuge reefs ($p < 0.05$, ANOVA/SNK). Throughout the year, total species (all size classes combined) were not significantly different on the deep reefs among months ($p > 0.05$) but were highest on the shallow reefs in July and November ($p < 0.05$, ANOVA/SNK).

Site Dependent Differences

Seventy-seven species were recorded at the deep site and 49 species at the shallow site. Of the 88 total species recorded, 40 were found exclusively deep (all refuge sizes combined) and 11 were only found shallow (Table 1).

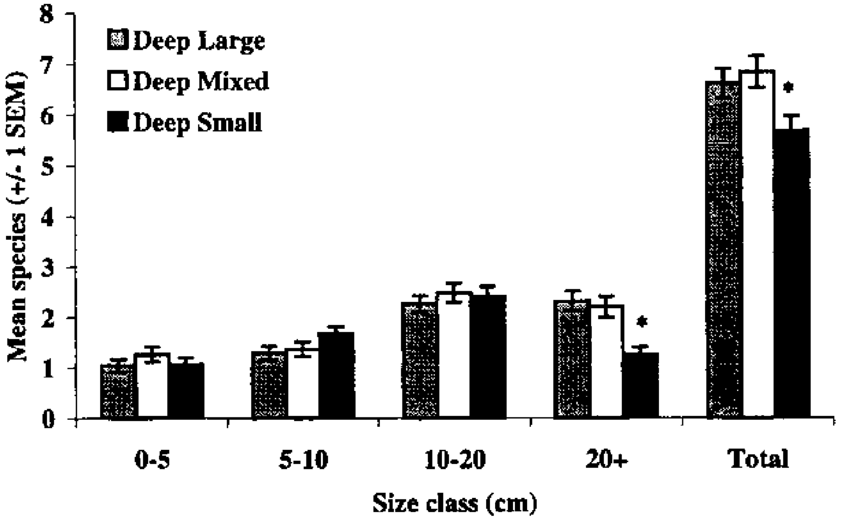


Figure 4. Mean number of total species (+/- 1 SEM) by size class on deep reefs with one of three treatments: large, small or mixed refuge sizes. Asterisked treatments are significantly different from other treatments within a size class ($p < 0.05$, SNK).

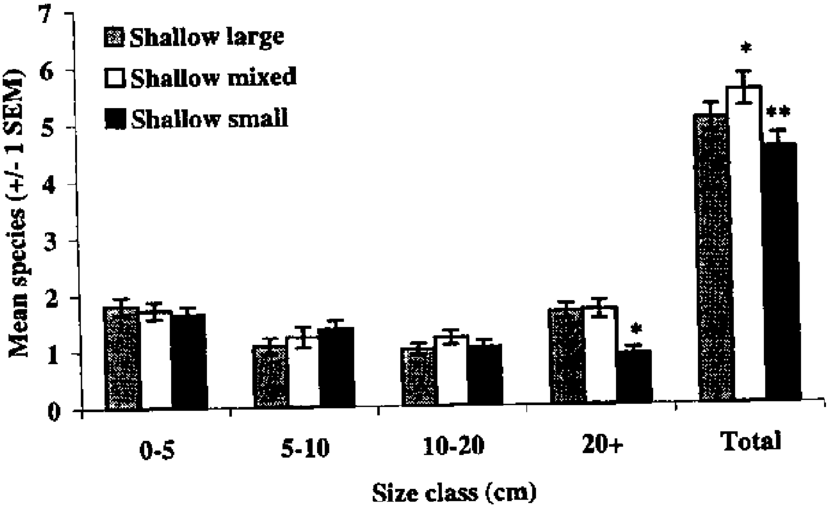


Figure 5. Mean number of total species (+/- 1 SEM) by size class on shallow reefs with one of three treatments: large, small or mixed refuge sizes. Asterisked treatments are significantly different from other treatments within a size class ($p < 0.05$, SNK).

Table 1. List of fishes

Common Name	Scientific Name	Deep	Shallow
STINGRAYS	DASYATIDAE		
Southern stingray	*Dasyatis americana*	X	
MORAY EELS	MURAENIDAE		
Purplemouth Moray	*Gymnothorax vicinus*	X	X
SEA BASSES	SERRANIDAE		
Black Grouper	*Mycteroperca bonaci*	X	
Scamp	*Mycteroperca phenax*	X	
Red Grouper	*Epinephelus morio*	X	
Graysby	*Epinephelus cruentatus*	X	
Coney	*Epinephelus fulvus*	X	
Sand Perch	*Diplectum formosum*	X	X
Butter Hamlet	*Hypoplectrus unicolor*	X	
Lantern Bass	*Serranus baldwini*	X	
Tobaccofish	*Serranus tabacarius*	X	
Harlequin Bass	*Serranus tigrinus*	X	
Tattler Bass	*Serranus phoebe*	X	
Belted Sandfish	*Serranus subligarius*	X	
CARDINALFISHES	APOGONIDAE		
Juvenile Apogonid	Apogon sp.	X	X
Flamefish	*Apogon maculatus*		
Twospot Cardinalfish	*Apogon pseudomaculatus*	X	X
JACKS	CARANGIDAE		
Juvenile Jacks	Carangid sp.		X
Blue Runner	*Caranx crysos*		X
Bar Jack	*Caranx ruber*		X
SNAPPERS	LUTJANIDAE		
Gray Snapper	*Lutjanus griseus*	X	X
Lane Snapper	*Lutjanus synagris*		X
Blackfin Snapper	*Lutjanus bucanella*	X	
Mutton Snapper	*Lutjanus analis*	X	X
GRUNTS	HAEMULIDAE		
Porkfish	*Anisotremus virginicus*	X	X
Juvenile Grunts	Haemulon sp.	X	X
Cottonwick	*Haemulon melanurum*	X	X
White Grunt	*Haemulon plumieri*		X
Tomtates	*Haemulon aurolineatum*	X	X
French Grunt	*Haemulon flavolineatum*	X	X
Bluestripe Grunt	*Haemulon sciurus*	X	X
Sailors Choice	*Haemulon parrai*	X	
Margate	*Haemulon album*	X	
PORGIES	SPARIDAE		
Saucereye Porgy	*Calamus calamus*	X	X

Table 1. Continued.

Common Name	Scientific Name	Deep	Shallow
Drums	SCIAENIDAE		
Highhat	*Equetus acuminatus*		X
Jacknifefish	*Equetus lanceolatus*	X	X
GOATFISHES	MULLIDAE		
Spotted Goatfish	*Pseudupeneus maculatus*	X	
BUTTERFLYFISHES	CHAETODONTIDAE		
Spotfin Butterflyfish	*Chaetodon ocellatus*	X	
Reef Butterflyfish	*Chaetodon sedentarius*	X	
ANGELFISHES	POMACANTHIDAE		
Queen Angelfish	*Holocanthus cilaris*	X	X
Blue Angelfish	*Holocanthus bermudensis*		X
Rock Beauty	*Holcanthus tricolor*	X	
French Angelfish	*Pomacanthus paru*	X	X
Gray Angelfish	*Pomacanthus arcuatus*	X	X
DAMSELFISHES	POMACENTRIDAE		
Dusky Damselfish	*Stegastes fuscus*	X	X
Bicolor Damselfish	*Stegastes partitus*	X	
Cocoa Damslefish	*Stegastes variabilis*	X	
Yellowtail Damselfish	*Microspathadon chrysurus*	X	
Sunshinefish	*Chromis insolatus*	X	
Yellowtail Reeffish	*Chromis enchrysurus*	X	
Purple Reeffish	*Chromis scotti*	X	
Blue Chromis	*Chromis cyanis*	X	
WRASSES	LABRIDAE		
Hogfish	*Lachnolaimus maximus*	X	X
Spotfin Hogfish	*Bodianus pulchellus*	X	
Spanish Hogfish	*Bodianus rufus*	X	
Clown wrasse	*Halichores maculipinna*	X	X
Slippery Dick	*Halichores bivittatus*	X	X
Puddingwife	*Halichores radiatus*	X	X
Yellowhead wrasse	*Halichores garnoti*	X	
Bluehead Wrasse	*Thalassoma bifasciatum*	X	X
PARROTFISHES	SCARIDAE		
Parrotfish	Scaridae spp.	X	X
Striped Parrot	*Scarus croicensis*	X	
Princess Parrot	*Scarus taeniopterus*	X	
Red tail Parrot	*Sparisoma chrospterum*	X	X
Redfin Parrot	*Sparisoma rubripinne*	X	
Redband Parrot	*Sparisoma aurofrenatum*	X	X
Stoplight Parrot	*Sparisoma virride*	X	
STARGAZER	DACTYLOSCOPIDAE		
Arrow Stargazer	*Gillellus greyae*	X	

Table 1. Continued.

Common Name	Scientific Name	Deep	Shallow
Combtooth Blennies	BLENNIDAE		
Redlip Blenny	*Ophioblennius atlanticus*		X
Barred Blenny	*Hypleurochilus bermudensis*		X
Seaweed Blenny	*Parablennius marmoreus*	X	X
GOBIES	GOBIIDAE		
Neon Goby	*Gobiosoma oceanops*	X	X
Bridled Goby	*Coryphopteru glaucofraenum*	X	X
Masked/Glass Goby	*Coryphopterus hyalinus/personatus*	X	
Goldspot Goby	*Gnatholepis thompsoni*	X	
SURGEONFISHES	ACANTHURIDAE		
Ocean Surgeon	*Acanthurus bahianus*	X	X
Doctorfish	*Acanthurus chirurgus*	X	X
Blue tang	*Acanthurus coeruleus*	X	X
LEATHERJACKETS	BALISTIDAE		
Orangespotted Filefish	*Cantherhines pullus*	X	X
Whitespotted Filefish	*Cantherhines macrocerus*		
Planehead Filefish	*Monocanthus hispidus*	X	X
Gray Trigger	*Balistes capriscus*	X	X
BOXFISHES	OSTRACIIDAE		
Scrawled cowfish	*Lactophrys quadricornis*	X	
Honeycomb cowfish	*Lactophrys polygonia*		X
Spotted trunkfish	*Lactrophrys trigonus*	X	X
Smooth trunkfish	*Lactrophrys triqueter*	X	X
PUFFERS	TETRAODONTIDAE		
Sharpnose Puffer	*Canthigaster rostrata*	X	X
Bandtail Puffer	*Sphoeroides spengleri*	X	
SPINY PUFFERS	DIODONTIDAE		
Balloonfish	*Diodon holocanthus*	X	
	Species per site	77	49
	Depth Exclusive Species	40	11
	Total species	88	

Some of the reefs positioned at the northern and western edges of the deep site (20 m) were placed near or on a destroyed tire reef. Thus, at the deep site, the immediate substrate varied based on the individual reefs proximity to a varying number of tires. This varying number of additional refuge spaces may have affected the study. However, statistical correlation of number of tires at distances of 5 m or less from the individual reefs against the total number of fishes (all species combined, $r^2 = -0.02$) or number of species ($r^2 = -0.04$) on the reefs was not significant ($p < 0 .05$, F-test). Therefore, treatment comparisons within the site were presumed to be equally affected by the presence of tires and data from the deep site were analyzed irrespective of tire data. However, the presence of the tire substrate at the deep site but not at the shallow site makes comparisons between reefs at the two sites questionable

.DISCUSSION

Refuge Size

Because the fishes associated with small refuge reefs did differ statistically in some size ranges from reefs with large refuges, in both total number of fishes and number of species, we accept the first prediction (Artificial reefs with large refuges will have different fish abundance, species richness, and biomass than reefs with small refuges). This supports the hypothesis that refuge size is an important aspect of artificial reef design for determining the associated assemblage of fishes. However, in this study small refuge reefs had lower numbers of total fishes and species, including juveniles, than large refuge reefs. These results contradict other studies where larger numbers of small fishes were associated with reefs with small refuges (Shulman 1984, Hixon and Beets 1989). Thus, although the results from this support the hypothesis that shelter size is important in artificial reef design, additional research is required to determine what those sizes should be relative to the local area and the species of interest.

Complexity

A basic premise of the experimental design in this study is: reefs with two different refuge sizes are more complex than identical reefs with a single refuge size. Mixed refuge reefs differed from small refuge reefs in numbers of fishes, species combined and numbers of species. However, the mixed refuge reefs never differed from large refuge reefs. In addition, there was no readily apparent differences in the species composition between large or mixed refuge reefs. These results, therefore, do not lend extensive support to our second prediction that mixed refuge reefs will have greater fish abundance, and species richness than either large refuge reefs or small refuge reefs.

Site Dependent Differences

There was a clear difference in species composition between the deep and shallow sites. It is not clear how much of this difference is due to differences in the substrate between the two sites (e.g. presence or absence of tires). However, similar site dependent differences in species composition were found in another study, using a different reef design, at similar depths (Sherman et al. 1999) where the results were not confounded by tire substrate.

This study was designed, in part, to replicate work done by Hixon and Beets (1984) in St. Thomas, VI, examining refuge size and fish assemblage formation. In their study, Hixon and Beets (1984) found a significant difference in species composition based on refuge size (i.e.: large holes – large fishes, small holes – small fishes). In that study, the majority of the large fishes on the reefs were piscivores (groupers). They found a negative relationship between resident piscivores and small fishes. In our study the correlation between fish size and refuge size, was not clear. The large refuge reefs had both more large fishes and more small fishes than the small refuge reefs. The primary difference in the results between these two studies appears to be driven by species composition. In St. Thomas, Hixon and Beets (1984) found the fish assemblages to be shaped by the presence of resident predators (groupers) while in South Florida, the fish assemblages were made up of primarily large (> 20 cm) herbivores (surgeonfishes), with few large resident piscivores.

CONCLUSION

The results of this study highlight again, the importance of site dependent factors (Sherman et al. 1999, Sherman et al. in press). Artificial reefs that exhibit specific results in one location may not necessarily yield the same results in another even within a limited geographical area. Additionally, an important recommendation to come out of this type of research, with regional comparisons in mind, is the importance of designing reefs with refuge sizes appropriate for the local species and life history stages being managed.

ACKNOWLEDGEMENTS

This study was partially funded by grants from Broward County Department of Planning and Environmental Protection, Biological Resources Division, Marine Resources Section and the Florida Department of Environmental Protection (OFMAS 050). South Florida Sport Fishing Classic Inc. and Patty Carr were instrumental in obtaining ancillary research funding for this project and RLS was partially funded by the Aylesworth Foundation for the Advancement of Marine Sciences. Nova Southeastern University Oceanographic Center graduate students provided unpaid, volunteer dive support and were essential to the completion of this project.

LITERATURE CITED

Bohnsack, J.A. and D.L. Sutherland 1985. Artificial reef research: A review with recommendations for future priorities. *Bull. Mar. Sci.* **37**(1):11-39.

Bohnsack, J.A., and D.E. Harper. 1988. Length-weight relationships of selected marine reef fishes from the southeastern United States and the Caribbean. NOAA Technical Memorandum. NMFS-SEFC-215, 31 pp.

Bohnsack, J.A. 1990. Habitat structure and the design of artificial reefs. Pages 412-426 in: S. Bell, E. McCoy, and H. Mushinsky (eds.) *Habitat*

Structure: The physical arrangement of objects in space. Chapman and Hall, N.Y., N.Y.

Bohnsack, J.A., D.L. Johnson and R.F. Ambrose. 1991. Ecology of artificial reef habitats and fishes. Pages 61-107 in: Seaman, W. Jr. and L.M. Sprague (eds.) *Artificial habitats for marine and freshwater fisheries.* Academic Press. San Diego, Ca. 285 pp.

Brock, R.E. and J.E. Norris 1989. An analysis of the efficacy of four artificial reef designs in tropical waters. *Bull. Mar. Sci.* **4**4(2):934-941.

Eckert, G.J. 1985. Settlement of coral reef fishes to different natural substrata at different depths. Proceedings of the Fifth International Coral Reef Congress, Tahiti, **5**.

Eklund, A.M. 1996. *The effects of post-settlement predation and resource limitation on reef fish assemblages.* Ph.D. Dissertation. University of Miami, Miami, Florida, USA.

Friedlander, A.M. and J.D. Parrish 1998. Habitat characteristics affecting fish assemblages on a Hawaiian coral reef. *J. Exp. Mar. Biol. Ecol.* **224**:1-30.

Gascon, D. and R.A. Miller 1982. Space utilization in a community of temperate reef fishes inhabiting small experimental artificial reefs. *Can. J. Zool.* **60**.

Gorham, J.C. and W.S. Alevizon 1989. Habitat complexity and the abundance of juvenile fishes residing on small scale artificial reefs. *Bull. Mar. Sci.* **4**4(2):662-665.

Hixon, M.A. and J.P. Beets 1989. Shelter characteristics and Caribbean fish assemblages: experiments with artificial reefs. *Bull. Mar. Sci.* **44**(2): 666-680.

Molles, M.C. 1978. Fish species diversity on model and natural reef patches: experimental insular biogeography. *Ecol. Monog.* **48**: 289-305.

Roberts, C.M. and R.F.G. Ormond 1987. Habitat complexity and coral reef diversity and abundance on Red Sea fringing reefs. *Mar. Ecol. Prog. Ser.* **4**1:1-8.

Seaman, W. 1996. Does the level of design influence success of an artificial reef? Proceedings of the 1st Conference of the European Artificial Reef Research Network, Ancona, Italy.

Sherman, R.L., D.S. Gilliam and R.E. Spieler 1999. A preliminary examination of depth associated spatial variation on small artificial reefs. *J. Appl. Ichthyol.* **15**: 116-121

Sherman, R.L., D.S. Gilliam and R.E. Spieler 1999. Site Dependent Differences in Artificial Reef Function: Implications for Coral Reef Restoration. *Bull. Mar. Sci.* In press.

Shulman , M.J. 1984. Resource limitation and recruitment patterns in a coral reef assemblage. *J. Exp. Mar. Biol.* 74:85-109.

Spieler, R.E. 1998. Artificial reef design: complexity and shelter design. Contract Completion Report submitted to Broward County Department of Natural Resource Protection.

Spieler, R.E., Gilliam, D.S. and R.L. Sherman. Artificial substrate and coral reef restoration: what do we need to know to know what we need. *Bull. Mar. Sci.*. In press.

Mangrove Habitat as Nursery Grounds for Recreationally Important Fish Species – Great Pond, St. Croix, U.S. Virgin Islands

WILLIAM J. TOBIAS
Department of Planning and Natural Resources
Division of Fish and Wildlife
Lagoon Street Complex, Room 203
Frederiksted, St. Croix, U.S. Virgin Islands 00841

ABSTRACT

The importance of fringe red mangrove (*Rhizophora mangle*) habitat as nursery grounds for recreationally important fish species was assessed in Great Pond, a 0.5 km square mangrove-lined, tidal pond on the south coast of St. Croix. Fish species composition and abundance were quantitatively sampled monthly in four areas of mangrove prop-root and pond habitat over a two-year period, using standardized fish traps and seine netting methods. A total of 1,403 fish were caught in traps ,representing 18 species and 12 families. The familiy Lutjanidae had the highest abundance (46.4%), followed by the families Gerridae (36.9%) and Carrangidae (10.0%). *L. apodus*, *G. cinereus*, *E. jonesi*, *C. latus* and *L. griseus* were the five most abundant species caught in order of abundance in the four trap areas. Mean number of species and individuals caught per trap and overall abundance of the five most common species showed differences by area relative to mangrove cover. A total of 1,433 individuals, representing 17 species and 13 families, were caught in seine net samples. The family Gerreidae had the highest abundance (92.1%). Permit (*Trachinotus falcatus*), snook (*Centropomus undecimalis*) and bonefish (*Albula vulpes*) were present in trap and net samples but in low abundance. Length-frequency distributions of the most abundant species caught in trap and net samples verify the importance of mangrove habitat for juvenile fish species.

KEY WORDS: Fish, mangroves, nursery

INTRODUCTION

Mangrove ecosystems develop in low-lying coastal embayments and are uniquely tolerant to high saline conditions. Mangrove ecosystems are extremely productive and support a high diversity of fish, birds and wildlife (UNESCO 1983). Mangrove lagoons are important habitat for juvenile of many fish species (Heald and Odum 1970, Austin 1971a and 1971b, Austin and Austin 1971, Cintron-Molero 1987, Thayer et. al. 1987, Boulon 1992, Adams and Tobias 1993, Tobias 1996) and can provide nursery areas for estuarine as well as reef fishes (Odum et. al. 1982, Boulon 1985 and 1992, Adams and Tobias 1993, Tobias 1996). Many juveniles use detritus and mangrove-associated invertebrates and fish as a food source (Zieman et. al. 1984, Thayer et. al. 1987). The complex prop-root habitat may also provide protection from predation (Orth et. al. 1984, Sogard and Olla 1993). In addition to providing important habitat, mangroves filter sediment and help maintain the integrity of the lagoon seagrass habitat (Cintron-Molero 1987), also an important nursery area (Dennis 1992). The utilization of mangrove habitats by recreationally and commercially valuable species and their prey species is important (Robertson and Duke 1987).

Mangrove habitat in the U.S. Virgin Islands is primarily mangrove fringe along lagoons and oceanic bays (Boulon 1992). Mangrove ecosystems in the U.S. Virgin Islands have been drastically altered by coastal development in the recent past without due consideration to their ecological significance. The remaining vestigial mangrove ecosystems are under severe stress due to natural and man-induced perturbations. Three prominent mangrove systems remain on St. Croix: Salt River, Altona Lagoon and Great Pond.

Great Pond, located on the south coast of St. Croix (Figure 1) is approximately 48 hectares in size (approximately 0.5 square kilometers); 30 hectares which continuously retain water and an additional 18 hectares of associated wetlands. Red mangroves fringe 640 linear m of shoreline with the heaviest densities occurring along the entrance channel and the southeastern end of the pond. The largest red mangroves, approximately 5 m in height, occur in the entrance channel. More than 500 red mangrove islets have formed within the pond , some of which contain red mangroves that are in excess of 3 m in height and have well-developed prop-root systems. A band of black mangroves (*Avicennia nitida*), 2 m to 20 m in width and 2 to 3 m in height, borders the remainder of the pond, except for the west end mud flat which is devoid tree vegetation (Knowles 1996). Pond depth is shallow, averaging 30 - 50 cm (Bruce et. al. 1989). The bottom of the pond is covered with an average of 30 cm of hydrogen sulfide-rich mud.

Great Pond is separated from Great Pond Bay to the south by a 1-km long baymouth bar, 25 to 100 m in width. Seawater exchange from Great Pond Bay into Great Pond is limited to a channel, 4 m wide and 1.5 m deep, at the southeastern corner of the pond.

The purpose of this study was to quantitatively determine the species composition and abundance of recreationally important fish species occurring in the mangrove fringe habitat of Great Pond.

METHODS

The fish community associated with red mangrove prop-root and pond habitat was sampled monthly with standardized fish traps and seine net. Due to the shallowness of Great Pond and high water turbidity, visual fish censuses were not possible. Data were analyzed for a 24-month period from 1 December 1995 - 31 December 1997.

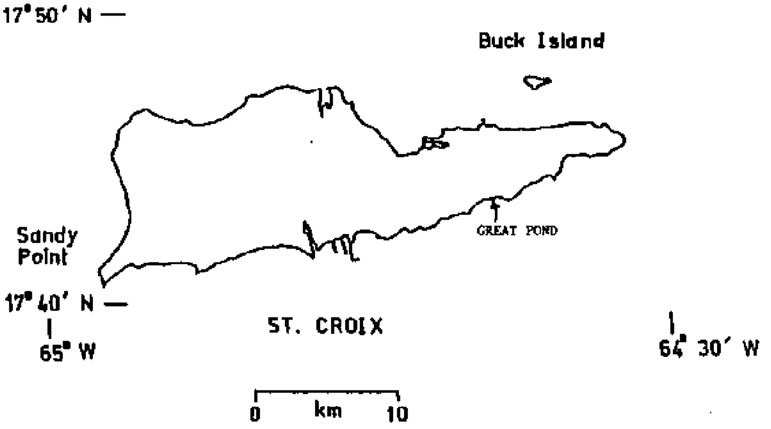


Figure 1. Great Pond location map

Four trap sampling areas were selected to sample mangrove prop-root and open pond habitat (Figure 2) and six trap sites per area were established and marked with PVC stakes. Six standardized rectangular fish traps, 92 cm x 57 cm x 19 cm, made from vinyl-coated 1.3 cm wire mesh, were baited with herring and set at the sampling sites. Each of the six sampling sites in the four areas was sampled once per month for a twenty-four hour period during consecutive days. All fish caught in the traps were identified, enumerated, measured (fork length and total length) and returned to the capture site.

Seine net sampling areas, 15 - 20 m in diameter, were selected in open areas between mangrove islets. The location of the sampling areas varied during the study. Four circular hauls in a north/south direction across the pond were made with a nylon seine net measuring 30.5 m x 122 cm x 1.3 cm. Upon enclosing an area with the net, the net was hauled towards the overlapping ends. All fish caught in the net were identified, enumerated, measured (fork length and total length) and returned to the capture site.

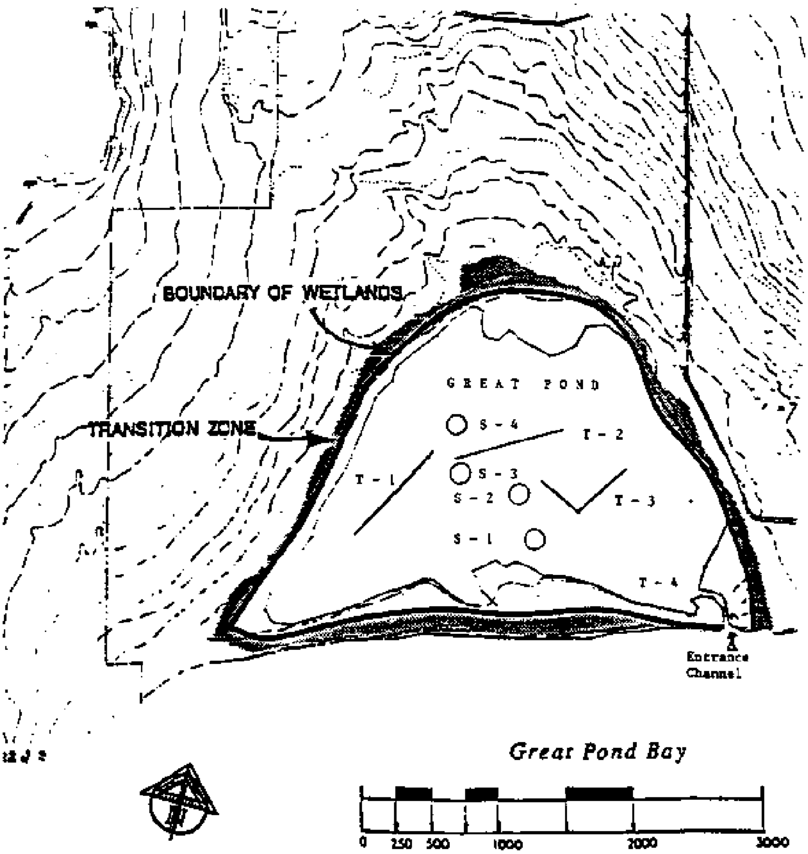


Figure 2. Great Pond study site map. The legend is measured in feet.

The number of species and number of fish caught per trap area were analyzed with a Kruskal-Wallis one-way ANOVA on ranks (Sokal and Rohlf 1981), after log(x+1) and square root transformations failed to pass Kolmogorov-Lilliefors normality test. A Dunn's Method pair-wise multiple comparison procedure was used to isolate trap areas that differ from others. Species were ranked in order of total abundance and the most abundant recreationally important species caught by trap (5) were examined for between-site variation in abundance with a Kruskal-Wallis one-way ANOVA , after log(x+1) and square root transformations failed to pass Kolmogorov-Lilliefors normality test. A Dunn's pair-wise multiple comparison procedure was used to isolate trap areas that differ from others.

RESULTS

Water depth measured at the fish trap sampling locations in the pond ranged from 29.3 - 42.0 cm in Area 1, 19.8 - 33.0 cm in Area 2, 13.0 - 30.8 cm in Area 3 and 31.0 - 90.0 cm in Area 4.

A total of 1,403 fish were caught in traps in Great Pond, representing 18 species and 12 families. The family Lutjanidae had the highest abundance (46.4%) (Figure 3), represented by two species: *Lutjanus apodus* and *Lutjanus griseus*. Of these two species, *L. apodus* and *L. griseus* represented 87.9% and 12.0%, respectively, of the total Lutjanidae abundance and 40.8% and 5.6%, respectively, of the total fish abundance caught in traps. The family Gerridae was second in family abundance (36.9%), represented by two species: *Gerres cinereus* and *Eucinostomus jonesi*. Of these two species, *G. cinereus* and *E. jonesi* represented 72.0% and 28.0%, respectively, of the total Gerridae abundance and 26.6% and 10.3%, respectively, of the total fish abundance caught in traps. Carangidae was third in family abundance (10.0%), represented by two species: *Caranx latus* and *Trachinotus falcatus*. *C. latus* represented 92.2% of the carangids. All other families had a relative abundance of < 3.0%.

Three species of crustaceans were caught in fish traps. Of these species, portunid crabs, spiny lobsters and penaeid shrimp represented 96.9%, 2.8% and 0.2% of the crustaceans caught. Portunid crabs were more abundant in the pond interior. Spiny lobsters were more abundant in the red mangrove-lined channel connecting Great Pond to Great Pond Bay.

The mean number of species and individuals caught per trap was significantly different for Area 4 vs Area 2 and Area 4 vs Areas 2 and 3, respectfully, based on a Dunn's Method of all pair-wise multiple comparison test (Table 1). Area 4 was located along the entrance channel into Great Pond where seawater exchange was the greatest and where the mangrove prop-root community was the most well-developed. Mean number of species and mean number of individuals per trap was greatest in Area 4 (1.31 and 3.98, respectively) and least in Area 2 (0.97 and 1.85, respectively) (Figures 4 and 5).

Table 1. Dunn's Method of pair-wise multiple comparison of means of species and number of individuals per trap by area in Great Pond.

Treatment	Area Comparison	Rank Difference	P	Q	P < 0.05*
No. Species	4 vs 2	51.193	4	2.737	S
	4 vs 3	25.629	3	1.374	NS
	4 vs 1	8.725	2	0.454	NTN
	1 vs 2	42.468	3	2.311	NS
	1 vs 3	16.903	2	0.923	NTN
	3 vs 2	25.565	2	1.421	NTN
No. Fish	4 vs 2	82.810	4	4.302	S
	4 vs 3	61.529	3	3.207	S
	4 vs 1	45.005	2	2.300	NS
	1 vs 2	37.805	3	1.999	NS
	1 vs 3	16.524	2	0.877	NTN
	3 vs 2	21.281	2	1.149	NTN

* At 95% confidence limits, S = significant, NS = not significant, NTN = no test needed

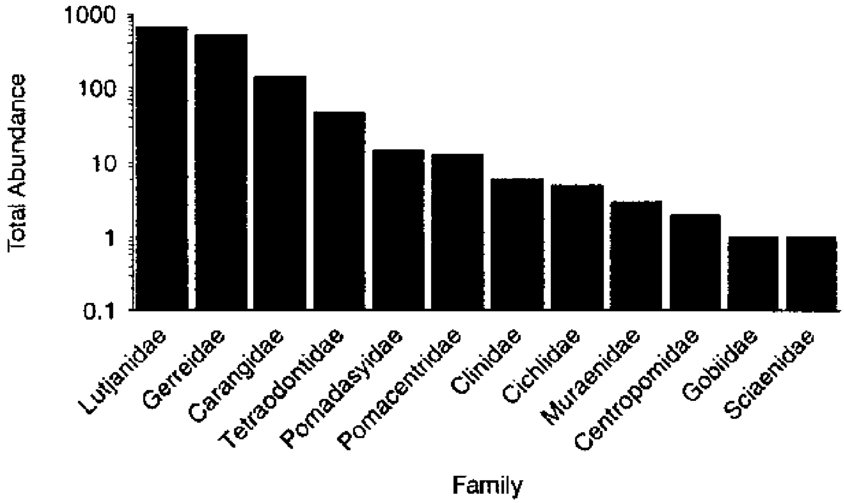


Figure 3. Fish family abundance - Great Pond traps

L. apodus, G. *cinereus*, *E. jonesi*, *C. latus* and *L. griseus* were the five most abundant species caught in order of abundance in the four trap areas (Table 2). G. *cinereus* and *C. latus* were most abundant in Area 1. In Areas 2 and 3, *G. cinereus*. and *L. apodus* were most abundant. *L. apodus* and *L. griseus* were most abundant in Area 4. Kruskal-Wallis one-way analysis of variance of total abundance for the five most abundant species indicated significant differences in median values of species among areas (*L. apodus* - H = 108.051, df = 3, p = <0.001; G. *cinereus* - H = 26.759, df = 3, p =<0.001; *E. jonesi* - H = 14.158, df = 3, p = 0.003; *C. latus* - H = 40.800, df = 3, p =<0.001; and *L. griseus* - H = 34.826, df = 3, p = <0.001). Dunn's Method of all pair-wise multiple comparison test (Table 3) shows a statistically significant difference for the following: *L. apodus* in Area 4 vs Area 1 and Area 2, *G. cinereus* in Area 1 vs Area 4 and Areas 2 & 3 vs Area 4, *E. jonesi* in Area 1 vs Area 4 and Area 2 vs Area 4), *C. latus* in Area 1 vs Areas 2, 3 and 4 and Area 3 vs Area 4 (), and *L. griseus* in Area 4 vs Areas 1, 2 and 3. Red mangrove cover increased from west to east (Area 1 to Area 4) in Great Pond. *L apodus* and *L. griseus* were most abundant where mangrove cover was the most dense (Area 4). *G. cinereus*, *E. jonesi* and *C. latus* were most abundant in more open areas where mangrove cover was sparse (Areas 1 and 2). The mean number of fish and mean number of species caught per trap were low, making annual recruitment events difficult to determine.

Table 2. Total abundance of the five most abundant fish species by area caught in traps in Great Pond.

	AREA				
Species	**1**	**2**	**3**	**4**	**Total**
L. apodus	43	53	86	388	572
Mean	0.33	0.36	0.57	2.98	
+/-SE	0.06	0.07	0.09	0.43	
G. cinereus	116	112	109	36	373
Mean	0.85	0.76	0.73	0.28	
+/-SE	0.85	0.12	0.11	0.07	
E. jonesi	43	47	51	4	145
Mean	0.31	0.32	0.34	0.03	
+/-SE	0.09	0.08	0.14	0.02	
C. latus	70	23	34	3	130
Mean	0.51	0.16	0.23	0.02	
+/-SE	0.10	0.04	0.06	0.02	
L. griseus	24	7	3	44	78
Mean	0.18	0.05	0.02	0.34	
+/-SE	0.18	0.02	0.01	0.07	
			TOTAL:	1,298	

Trap Number: Area 1 = 137, Area 2 = 148, Area 3 = 150, Area 4 = 130.

A total of 1,433 individuals, representing 17 species and 13 families, were caught in seine net samples (Figure 6). The family with the highest relative abundance was Gerreidae (92.1%), represented by four species, *E. jonesi*, *G. cinereus*, *Eucinostomus sp.* and *Diapterus olisthostomus*. *E. jonesi* accounted for 70.5% of all gerreids. All other families had a relative abundance of <2%. Second in abundance was Clupeidae (1.6%), represented by one species, *Opisthonema oglinum*. The families Carangidae (*Caranx latus*), Lutjanidae (*Lutjanus griseus* and *L. apodus*), Centropomidae (*Centropomus undecimalis*), and Cichlidae (*Tilapia mossambica*), had a relative abundance of 1.2, 1.2, 1.1 and 1.0%, respectively. Portunid crabs and penaeid shrimp were also caught in monthly seine net hauls.

The total abundance for the six most abundant fish species caught in seine net hauls is shown in Table 4. *E. jonesi* and *G. cinereus* were caught in greater abundance in net hauls near the shorelines in open areas, as opposed to areas with mangrove islets within the pond interior. Peak abundance of *E. jonesi* was observed during July and November of 1996 and April, July and November of

1997. Primary abundance of *G. cinereus* occurred during the months of May and July in 1997 with a secondary peak in October and November of 1996 and 1997.

Figures 7 and 8 show the length-frequency histograms for the most abundant fish species cauaght in trap and net samples. A comparison of the mean size of these and other recreationally important fish species caught in Great Pond to minimum size at sexual maturity (males) for the same species (as found in literature references) is shown in Table 5. With the exception of *E. jonesii* and *O. oglinum*, which may mature at a smaller size, 100% of the remaining eight species caught were juveniles.

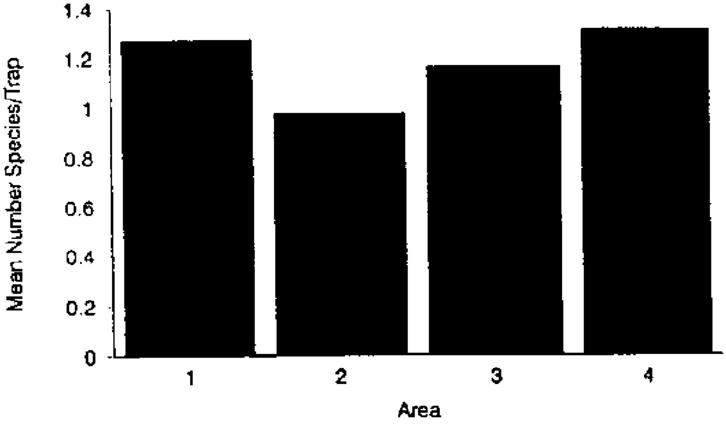


Figure 4. Mean numer of fish species/trap in Great Pond

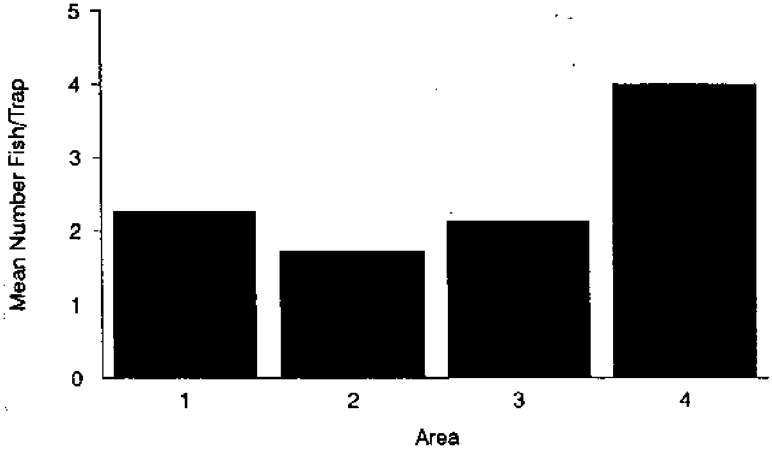


Figure 5. Mean number of individual fish/trap in Great Pond

Table 3. Dunn's Method of pair-wise multiple comparison of means for the five most abundant species by area caught in traps in Great Pond.

Species	Area of Comparison	Rank Difference	P	Q	P < 0.05*
L. apodus	4 vs 1	155.462	4	9.043	S
	4 vs 2	150.972	3	8.946	S
	4 vs 3	122.075	2	7.256	S
	3 vs 1	33.387	3	2.008	S
	3 vs 2	28.897	2	1.773	NTN
	2 vs 1	4.490	2	0.269	NTN
G. cinereus	1 vs 4	81.874	4	4.925	S
	1 vs 2	22.368	3	1.387	NS
	1 vs 3	21.500	2	1.337	NTN
	3 vs 4	60.375	3	3.712	S
	3 vs 2	0.868	2	0.551	NTN
	2 vs 4	59.506	2	3.647	
E. jonesi	1 vs 4	37.097	4	3.289	S
	1 vs 3	9.066	3	0.831	NS
	1 vs 2	1.174	2	0.107	NTN
	2 vs 4	35.923	3	3.245	S
	2 vs 3	7.892	2	0.738	NTN
	3 vs 4	28.032	2	2.510	NS

Table 3 continued.

Species	Area of Comparison	Rank Difference	P	Q	P < 0.05*
C. latus	1 vs 4	72.188	4	6.243	S
	1 vs 2	48.133	3	4.291	S
	1 vs 3	42.174	2	3.772	S
	3 vs 4	30.014	3	2.653	S
	3 vs 2	5.959	2	0.544	NS
	2 vs 4	24.055	2	2.119	NS
L. griseus	4 vs 3	53.251	4	5.485	S
	4 vs 2	45.717	3	4.695	S
	4 vs 1	29.764	2	3.000	S
	1 vs 3	23.487	3	2.448	NS
	1 vs 2	15.953	2	1.658	NTN
	2 vs 3	7.534	2	0.801	NTN

* At 95% confidence limits, S = significant, NS = not significant, NTN = not test needed.

Table 4. Total abundance for the six most abundant fish species caught in seine net hauls in Great Pond.

	AREA				
Species	**1**	**2**	**3**	**4**	**Total Abundance**
E. jonesi	244	209	242	315	1,010
Mean	12.84	11.00	12.74	16.58	
+/-SE	3.49	2.47	3.06	5.71	
G. cinereus	84	63	71	73	291
Mean	4.42	3.32	3.74	3.84	
+/-SE	1.29	0.81	0.84	1.23	
*O. oglinum**	6	12	3	2	23
Mean	0.30				
+/-SE	0.11				
*C. latus**	3	4	6	4	17
Mean	0.22				
+/-SE	0.79				
*C. undecimalis**	2	2	8	4	16
Mean	0.21				
+/-SE	0.08				
*L. griseus**	3	-	3	10	16
Mean	0.21				
+/-SE	0.09				

*Mean and SE calculations computed for total species abundance from the four net areas combined.

Table 5. Comparison of mean size (mm) of the most abundant and recreationally important fish species caught in Great Pond to minimum sexual maturity. TL = Total Length, FL = Fork Length, SL = Standard Length.

Species	Sampling Method (TL)		Sexual Maturity
	Trap	**Seine Net**	
L. apodus	105.2 (488)*	108.0 (1)	250 FL (CFMC, 1998)
G. cinereus	87.3 (363)	103.5 (234)	190 TL (Claro, 1987)
E. jonesii	80.8 (134)	92.0 (444)	N/A
C. latus	103.9 (129)	133.6 (14)	340 TL (Claro, 1987)
L. griseus	105.2 (78)	140.6 (16)	185 SL (Starck & Schroeder, 1971)
O. oglinum	- (21)	108.9	N/A
C. undecimalis	- (2)	176.7 (16)	330 SL (Peters et al., 1998)
T. falcatus	112.8 (5)	-	N/A
A. vulpes	- (2)	141.5	418 FL (Crabtree et al., 1997)
S. barracuda	-	246.0 (2)	500 FL (DeSylva, 1963)

* Total number of fish.

N/A = Not Available

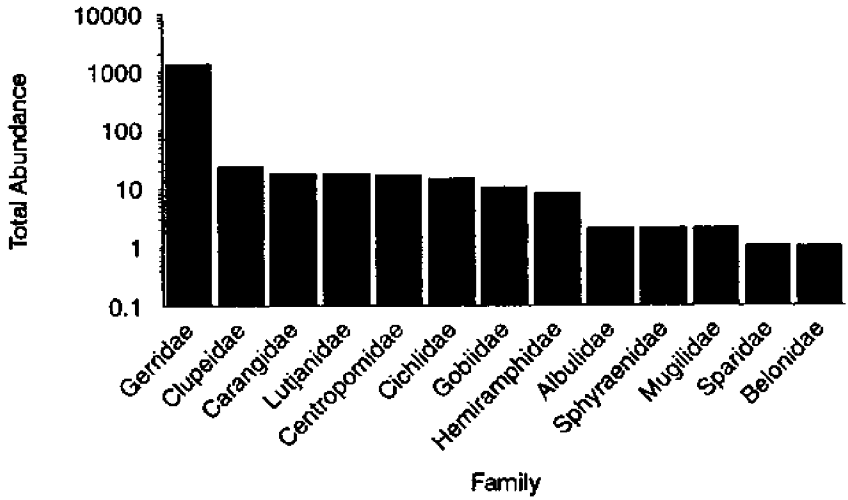
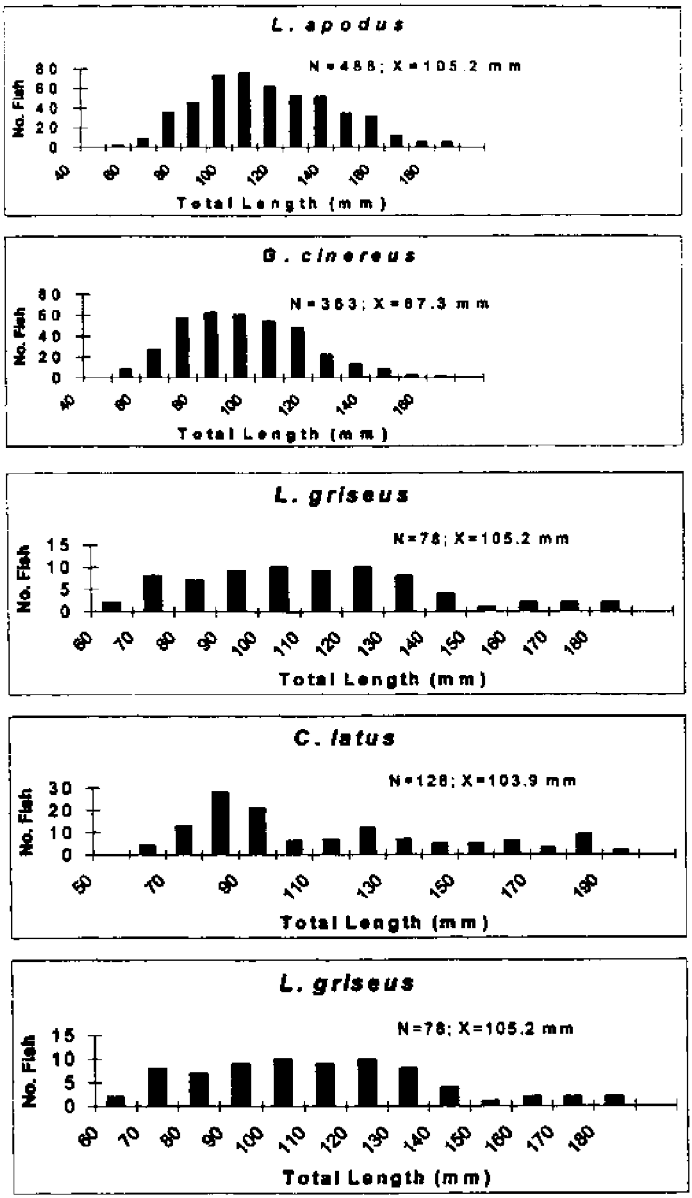


Figure 6. Fish family abundance - Great Pond seine net

DISCUSSION

The present study demonstrates that the fringing red mangrove community in Great Pond provides important nursery habitat for juveniles of reef fish species, many of which are recreationally important as primary target species or prey species for primary target species. Species diversity and abundance of fishes for Great Pond were lower than that observed in Salt River Bay and Altona Lagoon mangrove ecosystems on St. Croix (Adams and Tobias 1993, Tobias 1996); however, many of the most abundant species were common in all three mangrove areas. Lower species diversity and abundance of fishes in Great Pond may be due to greater fluctuations in temperature, salinity and dissolved oxygen due to shallow water depth, restricted seawater exchange and freshwater input (Tobias 1996). Larval recruitment may also be restricted by the limited volume of water entering the pond from Great Pond Bay through the entrance channel, which is reduced in width by terrestrial and calcium carbonate sediments and mangrove encroachment. Fish species richness and diversity may further be limited by the buildup of mud from terrestrial runoff (up to 30 cm in depth), which adversely impacts benthic community development.


L. apodus

G. cinereus

L. griseus

C. latus

L. griseus

Figure 7. Length-frequency histograms of most abundant trap species.

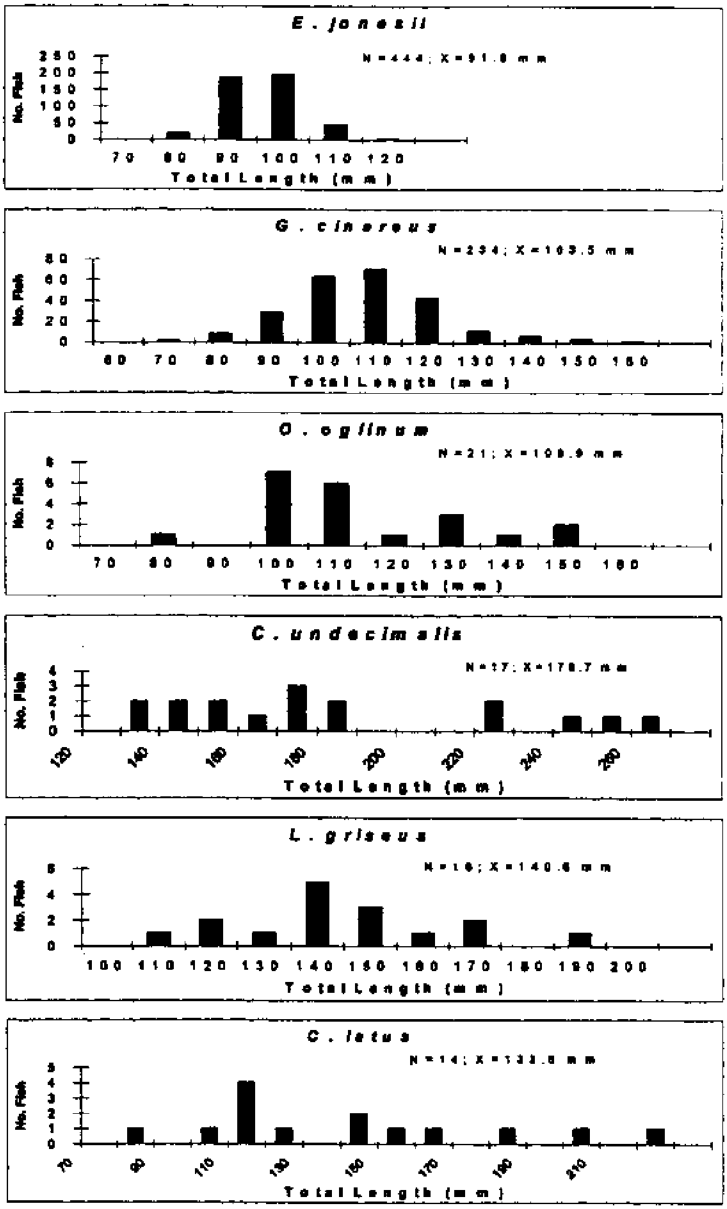


Figure 8. Length-frequency histograms of most abundant net species

Similar to Salt River and Altona Lagoon , the most abundant species caught in Great Pond by fish trap and seine net sampling techniques combined were *E. jonesi*, *G. cinereus*, *L. apodus* and *C. latus* and *L. griseus*. Lutjanids were most commonly caught in fish traps placed adjacent to red mangrove prop-roots, a forage area for crustaceans, invertebrates and small fishes and refuge area from predators. Gerrids were most commonly caught by traps and seine net over open areas of mud bottom, a forage area for infaunal invertebrates. Carangids (*C. latus* and *T. falcatus*) were also found in seine net samples over open areas. *Lutjanus apodus*, *L. griseus*, *C. latus* and *G. cinereus* are directly targeted in the recreational fishery. *E. jonesi* is a favorite baitfish species used for snappers and jacks. Snook, permit, and bonefish, important recreational sport fish not sampled in Salt River Bay and Altona Lagoon mangrove systems (Tobias 1996), were present in low abundance in Great Pond. Other predator species such as barracuda were present. White mullet, *Mugil curema*, were abundant in the Great Pond but were not sampled adequately by either trap or seine net collection methods due to their ability to avoid capture. *Tilapia, mossambica*, a species capable of withstanding temperature, salinity and dissolved oxygen extremes, were found Great Pond.

The fish community in Great Pond was similar to other mangrove lagoon communities (Baelde 1990, Van der Velde et al. 1992, Rooker and Dennis 1991, Thayer et al. 1987, Tzeng and Wang 1992, and Dennis 1992). Lutjanids, primarily found on reefs as adults, are common in mangrove lagoons as juveniles (Baelde 1990, Van der Velde et al. 1992, Rooker and Dennis 1991). Gerreids are found in many estuarine systems throughout the world, including mangrove lagoons (Matthes and Kapetsky 1988, Baelde 1990, Rooker and Dennis 1991, Thayer et al. 1987).

The abundance of juvenile fishes in trap and seine net samples from Great Pond supports the hypothesis that the fringing red mangrove prop root community and mangrove islets serve as important nursery habitat (CFMC 1998, Matthes and Kapetsky 1988). The mean length of individuals and the juvenile/adult ration remained relatively constant over time. With the exception of *E. jonesi* and *O. oglinum*, the length-frequency histograms were highly skewed toward smaller fish. This is similar to what was reported for fish in traps and visual censuses for Salt River Bay and Altona Lagoon (Adams and Tobias 1993, Tobias 1996). Matthes and Lapetsky (1988) indicated that reef fish species similar to those found in Great Pond are dependent on mangrove habitat during their juvenile-adult stage. The complex habitat afforded by the mangrove prop-root community reduces predation (Orth et al. 1984, Sogard and Olla 1993, Hixon 1991) and increases the overall number of small fish (Hixon and Beets 1993).

Similar species abundances were reported from monthly trap and seine net data. Recruitment trends based on monthly trap abundance values were not discernable due to insufficient numbers of individuals caught. Only *E. jonesi* and *G. cinereus* were present in sufficient numbers from net samples to follow possible recruitment trends. Recruitment events varied from year to year and may be dependent upon wet and dry cycles affecting pond salinity and temperature characteristics.

LITERATURE CITED

Adams, A.J. and W.J. Tobias. 1993. Red mangrove prop-root habitat as a finfish nursery area. A case study of Salt River, St. Croix, U.S.V.I. *Proc. Gulf Carib. Fish. Inst.* **46**:22-47.

Austin, H.M. 1971. A survey of the ichthyofauna of the mangroves of western Puerto Rico during December 1967 - August 1968. *Carib. Jour. Sci.* **11**:27-39.

Austin, H.M. 1971a. The feeding habits of some juvenile marine fishes from the mangroves in western Puerto Rico. *Carib. Jour. Sci.* **11**:171-178.

Austin, H.M. and S.E. Austin. 1971b. Juvenile fish in two Puerto Rican mangroves. *Underwater Naturalist* **7**:26-30.

Baelde, P. 1990. Differences in the structures of fish assemblages in *Thalassia testudinum* beds in Guadeloupe, French west Indies, and their ecological significance. *Mar. Biol.* **105**:163-173.

Boulon, R.H. 1985. Distribution and species composition of fisheries habitats within the Virgin Islands Biosphere Reserve. Report submitted to the Virgin Islands Resources Management Cooperative. 52 pp.

Boulon, R.H. 1992. Use of mangrove prop root habitats by fish in the northern U.S. Virgin Islands. *Proc. Gulf Carib. Fish. Inst.* **41**:189-204.

Bruce, G.L., J.B. Harkness, J.S. Hewlett, M.S. Hill, D.K. Hubbard, T. McGovern, C.C. Reed and H.H. Roberts. 1989. Sedimentary environments of Great Pond Bay, St. Croix, U.S.V.I. Pages 161-165 in: Multer, H.G. and L.C. Gerhard (eds.) *Guidebook to the Geology and Ecology of Some Marine and Terrestrial Environments, St. Croix, U.S. Virgin Islands.* West Indies Laboratory Special Publication No. 5.

Caribbean Fishery Management Council. 1998. Draft - Essential fish habitat (EFH) generic amendment to the fishery management plans (FMPs) of the T.S. Caribbean. Vol. 1&2. 159 pp.

Cintron-Molero, G. 1987. Caribbean mangroves: a manager's guide. Government of the U.S. Virgin Islands, DPNR, Div. Fish Wild. 21 pp.

Dennis, G.H. 1992. Island mangrove habitats as spawning and nursery areas for commercially important fishes in the Caribbean. *Proc. Gulf Carib. Fish. Inst.* **41**:205-225.

Heald, E. and W. E. Odum. 1970. The contribution of mangrove swamps to Florida fisheries. *Proc. Gulf Carib. Fish. Inst.* **22**:130-135.

Hixon, M.A. 1991. Predation as a process structuring coral reef fish communities. in: P.F. Sale (ed.). *The Ecology of Fishes on Coral Reefs.* Academic Press, N.Y. 754 pp.

Hixon, M.A. and J.P. Beets. 1993. Predation, prey refuges and the structure of coral-reeffish assemblages. *Ecol Monogr.* **63**(1):77-101.

Knowles, W.C. 1996. Wildlife use of saltwater wetlands on St. Croix, U.S. Virgin Islands. Final Rept., Dept Int., U.S. Fish Wildl. Serv. 261 pp.

Matthes, H. and J.M. Kapetsky. 1988. Worldwide compendium of mangrove-associated aquatic species of economic importance. FAO Fish. Circ. No. 814.

Multer, H.G. 1974. Summary of the Holocene geology of the Great Pond area, southeast St. Croix. Pages 77-110 in: Multer, H.G. and Gerhard, L.C. (eds.) *Guidebook to the Geology and Ecology of Some Marine and Terrestrial Environments, St. Croix, U.S. Virgin Islands.* West Indies Laboratory Special Publication No..

Odum, W.E., C.C. McIvor and T.J. Smith, III. 1982. The ecology of the mangroves of south Florida: a community profile. U.S. Fish Wildl. Serv. FWS/OBS-81/24. 144 pp.

Orth, RJ, G.L. Heck and J. Van Montfrans, Jr. 1984. Faunal communities in seagrass beds: a review of the influence of plant structure and prey characteristics on predator-prey relationships. *Estuaries.* **7**:339-350.

Robertson, A.I. and N.C. Duke. 1987. Mangroves as nursery sites: comparisons of the abundance and species decomposition of fish and crustaceans in mangroves and other nearshore habitats in tropical Australia. *Mar. Biol.* **96**:193-205.

Rooker, J.R. and G.D. Dennis. 1991. Diel lunar and seasonal changes in a mangrove fish assemblage off southwestern Puerto Rico. *Bull. Mar. Sci.* **49**:(3):684-698.

Saenger, P. and C.C. McIvor. 1974. Water quality and fish populations in a mangrove estuary modified by residential canal developments. Int. Symp. Biol. Mngt. Mangroves. Honolulu.

Sogard, S.M. and B.L. Olla. 1993. The influence of predator presence on utilization of artificial seagrass habitats by juvenile walleye pollack, *Theragra chalcogramma*. *Environ. Biol. Fish.* **37**:57-65.

Sokal, R.R. and F.L. Rohlf. 1981. *Biometry*. W.H. Freeman & Co., San Francisco. 859 pp.

Thayer, G., D. Colby and W. Heetler. 1987. Utilization of the red mangrove prop root habitat by fishes in south Florida. *Mar. Ecol. Progr. Ser.* **35**:25-38.

Tobias, W.J. 1996. The determination of mangrove habitat for nursery grounds of recreational fisheries in Salt River and Altona Lagoon, St. Croix, U.S. Virgin Islands. Final Rept., Dept Int, U.S. Fish Wildl. Ser. 30 pp.

Tzeng, W.N. and Y.T. Wang. 1992. Structure, composition and seasonal dynamics of the larval and juvenile fish community in the mangrove estuary of Tanshui River, Taiwan. *Mar. Biol.* **113**:481-490.

UNESCO-I O CARIBE. 1983. *Coral reefs, seagrass beds and mangroves: their interaction in the coastal zones of the Caribbean.* Ogden and Gladfelter, eds. 113 pp.

U.S. Department of Interior. 1990. The Coastal Barriers Resource System: U.S. Virgin Islands. 15 pp.

Van der Velde, G., M.W. Gorissen, C. Den Hartog, T. van't Hoff and G.J. Meijer. 1992. Importance of the Lac-lagoon (Bonaire, Netherlands Antilles) for a selected number of reef fish species. *Hydrobiol.* **247**:139-140

Zieman, J.C., S.A. Mack and A.L. Mills. 1984. Role of seagrasses mangroves in estuarine food webs: temporal and spatial changes in stable isotope composition and amino acid content during decomposition. *Bull. Mar. Sci.* **35**:380-392.

Seasonal Colonization of Low Profile Artificial Reefs in Mississippi Coastal Waters: Vertebrates

KIRSTEN M. LARSEN, HARRIET M. PERRY,
JAMES S. WARREN, and CHRISTINE B. TRIGG
University of Southern Mississippi
Institute of Marine Sciences, Gulf Coast Research Laboratory
Ocean Springs, MS 39564 USA

ABSTRACT

The popularity of low profile reefs as fishing banks prompted the State of Mississippi to develop new artificial reefs and to augment existing oyster shell reefs. The creation of artificial fishing reefs from concrete rubble, crushed limestone, and oyster shells in Mississippi coastal waters provided an opportunity to obtain information on the colonization and utilization of these different substrates by benthic fauna. In this study, vertebrate colonization was compared between two substrate types, crushed limestone gravel and oyster shell. Colonization was determined by placing trays containing the reef material on an existing shell/gravel reef approximately 300 meters from shore in central Mississippi Sound. Every three months the trays were returned to the laboratory and all organisms were removed and identified to the lowest taxonomic level. Fish were measured to the nearest 0.1 mm total length and weighed to the nearest 0.01 g. Fish colonizing the reefs included members of the following families: Gobiidae (*Gobiosoma bosc*), Blenniidae (*Hysoblennius ionthas*), Gobiesocidae (*Gobiesox strumosus*), Ophichthidae (*Myrophis punctatus*), and Batrachoididae (*Opsanus beta*). The structural complexity of the reef appears to control population size structure and density. Differences in species composition and size may be related to the availability and size of "niches" provided by the oyster shell and limestone gravel. Although species composition between the two substrates was similar, significantly larger animals colonized the oyster shell than the crushed limestone gravel. Oyster shells provided fewer, but much larger niches than those found in crushed limestone gravel.

KEY WORDS: Estuarine, fish colonization, low profile artificial reefs

INTRODUCTION

Artificial reefs serve as fish attractants and may increase production of some species by increasing habitat. In an effort to enhance already established recreational fisheries and to increase numbers of and access to less common structure-associated fishes, Mississippi began building new low profile artificial reefs and augmenting existing ones. Although over twenty low profile artificial reefs (oyster shell, concrete rubble, limestone gravel) have been constructed in

Mississippi inshore waters, there are no data on reef community structure or the association of fish populations with these reefs. A study of the faunal assemblages associated with reef colonization in Mississippi Sound was begun in December 1998 as part of a larger program to assess productivity of these reefs in relation to recreational fishing opportunities. The research reported on herein is part of a long-term study addressing seasonal colonization/succession of fauna associated with limestone gravel and oyster shell reefs in estuarine waters of Mississippi Sound. Data represent initial colonization studies conducted in the summer of 1998 and the winter of 1998/99.

MATERIALS AND METHODS

Colonization was studied by placing a series of crates filled with $0.025m^3$ of limestone gravel or oyster shell on a newly created, nearshore limestone/shell reef. The area adjacent the artificial reef is characterized by small patch subtidal oyster beds. Average depth is approximately 1.5 m. Site location is shown in Figure 1. Thirty-two samplers (16 limestone, 16 oyster shell) were deployed in the summer of 1998. The crates were placed on eight plastic pallets, four trays to a pallet (Figure 2). Four pallets held crates filled with limestone and four pallets held crates with oyster shell. All crates and pallets were labeled. A schedule of sampling was established that removed one crate from each pallet (four limestone gravel and four oyster shell) after an initial soak time of 3 months. At six, nine, and twelve month intervals the remaining crates were pulled (four limestone gravel and four oyster shell crates per sampling period). Additionally, a set of crates was replaced (four limestone gravel and four oyster shell) each sampling period in order to obtain a three-month set of samples each season. Original sampling schedule called for removal of a set of three month samples and a set of three and six month

samples during summer and fall of 1998, respectively. Summer samples were collected; however, Hurricane Georges in September 1998 destroyed the pallets and no fall samples were collected. Samplers were re-deployed in December 1998. Data presented in this study are from the three month summer samples (original study) and the three month winter samples that began the new sampling regime.

Crates were removed from the water, immediately placed in seawater soaked oyster sacks and returned to the Gulf Coast Research Laboratory for processing. Contents of each crate were washed over screening and all organisms collected. Samples were frozen prior to analysis. Each sample was sorted to species and the total number and weight recorded. When available, fifty individuals of each species were randomly selected for measurement. Individuals were measured to the nearest 0.1 mm total length (TL) using digital calipers. A Sartorius analytical balance was used to measure weight to the nearest 0.001 g.

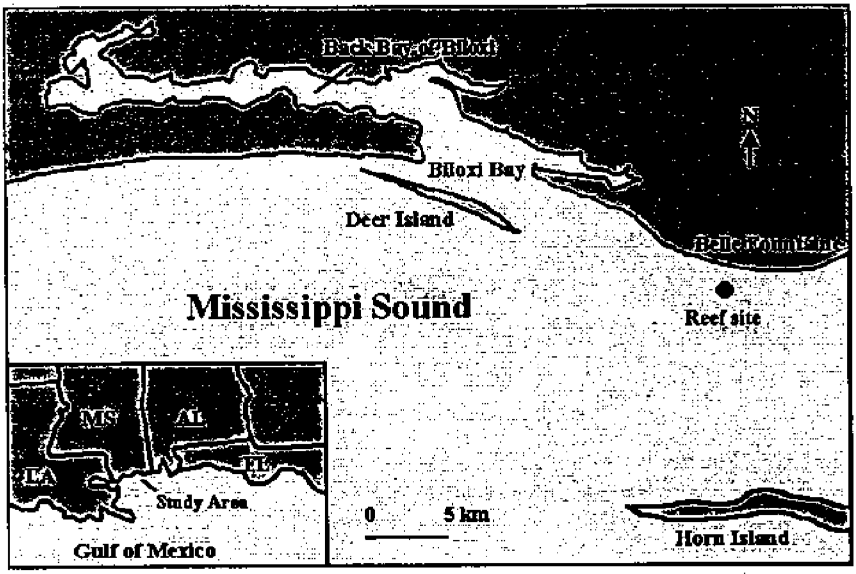


Figure 1. Location of low profile artificial reef.

Student's *t*-test (α=0.05) was used to compare species abundance and size between substrates within a season and substrates between seasons. Brillouin's diversity index (H) and Spearman's rank correlation (r_S) were calculated to assess species diversity and composition among the samples for each substrate each season. Data were analyzed using the statistical package Quattro Pro Version 8.

RESULTS AND DISCUSSION

Fish associated with limestone gravel and oyster shell substrates for summer and winter samples are listed in Tables 1 and 2. Because the present data are concerned with a limited number of samples that represent a small portion of the uncompleted total study, results are presented with minimal interpretation and discussion.

For some species, observed differences in numbers of individuals and size between summer and winter collections appeared to be related to recruitment, and differences in size between substrates to niche availability. Most of the fish occurring in samples are crevice-dwelling and have been identified as oyster reef associates (White and Wilson 1996).

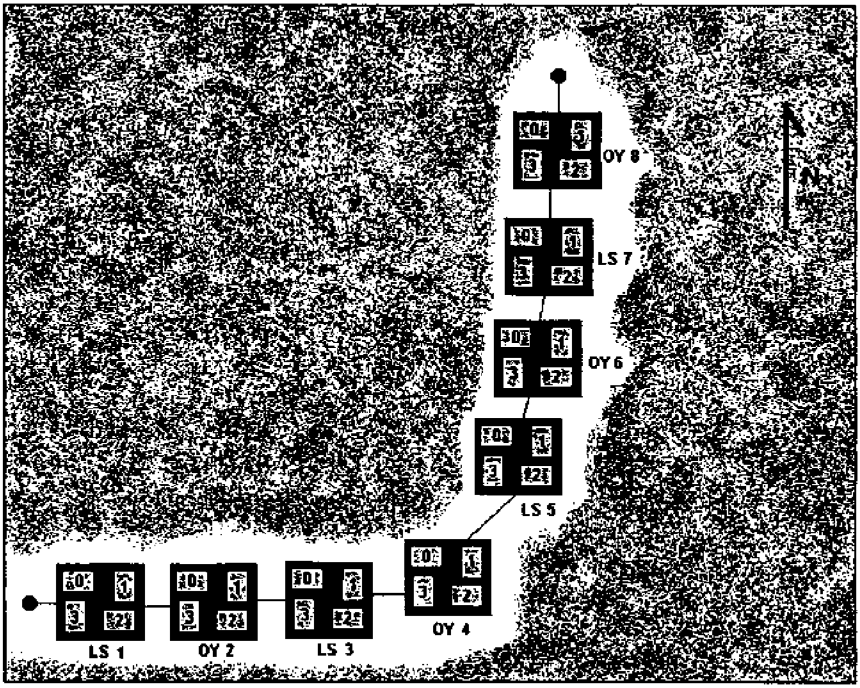


Figure 2. Sampler array

The most abundant fish species found in the samplers was *Gobiosoma bosc*. In summer samples there were significantly higher numbers of *G. bosc* in the limestone than in the oyster shell (Table 1). This was not the case with winter samples due to both low numbers and high variances (Table 2). Numbers of *G. bosc* were significantly higher in the summer samples for both limestone and oyster shell. No size differences were found between substrates in either season (Tables 1 and 2). Gobies were significantly larger in winter limestone samples than summer samples; no seasonal difference in size occurred for oyster shell samples (Table 3). Other gobiid representatives from the winter samples were *G. robustum* and *Bathygobius soporator* with one and two individuals, respectively. *Gobiosoma bosc* is more commonly found on oyster reefs whereas *G. robustum* prefers grassbeds (Hoese and Moore 1998). *Bathygobius soporator* is usually associated with rocky areas and is uncommon in Mississippi Sound (Dawson 1969).

Table 1. Vertebrates associated with gravel and oyster shell substrates during the summer months (N.S.D. = no significant difference, LS = limestone, OY = oyster).

Species	Common Name	Total #		*t*-test for Mean #		Mean Length (mm)		*t*-test for Length	
		LS	OY	*t*	P	LS	OY	*t*	P
Gobiesox strumosus	Skilletfish	25	79	6.57	0.001	28.9 ± 8.5	40.1 ± 8.9	5.57	0.000
Gobiosoma bosc	Naked goby	154	38	4.35	0.022	19.2 ±7.8	21.4 ± 7.4	N.S.D.	
Hypsoblennius ionthas	Freckled blenny	14	38	4.08	0.007	31.8 ± 7.8	50.6 ± 14.2	4.68	0.000
Opsanus beta	Gulf toadfish	8	26	4.70	0.003	17.0 ± 2.8	56.7 ± 47.7	4.22	0.000
Lutjanus griseus	Grey snapper	2	0	N.S.D.		23.5 ± 4.5	-	-	
Myrophis punctatus	Speckled worm eel	11	0	3.67	0.010	89.8 ± 8.5	-	-	
Hypleurochilus geminatus	Crested blenny	0	1	N.S.D.		-	32.4	-	
Hypsoblennius hentz	Feathered blenny	2	0	N.S.D.		16.9 ± 7.1	-	-	
Labrisomus nuchipinnis	Hairy blenny	1	0	N.S.D.		23.1	-	-	
Scartella cristata	Molly Miller	0	1	N.S.D.		-	15.1	-	

Table 2. Vertebrates associated with gravel and oyster shell substrates during the winter months (N.S.D. = no significant difference, LS = limestone, OY = oyster).

Species	Common Name	Total #		*t*-test for Mean #		Average Length (mm)		*t*-test for Len	
		LS	OY	*t*	P	LS	OY	*t*	P
Gobiesox strumosus	Skilletfish	1	23	8.09	0.015	27.3	49.4 ± 11.1	-	
Gobiosoma bosc	Naked goby	16	5	N.S.D.		29.2 ± 13.1	27.0 ± 18.1	N.S.D.	
Hypsoblennius ionthas	Freckled blenny	0	18	12.42	0.000	-	55.0 ± 13.1	-	
Opsanus beta	Gulf toadfish	0	4	N.S.D.		-	47.9 ± 59.8	-	
Bathygobius soporator	Frillfin goby	2	0	N.S.D.		29.5 ± 6.3	-	-	
Chasmodes bosquianus	Striped blenny	0	1	N.S.D.		-	66.7	-	
Gobiosoma robustum	Code goby	1	0	N.S.D.		31.2	-	-	

Table 3. Seasonal comparison of numerical abundance and size for *G. Bosc, H. Ionthas*, and *G. strumosus*. (N.S.D. = no significant difference, LS = limestone, OY = oyster).

Species	Substrate	Mean Number		*t*-test for Mean		Mean Length (mm)		*t*-test for Mean	
		Summer	Winter	*t*	P	Summer	Winter	*t*	P
Gobiosoma bosc	LS	39 ± 13	4 ± 3	5.02	0.015	19.2 ± 7.8	29.2 ± 13.1	8.16	0.000
	OY	10 ± 1	2 ± 2	7.38	0.000	21.4 ± 7.4	27.0 ± 18.1	N.S.D.	
Hypsoblennius ionthas	LS	4 ± 2	0	4.04	0.027	31.8 ± 7.8	-	-	
	OY	10 ± 6	6 ± 1	N.S.D.		50.6 ± 14.2	55.0 ± 13.1	N.S.D.	
Gobiesox strumosus	LS	6 ± 3	1 ± 1	3.78	0.009	28.9 ± 8.5	27.3	-	
	OY	19 ± 2	8 ± 2	6.81	0.001	40.1 ± 8.9	49.4 ± 11.1	3.69	0.001
Opsanus beta	LS	2 ± 1	0	N.S.D.		17.0 ± 2.8	-	-	
	OY	7 ± 1	1 ± 2	3.49	0.040	56.7 ± 47.7	47.9 ± 59.8	N.S.D.	

Hypsoblennius ionthas was the most common blenny associated with the substrate samplers. Blennies were significantly more abundant and larger on the summer oyster samples than on limestone (Table 1). Higher percentages of small blennies were found in the summer limestone samples whereas larger blennies preferred the summer oyster shell samplers (Figure 3). The blennies from the winter oyster samples were not significantly larger than those in the summer oyster samples (Table 3). No blennies were found on the winter limestone samples (Table 2). Other blenniids collected included: *Hypsoblennius hentz*, *Hypleurochilus geminatus*, *Labrisomus nuchipinnis*, and *Scartella cristata* in the summer, and *Chasmodes bosquianus* in the winter. *Hypsoblennius ionthas* prefers hard bottom substrates, whereas *H. hentz* resides on softer muddy bottoms, and may explain the higher numbers of *H. ionthas* found in the samples compared to other blenniids.

Gobiesox strumosus, skilletfish, was significantly more abundant and larger on oyster shell samplers during both summer and winter (Tables 1 and 2). Summer samples had significantly higher numbers of *G. strumosus* than did winter samples for both substrates (Table 3). In the winter limestone samples, skilletfish were significantly larger than those found in the summer. Skilletfish live on oyster reefs and lay their eggs in empty oyster shells (Runyan 1961). Size distribution of *G. strumosus* from summer samples indicates that smaller individuals utilize the smaller spaces in the limestone samplers while larger animals inhabit the larger spaces provided by the oyster shells (Figure 3). The size distribution from the summer oyster samplers shifted for *G. strumosus* in the winter; no smaller animals were found in any of the winter samples indicating growth and lack of recruitment during this time of year.

Opsanus beta was significantly more numerous and larger in oyster shell than limestone during summer sampling (Table 1). No animals occurred in limestone collectors during the winter (Table 2), and significantly fewer animals were found in winter oyster shell samples than summer samples (Table 3). Large toadfish utilized the larger niches of the oyster samplers and smaller specimens were found in both substrates. The toadfish, or "oyster dog," is a common inhabitant of oyster reefs and rocky jetties (Hoese and Moore 1998).

The speckled worm eel, *Myrophis punctatus*, was only present in summer limestone samplers (Table 1). These eels like to burrow and the small spaces in the limestone samplers offer protection. Small recruiting grey snapper, *Lutjanus griseus*, were only found in summer limestone samplers (Table 1).

The four most dominant species overall were *G. bosc*, *G. strumosus*, *H. ionthas*, and *O. beta*. The Brillouin diversity index (*H*, base 2 scale) was calculated to assess species diversity for each substrate each season. As the species diversity increases the index increases and does not usually exceed 5.0 in biological samples (Krebs 1989). Values were as follows: limestone summer, 1.227; oyster summer, 1.871; limestone winter, 0.791, and oyster winter, 1.597. These indices showed that species diversity was higher in oyster samples both seasons, and higher in summer than winter for both substrates.

Spearman's rank correlation was used to assess the similarity in species composition between the two substrates was. The species composition between the two summer substrates was significantly correlated ($r_S = 0.59$, $p < 0.05$), however, winter samples were not significantly correlated ($r_S = -0.14$, $p > 0.05$). The extremely small number of animals collected in the winter samples may have contributed to the lower correlation.

CONCLUSIONS

For dominant fish species, observed differences in numbers of individuals and size between seasons is related to recruitment. Post-larval and juvenile fish suffer from high mortality rates due to predation. Winter samples had lower numbers but larger animals as the surviving recruits grew. Differences in species composition or size of individuals between substrates were affected by niche availability, the larger the holes the larger the animals that will inhabit them.

Present data represent the first stage of a much larger project. Completion of the seasonal sampling regime will provide a more comprehensive data set from which to evaluate the effect of season and substrate on fish assemblages associated with low profile reefs in estuarine waters. Concurrent studies include a fishery independent finfish sampling program near reef areas and an analysis of the stomach contents of selected fish species in those collections to determine if they are foraging on the reef inhabitants.

ACKNOWLEDGMENTS

The Mississippi Department of Marine Resources provided funding for this study through their Tidelands Trust Program. We thank the Gulf Coast Research Laboratory Fisheries personnel, Bradley Randall, Jude LeDoux, Wes Devers, and John Anderson, for deployment and retrieval of collectors. Lisa Engel (GCRL) and Michael Buchanan along with additional personnel of the Mississippi Department of Marine Resources assisted in sample wash down. Jamie McFerrin, Danielle Slade, and Virginia Shervette helped to separate samples. We acknowledge Joanne Lyczkowski-Schulz of the National Marine Fisheries Service for sharing her photographic equipment and expertise with us. Windsor Aguirre gave up his free time to take our animal photographs.

LITERATURE CITED

Dawson, C. E. 1969. Studies on the gobies of Mississippi Sound and adjacent waters II: an illustrated key to the gobioid fishes. Publ. Gulf Coast Research Laboratory Museum. 59pp.

Hoese, H. D., R. H. Moore. 1998. Fishes of the Gulf of Mexico (2nd edition). Texas A&M University Press: College Station. 422 pp.

Krebs, C.J. 1989. *Ecological Methodology*. Harper Collins Publisher, New York. 654 pp.

Runyan, S. 1961. Early development of the clingfish, *Gobiesox strumosus* Cope. Chesapeake Sci. **2**:113-140.

White, M.E. and E.A. Wilson. 1996. Predators, Pests, and Competitors. Pages 559-579 in: V.S. Kennedy, R.I.E. Newell, and A.F. Eble, (eds.) *The Eastern Oyster, Crassostrea virginica*. Maryland Sea Grant Book, MD.

Seasonal Colonization of Low Profile Artificial Reefs in Mississippi Coastal Waters: Invertebrates

HARRIET M. PERRY, KIRSTEN LARSEN,
JAMES WARREN, and CHRISTINE TRIGG
The University of Southern Mississippi,
Institute of Marine Sciences
Gulf Coast Research Laboratory,
P.O. 7000
Ocean Springs, Mississippi 39566-7000 USA

ABSTRACT

Artificial reefs serve as fish attractants and may increase production of some species by increasing habitat. In an effort to enhance already established recreational fisheries and to increase numbers of and access to less common structure-associated fishes, Mississippi began building new low profile artificial reefs and augmenting existing ones. Over twenty inshore low profile oyster shell reefs were developed prior to 1995. Subsequent reef development has utilized limestone gravel in conjunction with oyster shell or limestone alone. The creation of limestone gravel and/or oyster shell reefs provided an opportunity to obtain information on the colonization of these different substrates by benthic invertebrates. Colonization studies were conducted using crates of artificial substrate placed on a gravel/oyster shell reef located approximately 300 meters offshore in central Mississippi Sound. Crates contained crushed limestone gravel or oyster shells. Crates were pulled every three months and all organisms were removed. Invertebrates were identified to the lowest taxonomic level, measured to the nearest 0.1 mm and weighed to the nearest 0.001 g. Dominant invertebrate taxa included representatives of the following groups: Xanthidae (*Menippe adina, Eurypanopeus depressus, Panopeus simpsoni*), Porcellanidae (*Petrolisthes armatus*), Alpheidae (*Alpheus angulatus, A. heterochaelis*), Palaemonidae (*Palaemonetes vulgaris*), Amphipoda (*Melita, Apocorophium*), Polychaeta (*Neanthes succinea*), Bivalvia (*Ischadium recurvum*), and Gastropoda (*Stramonita haemastoma*). Observed differences in numbers of individuals and size between summer and winter collections appeared to be related to recruitment. Data are presented on species abundance and size by substrate type and season.

KEY WORDS: Estuarine, invertebrate assemblages, low profile artificial reef

INTRODUCTION

Artificial reefs serve as fish attractants and may increase production of some species by increasing habitat. In an effort to enhance already established recreational fisheries and to increase numbers of and access to less common structure-associated fishes, Mississippi began building new low profile artificial reefs and augmenting existing ones. Although over twenty low profile artificial reefs (oyster shell, concrete rubble, limestone gravel) have been constructed in Mississippi inshore waters, there are no data on reef community structure or the association of fish populations with these reefs. A study of the faunal assemblages associated with reef colonization in Mississippi Sound was begun in December 1998 as part of a larger program to assess productivity of these reefs in relation to recreational fishing opportunities. The research reported on herein is part of a long-term study addressing seasonal colonization/succession of fauna associated with limestone and oyster shell reefs in estuarine waters of Mississippi Sound. Data presented represent initial colonization studies conducted in the summer of 1998 and the winter of 1998/99.

MATERIALS AND METHODS

Colonization was studied by placing a series of crates filled with 0.025m^3 of limestone gravel or oyster shell on a newly created, nearshore limestone/shell reef. The area adjacent the artificial reef is characterized by small patch subtidal oyster beds. Average water depth is approximately 1.5 m. Site location is shown in Figure 1. Thirty-two samplers (16 limestone, 16 oyster shell) were deployed in the summer of 1998. The crates were placed on eight plastic pallets, four trays to a pallet (Figure 2). Four pallets contained crates filled with limestone and four pallets contained crates with oyster shell. All crates and pallets were labeled. A sampling schedule was established that removed one crate from each pallet (four limestone gravel and four oyster shell) after an initial soak time of three months. At six, nine, and twelve month intervals the remaining crates were pulled (four limestone gravel and four oyster crates per sampling period). Additionally, a set of crates was replaced (four limestone gravel and four oyster) each sampling period so that a three-month set of samples was obtained for each season. Original sampling schedule called for removal of a set of three month samples and a set of three and six month samples during summer and fall of 1998, respectively. Summer samples were collected; however, Hurricane Georges in September 1998 destroyed the pallets and no fall samples were collected. Samplers were re-deployed in December 1998. Data presented in this study are from the three month summer samples (original study) and the three month winter samples that began the new sampling regime.

Crates were removed from the water, immediately placed in seawater soaked oyster sacks and returned to the Gulf Coast Research Laboratory for processing. Contents of each crate were washed over screening (winter samples) and all organisms collected. Samples were frozen prior to analysis. Each sample was sorted to species and the total number and weight recorded. When available, fifty individuals of each species were randomly selected for measurement. Individuals were measured to the nearest 0.1 mm in length or width (brachyuran crabs) using digital calipers. A Sartorius analytical balance was used to measure weight to the nearest 0.001 g.

Kruskal-Wallace ANOVA was used to test for significant differences in species abundance and size between seasons by substrate. Student's *t*-test was used to compare species abundance and size between substrates within a season. The level of significance for all tests was set at α=0.05. Data were analyzed using the statistical packages SPSSx and Quattro Pro Version 7.

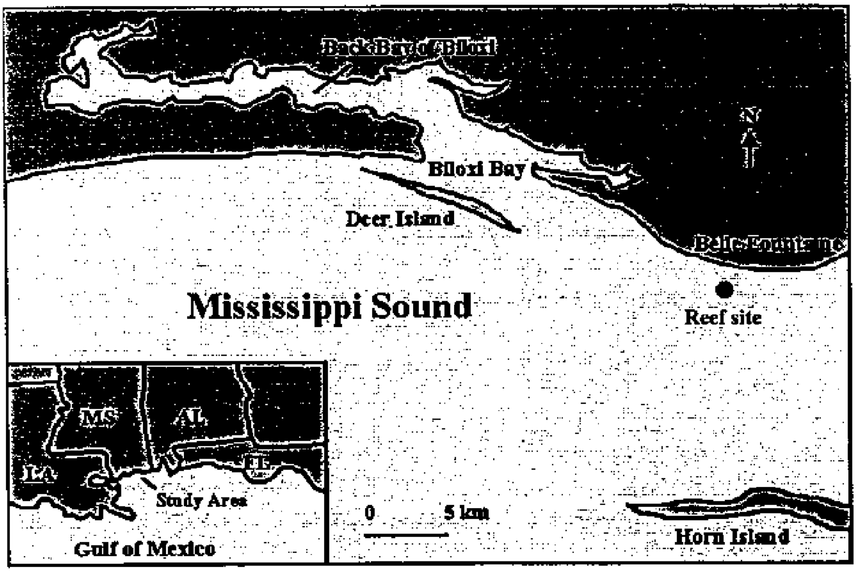


Figure 1. Location of low profile artificial reef.

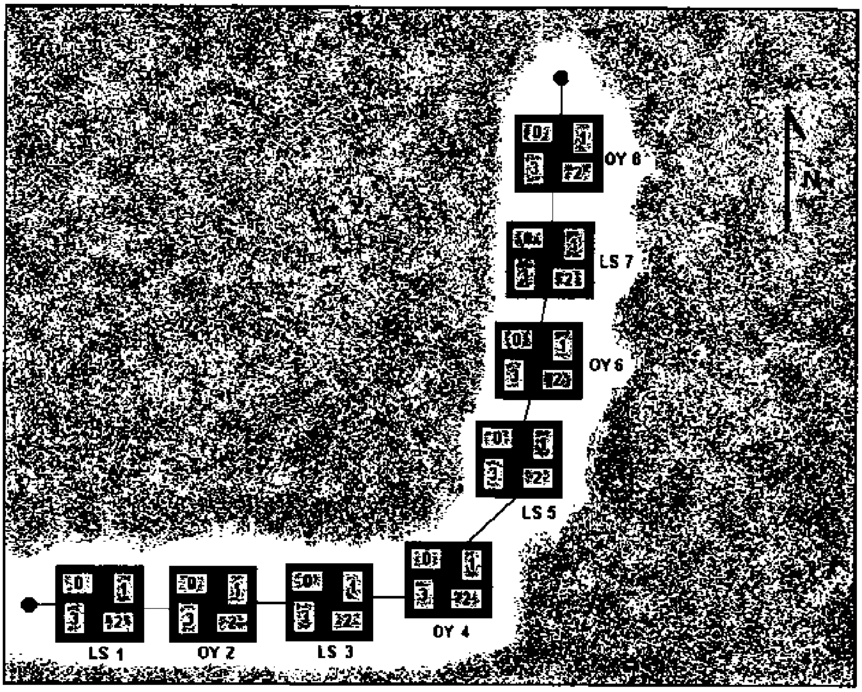


Figure 2. Sampler array

RESULTS AND DISCUSSION

Invertebrates associated with limestone gravel and oyster shell substrates for summer and winter samples are listed in Tables 1 and 2. Limestone gravel is hereafter referred to as gravel and oyster shell as shell. Because the present data are concerned with a limited number of samples that represent a small portion of the uncompleted total study, results are presented with minimal interpretation and discussion. For many species, observed differences in numbers of individuals and size between summer and winter collections appeared to be related to recruitment. Most of the species occurring in samples have been identified as oyster reef associates in Mississippi Sound (Heard 1979). The three xanthid crab species are known oyster predators and *Neanthes succinea* is one of the most commonly encountered nereids in oyster reef communities. *Melita nitida, Ischadium recurvum, Palaemonetes vulgaris, Alpheus angulatus*, and *A. heterochaelis* are known from oyster reefs in the area.

Table 1. Invertebrates associated with gravel and oyster shell substrates during the summer months (N.S.D. = no significant difference, LS = limestone, OY = oyster).

Species	Common Name	Total Number		*t*-test for Mean Number		Mean Size		*t*-test for Mean Size	
		LS	OY	*t*	P	LS	OY	*t*	P
Eurypanopeus depressus	Mud crab	5096	3376	N.S.D.		4.9 ± 1.3 mm	5.2 ± 2.6 mm	N.S.D.	
Neanthes succinea	Worm	2608	1146	3.89	0.008	0.029 ± 0.002 g	0.008 ± 0.005 g	7.14	0.002
Palaemonetes vulgaris	Grass shrimp	215	844	4.56	0.004	16.2 ± 4.5 mm	15.6 ± 4.4 mm	N.S.D.	
Alpheus angulatus	Snapping shrimp	442	179	7.55	0.000	14.0 ± 3.6 mm	15.9 ± 7.0 mm	3.14	0.002
Menippe adina	Stone crab	221	305	N.S.D.		7.9 ± 6.0 mm	8.5 ± 8.5 mm	N.S.D.	
Ischadium recurvum	Hooked mussel	127	155	N.S.D.		7.4 ± 2.5 mm	9.1 ± 4.1 mm	4.31	0.000
Stramonita haemastoma	Oyster drill	12	27	3.38	0.015	22.8 ± 25.4 mm	45.9 ± 31.2 mm	2.25	0.031
Turbellaria	Flatworms	30	14	N.S.D.		-	-	-	
Petrolisthes armatus	Porcelain crab	11	34	N.S.D.		-	-	-	
Panopeus simpsoni	Mud crab	10	3	2.18	0.072	16.8 ± 4.8 mm	26.0 ± 2.7 mm	2.45	0.006
Clibanarius vittatus	Striped hermit crab	3	2	N.S.D.		16.3 ± 8.6 mm	13.9 ± 5.1 mm	N.S.D.	
Nassarius vibex	Southern nassa	2	0	N.S.D.		10.6 mm	-	-	

Table 2. Invertebrates associated with gravel and oyster shell substrates during the winter months (N.S.D. = no significant difference, LS = limestone, OY = oyster).

Species	Common Name	Total Number		*t*-test for Mean #		Mean Size		*t*-test for Mean Size	
		LS	OY	*t*	P	LS	OY	*t*	P
Eurypanopeus depressus	Flat mud crab	65	42	N.S.D.		9.4 ± 3.2 mm	11.4 ± 3.5 mm	2.88	0.005
Neanthes succinea	Worm	1,382	562	N.S.D.		0.021 ± 0.010 g	0.018 ± 0.015 g	N.S.D.	
Palaemonetes vulgaris	Grass shrimp	1	9	4.37	0.022	24.5 mm	21.6 ± 2.7 mm	N.S.D.	
Alpheus heterochaelis	Snapping shrimp	1	0	N.S.D.		23.7 mm	-	-	
Menippe adina	Stone crab	1	0	N.S.D.		19.7 mm	-	-	
Ischadium recurvum	Hooked mussel	0	4	N.S.D.		-	6.8 ± 4.3 mm	-	
Panopeus simpsoni	Mud crab	3	3	N.S.D.		19.2 ± 3.5 mm	25.4 ± 2.4 mm	N.S.D.	
Callinectes sapidus	Blue crab	14	2	N.S.D.		7.6 ± 2.6 mm	34.2 ± 9.9 mm	10.31	0.000
Callinectes similis	Lesser blue crab	2	0	N.S.D.		4.8 ± 0.4 mm	-	-	
Melita nitida	Amphipod	10,947	3170	N.S.D.		-	-	-	
Melita longisetosa	Amphipod	1,101	447	N.S.D.		-	-	-	
Apocorophium louisianum	Amphipod	204	323	N.S.D.		-	-	-	
Monocorophium acherusicum	Amphipod	0	9	N.S.D.		-	-	-	

Crustaceans

Amphipods were the most abundant macro-crustaceans associated with the reef materials. Because the size of these organisms allowed for escapement through the small holes in the crates during sampler retrieval and use of netting to retain these organisms wasn't begun until the winter samples, numerical estimates of relative abundance are available only for the latter collections. Amphipods were identified using the keys of Sheridan (1979) and Bousfield and Hoover (1997).

Two genera of free-living gammarideans dominated the amphipod fauna on both gravel and shell substrates: *Melita* and *Apocorophium* (Table 2). Species of both genera are associated with oyster reefs and shell material in Mississippi waters (Heard 1979). *Melita nitida* was extremely abundant; *Melita longisetosa* and *Apocorophium louisianum* (= *Corophium louisianum*) were common in samples. There was no significant difference in species composition between the two substrates in winter samples. *Monocorophium acherusicum (=Corophium acherusicum)* was found only in oyster shell material and in limited numbers.

Eurypanopeus depressus was the most the most abundant brachyuran crab collected. This species is associated with oyster reefs throughout much of its range and it is often present in large numbers (Ryan 1956, Tabb and Manning 1961, Rouse 1970). It is a known predator of oyster spat (McDermott 1960). Ovigerous females were numerous in summer samples in both substrate types. Thirty-one percent of the females collected in gravel and 25% of the females taken in shell were egg-bearing. Size of ovigerous females ranged from 3.8 to 13.4 mm in carapace width (CW). Width frequency distributions for summer and winter samples by substrate are shown in Figure 3. Recruitment was evident in summer samples and newly recruited crabs were abundant on both substrates. There was no significant difference in abundance or size of crabs in either substrate (Table 1). As in summer samples, winter densities in the two substrates were not significantly different; however, there was a significant difference in size of crabs occupying gravel and shell during the winter with mean carapace width greater in shell (Table 2). Comparing seasons, there were significant differences in crab densities and size between summer and winter in both gravel and shell with crabs more numerous in summer samples on both substrates (Tables 3 and 4). Crabs were significantly larger on both substrates in winter samples. Increase in size in winter samples is, in part, related to decreased recruitment and growth of summer recruits. Variables responsible for reduction in crab densities in winter samples may include emigration in response to environmental conditions and/or mortalities related to predation and competition for food. Greater abundance of crabs in oyster substrate in winter may be related to their increased size and larger niche availability in shell habitats.

Table 3. Seasonal comparison of numerical abundance and size for *E. depressus* and *N. succinea* (N.S.D. = no significant difference, LS = limestone, OY = oyster).

Species	Substrate	Mean Number		ANOVA for Mean Number	
		Summer	Winter	χ^2	P
Eurypanopeus depressus	LS	1274 ± 413	16 ± 10	5.333	0.021
	OY	844 ± 321	14 ± 1	4.500	0.034
Neanthes succinea	LS	652 ± 123	346 ± 222	N.S.D.	
	OY	286 ± 142	187 ± 227	N.S.D.	

Table 4. Seasonal comparison of mean size for *E. depressus* and *N. succinea* (N.S.D. = no significant difference, LS = limestone, OY = oyster).

Species	Substrate	Mean Size		ANOVA for Mean Size	
		Summer	Winter	χ^2	P
Eurypanopeus depressus	LS	4.9 ± 1.3 mm	9.4 ± 3.2 mm	92.02	0.000
	OY	5.2 ± 2.6 mm	11.4 ± 3.5 mm	70.48	0.000
Neanthes succinea	LS	0.029 ± 0.002 g	0.021 ± 0.010 g	N.S.D.	
	OY	0.008 ± 0.005 g	0.018 ± 0.015 g	N.S.D.	

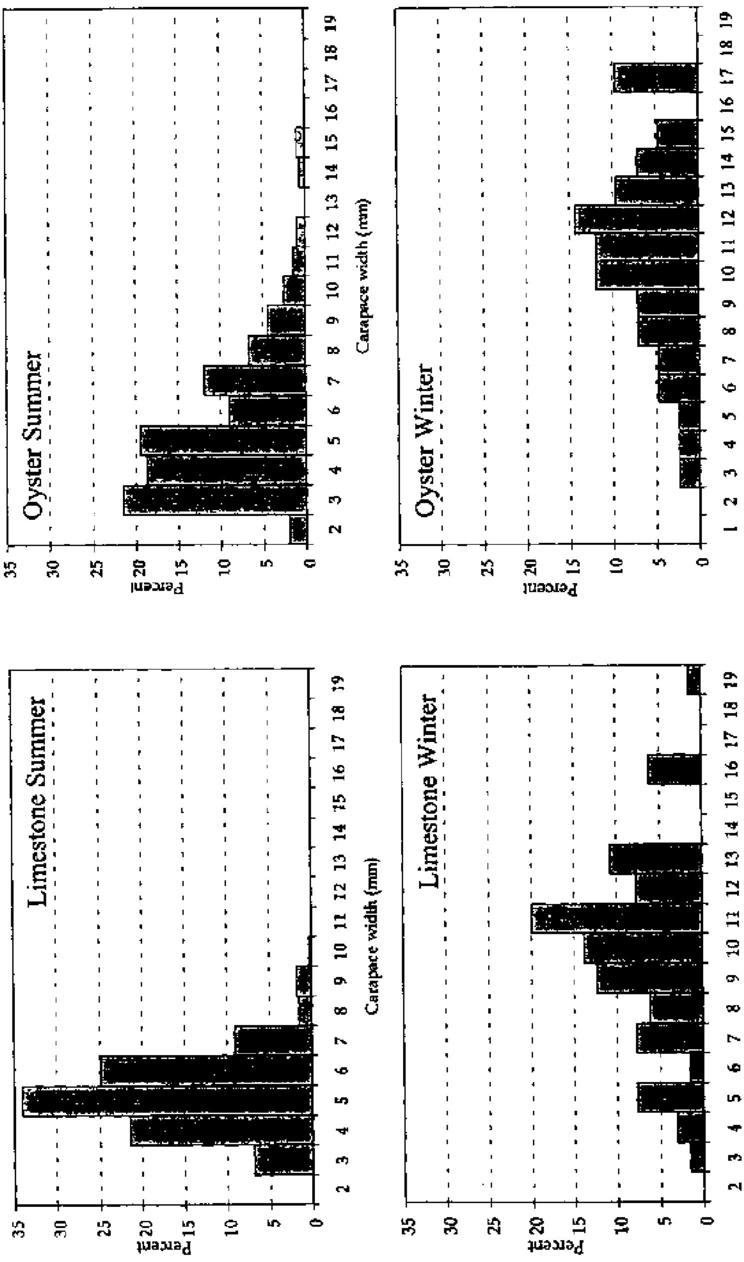


Figure 3. Width frequency distributions of *Eurypanopeus depressus*

Stone crabs, *Menippe adina*, were common only in summer samples; a single specimen was identified from winter collections. Stone crabs were numerous in both substrate types and there was no significant difference in abundance or size between gravel and shell (Table 1). The majority of the crabs collected were under 10.0 mm CW (Figure 4). Stone crabs spawn in the summer and recruits and early juveniles are common on oyster reefs and stone rubble breakwaters in Mississippi Sound (Stuck and Perry 1992). Lack of stone crabs in winter samples may reflect seasonal emigration from shallow reef areas and/or increased predation associated with decreased niche availability for larger juveniles. Size-specific shelter limitation has been implicated as a factor in determining population levels of large juveniles in northern GOM estuaries (Beck 1995, 1997).

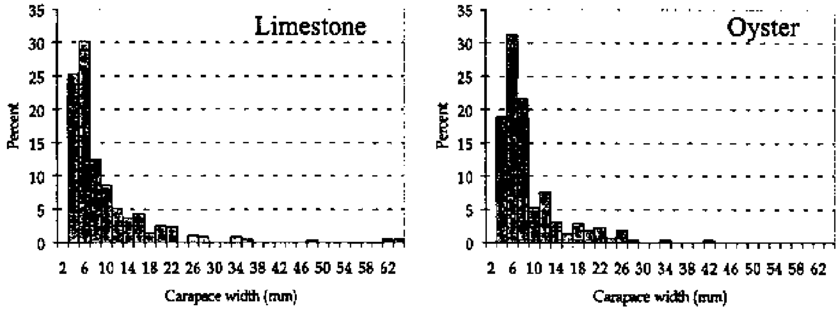


Figure 4. Width frequency distributions of *Menippe adina* in summer samples

Other crab species occurring in samples included Simpson's mud crab, *Panopeus simpsoni*, the porcellanid crab, *Petrolisthes armatus* and the striped hermit crab, *Clibanarius vittatus*. Porcelain crabs and the striped hermit were collected only in summer samples and there was no significant difference in density between the two substrate types (Table 1). *Petrolisthes armatus* is a filter feeder but this species will also scrape algae from rock and shell (Fotheringham and Brunenmeister 1989). The striped hermit is an omnivorous scavenger common in tidal marsh habitats, on rock jetties, and around mud flats (Heard 1979). Adults often utilize shells of the oyster drill. Simpson's mud crabs were collected in both seasons in small numbers. In summer samples, they were significantly more abundant in gravel, but were significantly larger in shell (Table 1). In winter collections, there was no significant difference in numbers or size between the substrate types, but larger individuals were associated with shell (Table 2). *Panopeus simpsoni* is an oyster associate that occurs subtidally and intertidally in Mississippi Sound and it often co-occurs with *E. depressus* (Heard 1979).

Callinectes sapidus, the blue crab, and *C. similis*, the lesser blue crab, were found only in winter samples in small numbers. There was no significant difference in the numbers of blue crabs between the two substrates, but crabs in shell samples were significantly larger (Table 2). Blue crabs in winter gravel samples were summer/fall recruits (mean size was <10.0 mm CW) that move to deeper waters with decreasing water temperatures (Steele and Perry 1990). Peak settlement of blue crab megalopae occurs in the late summer/early fall (Perry *et al.* 1995, Johnson and Perry 1999). Both megalopae and early crab stages are associated with shallow nearshore waters.

The snapping shrimp, *Alpheus angulatus*, was found only in the summer. Densities were significantly higher in gravel, but size was significantly larger in shell (Table 1). A single specimen of *Alpheus heterochaelis*, occurred in winter samples in gravel (Table 2). Both species of *Alpheus* are associated with oyster reef communities in the northern GOM (McClure 1995). The grass shrimp, *Palaemonetes vulgaris*, was common in summer samples. There were significant differences in summer catch of grass shrimp between substrates, but no significant difference in size (Table 1). Grass shrimp were more abundant in shell than in gravel. Heard (1979) noted that this species of grass shrimp was often associated with fouling communities and shell bottoms and may occur with *P. pugio* around oyster reefs.

Polychaetes/platyhelminths

The polychaete, *Neanthes succinea*, was the only other species abundant enough during both sampling periods to allow for comparative testing between seasonality and abundance and size and to test for seasonal differences in abundance and size between substrates. There were no significant differences in polychaete densities or size between winter and summer samples for either substrate type (Tables 3 and 4). In comparing gravel and shell substrates in summer samples, there was a significant difference in density and size of polychaetes. Polychaetes were more abundant in gravel and were larger (Table 1). In winter samples there was no significant difference in density or size (Table 2). Flatworms (turbellarians) occurred in summer samples with no significant difference in numbers between the two substrate types.

Molluscs

Molluscs associated with samples included the hooked mussel, *Ischadium recurvum*, the oyster drill, *Stramonita haemastoma*, and the southern nassa, *Nassarius vibex*. Only hooked mussels occurred in both summer and winter samples. Mussels in summer samples occurred in both substrates with larger individuals found in shell samples (Table 1). Small size of specimens in summer samples and much reduced numbers of individuals in winter suggests recruitment and settlement occurs in the warmer months. Oyster drills were significantly more abundant and larger in shell samples than in gravel (Table 1).

CONCLUSIONS

High abundance indices for many of the invertebrates appearing in summer samples were related to recruitment. Reduced numbers of individuals in winter samples occurred as organisms migrated from the reef area with falling water temperatures in the winter or were subject to increased predation rates associated with a larger suite of predators in warmer months. Early juvenile stages of brachyuran crabs suffer high mortality rates with both intra- and inter-specific predation influential in controlling population size. Present data represent a small portion of the overall study. Completion of the seasonal sampling regime will provide a more comprehensive data set from which to evaluate the effect of season and substrate on invertebrate assemblages associated with low profile reefs in estuarine waters. This study also provides the baseline data on species composition and abundance necessary to evaluate utilization of these reefs by recreational fish species. Concurrent studies include a fishery independent finfish sampling program near reef areas and an analysis of the stomach contents of selected fish species in those collections. Creel surveys are also ongoing.

ACKNOWLEDGMENTS

The Mississippi Department of Marine Resources provided funding for this study through their Tidelands Trust Program. We are indebted to the many talented invertebrate taxonomists at the Gulf Coast Research Laboratory. Sara LeCroy, Richard Heard, and Jerry McLelland provided patient and invaluable assistance in teaching us to differentiate the amphipod and polychaete species taken in samples. Bradley Randall, Wes Devers, Jude LeDoux, and John Anderson assisted with field retrieval of collectors. Lisa Engel (GCRL) and Michael Buchanan along with additional personnel of the Mississippi Department of Marine Resources assisted in sample wash down. Jamie McFerrin, Danielle Slade, and Virginia Shervette helped separate samples. We acknowledge Joanne Lyczkowski-Schulz of the National Marine Fisheries Service for sharing her photographic equipment and expertise with us and Windsor Aguirre for photographs.

LITERATURE CITED

Beck, M.W. 1995. Size-specific shelter-limitation in stone crabs: a test of the demographic bottleneck hypothesis. *Ecology* **76**(3):968-980.

Beck, M.W. 1997. A test of generality of the effect of shelter bottlenecks in four stone crab populations. *Ecology* **78**(8):2487-2503.

Bousfield, E.L. and P.M. Hoover. 1997. The amphipod superfamily Corophioidea on the Pacific Coast of North America. Part V. Family Corophiidae. Corophiinae, new subfamily. Systematics and distributional ecology. *Amphipacifica* **2**(3):67-139.

Fotheringham, N. and S. Brunenmeister. 1989. *Guide to Gulf Coast Marine Life*. Gulf Publishing Company, Houston, Texas. 142 pp.

Heard, R. 1979. Guide to common tidal marsh invertebrates of the northeastern Gulf of Mexico. Mississippi/Alabama Sea Grant Consortium, MASGP-79-004:1-82.

Johnson, D. R. and H.M. Perry. 1999. Blue crab larval dispersion and retention in the Mississippi Bight. *Bull. Mar. Sci.* **65**(1):129-149.

McClure, M.R. 1995. ***Alpheus angulatus***, a new species of snapping shrimp from the Gulf of Mexico and northwestern Atlantic, with a redescription of *Alpheus heterochaelis* Say, 1818 (Decapoda: Caridea: Alpheidae). *Proc. Biol. Soc. Wash.* **108**(1):84-97.

McDermott, J.J. 1960. The predation of oysters and barnacles by crabs of the family Xanthidae. *Proc. Penn. Acad. Sci.* **34**:199-211.

Perry, H.M. and K.C. Stuck. 1982. The life history of the blue crab in Mississippi with notes on larval distribution. Pages 17-22 in: H.M. Perry and W.A. Van Engel (eds.), *Proceedings of the Blue Crab Colloquium*, Publication 7, Gulf States Marine Fisheries Commission, Ocean Springs, Mississippi.

Perry, H.M., C.K. Eleuterius, C.B. Trigg, and J.R. Warren. 1995. Settlement patterns of *Callinectes sapidus* megalopae in Mississippi Sound: 1991, 1992. *Bull. Mar. Sci.* **57**(3):821-833.

Perry, H.M., J. Warren, C. Trigg and T. Van Devender. 1998. The blue crab fishery of Mississippi. *J. Shellfish Res.* **17**(2):425-433.

Rouse, W.L. 1970. Littoral Crustacea from southwest Florida. *Quart. Jour. Fla. Acad. Sci.* **32**(2):127-152.

Ryan, E.P. 1956. Observations on the life histories and the distribution of the Xanthidae (mud crabs) of Chesapeake Bay. *Amer. Midl. Natur.* **56**:138-162.

Sheridan, P.F. 1979. Three new species of *Melita* (Crustacea: Amphipoda), with notes on the amphipod fauna of the Apalachicola estuary of Northwest Florida. *Northeast Gulf Sci.* **3**(2):60-73.

Steele, P. and H. M. Perry (editors). 1990. The blue crab fishery of the Gulf of Mexico United States: A regional management plan. Publication Number 21. Gulf States Marine Fisheries Commission, Ocean Springs, Mississippi.

Stuck, K. and H.M. Perry. 1992. Life history characteristics of *Menippe adina* in Mississippi coastal waters. Pages 82-98 in: T.M. Bert (ed.), *Proceedings of a Symposium on Stone Crab (Genus Menippe) Biology and Fisheries*. Fla. Mar. Res. Inst. Publ. 50.

Tabb, D.C. and R.B. Manning. 1961. A checklist of the flora and fauna of northern Florida Bay and adjacent brackish waters of the Florida mainland collected during the period July, 1957 through September, 1960. ***Bull. Mar. Sci. Gulf Caribb.*** **1** 1(4):552-649.

The Role of Nearshore Habitats as Nursery Grounds for Juvenile Fishes on the Northeast Coast of St. Croix, USVI

IVAN MATEO and WILLIAM J. TOBIAS
USVI Division of Fish and Wildlife
Room 203 Lagoon Street Complex
St. Croix, US Vrgin Islands 00841

ABSTRACT

Three protected backreef embayments on St. Croix's northeast coast were sampled quantitatively to determine species composition and juvenile fish abundance from October 1998 to September 1999. The study sites consisted of Yellowcliff Bay, Teague Bay and Cottongarden Bay. Juvenile reef fish assemblages were monitored using three complementary sampling methods: a visual strip transect census, fish traps and beach seine net. By comparing juvenile fish communities from distinct habitats (patch reef, seagrass, rubble, algal plains, and sand) within the three embayments, significant differences in species richness and abundance of juveniles were identified. Patch reef habitats had more species diversity than any other habitat among the three embayments. Most juvenile fish observed among the sites were scarids, labrids and haemulids. The slippery dick *Halichoeres bivittatus*, the bucktooth parrotfish *Sparisoma radians*, the spotted goatfish *Pseudupuneus maculatus* and *Haemulon spp.* were among the most abundant species. Preference of nearshore habitats by economically important juvenile reef fishes, suggests a serious concern for habitat conservation

KEY WORDS: Juvenile, nursery, recruitment

INTRODUCTION

The role of coastal habitats as potential nursery grounds for coral reef fishes and many invertebrates has been widely accepted by many authors (Springer and McErlean 1962, Austin 1971, Macnae 1974, Beumer 1978). Nearshore habitats such as mangrove systems, seagrass meadows and backreef areas constitute an array of multispecies communities and their ecological relationships and connectivity is often wide-ranging and complex. Acting together, these coastal habitats provide food, habitat, and shelter functions for many organisms in their early stages (Ogden and Zieman 1977, Shulman 1984, Shulman 1985). Several investigators have identified essential factors possessed by nearshore habitats within backreef areas such as food availability and potential refuge sites in these systems, which have been recalled as determining factors of a nursery function. The interaction between the structural and functional characteristics of these components results in a heterogenic environment that provides refuge and shelter as well as food for a great number of organisms (Phillips and Menez 1988). The ecological importance of various nearshore habitats in backreef areas as nursery areas for fishes of recreational and commercial fisheries have been documented (Ogden and Zieman 1977, Zieman et al. 1982, Robblee and Zieman 1984, Rooker and Dennis 1991). Economically important species have been reported to use these habitats for nursery grounds including grunts, parrotfishes, several species of snappers and groupers, the spiny lobster, the queen conch and others (Boulon 1986, Stoner and Waite 1990, Appeldoorn et al. 1997).

Major issues of concern affecting these nearshore environs include the status of the habitats supporting the recreational and commercial fisheries, the fishes associated to a particular habitat and to what extent degradation of coastal habitats could adversely affect these nursery grounds. The present study presents information on the community structure of the fish assemblages in nearshore habitats within backreef lagoon areas on the Northeast end of St. Croix (Figure 1). This information is essential in the management of these areas to allow enhancement of their ecological and fisheries value as nursery grounds.

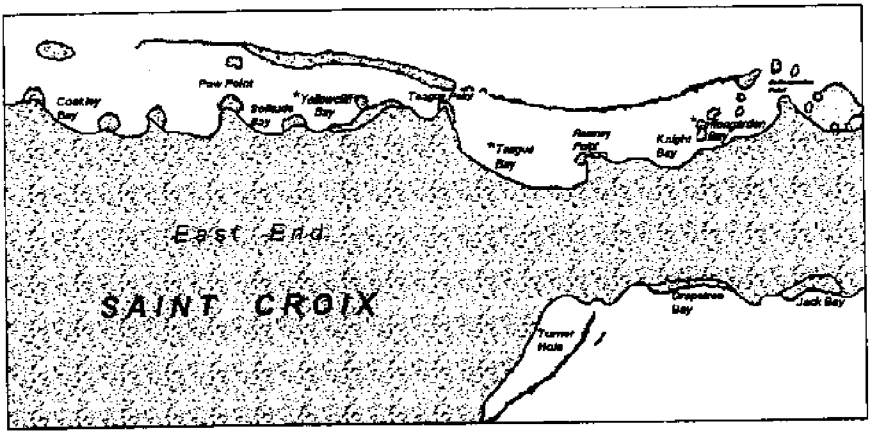


Figure 1. Location of the three *embayments studied at the Northeast Coast of St. Croix, USVI.

METHODS

The nearshore nursery habitat in three protected backreef embayments on St. Croix's northeast coast were sampled from October 1998 to September 1999 to quantitatively determine species composition and fish abundance. The study sites consisted of Cottongarden Bay, Teague Bay and Yellowcliff Bay. The distance from the shore to the reef within these bays varies from approximately 500 to 700 m. These areas have a very high propensity for habitat degradation due to the impact of human activities. Development activities include: a public beach and residential shoreline development in Cottongarden Bay, a yacht club, residential homes, restaurant and condominiums in Teague Bay; and residential development in Yellowcliff Bay.

Fish traps, visual transects and seine nets were used as assessment techniques. Ten 50 m transects were randomly established for location and direction in each of the three backreef embayments to conduct visual fish censuses. The total number of transects required to sample all habitat types was based on preliminary fish census cumulative species/transect counts. Two divers simultaneously conducted monthly fish censuses along a 2-m wide strip either side of the transect line. Habitat type, fish species, fish number and fish size range were recorded for each transect. Size categories were characterized by <5 cm, 5 - 10 cm, >10 cm. For most species juveniles less than or equal to 5 cm were recorded as recruits. A total of 10 rectangular, 92 x 57 x 19 cm, baited fish traps made from vinyl-coated 1.3 cm wire mesh, were set for 24 hours at randomly selected locations in each embayment. The traps were rebaited and relocated to the next embayment. A total of two beach seine hauls were conducted in each area on a monthly basis during the trapping period. The beach seine net measured 30.5 m x 122 cm with weights and floats attached; mesh size was 1.3 cm. Two PVC pipes bent at the bottom were fixed vertically at the ends of the net and pushed by two persons for a swept area of 20 - 25 m adjacent to the shoreline. All fish caught in the traps and seine net were identified, enumerated, measured (fork length and total length) to the nearest millimeter, and released at the point of capture.

Total number of fishes and species per census were examined by site and by habitat individually with a Kruskall-Wallis one-way ANOVA on ranks (Sokal and Rohlf 1981) after log (x+1) and square root transformations failed to pass Kolmogorov-Lilliefors normality test. Differences in the size distribution of individuals among habitats and within sites were tested with Chi-square contingency tables. Fish species diversity was calculated from the Shannon-Weaver Diversity Index (H') (Shannon and Weaver, 1949) and the evenness index (J') was calculated as well for each census (Pielou 1978).

Number of fish and species abundance caught on fish traps and seine nets were examined by area with a Kruskall-Wallis one-way ANOVA on ranks (Sokal and Rohlf 1981) after log (x+1) and square root transformations failed to pass Kolmogorov-Lilliefors normality test. Species were ranked in order of abundance.

RESULTS

Visual Strip Transect Census

A total of 4471 individuals were observed in visual censuses representing 54 species and 21 Families in Cottongarden Bay. The most abundant fish was the slippery dick *Halichoeres bivitattus* (38%) followed by the Bucktooth parrotfish *Scarus radians* (29%) (Table 1). A total of 7829 fishes representing 66 species and 25 Families were observed in Teague Bay. Species composition were dominated by Haemulid recruits *Haemulon spp.* (38%) followed by the slippery dick *Halichoeres bivitattus* (13%) of the total fish abundance (Table 1). The species composition at Yellowcliff Bay consisted of a total of 7388 individuals representing 74 species and 27 Families. Once again grunt recruits (Haemulidae) dominated the species composition accounting for 33% of the total fish abundance followed by the slippery dick *Halichoeres bivitattus* (27%) (Table 1).

Seagrass was the most abundant habitat found in all Cottongarden Bay transects accounting almost 94% of the habitats found. Seagrass and algal plain were the most abundant habitats found in all Teague Bay transects accounting 84% and 11% of the habitat, respectively. Most of the habitat found was seagrass (60%) and algal plain (26%) in Yellowcliff Bay (Figure 2).

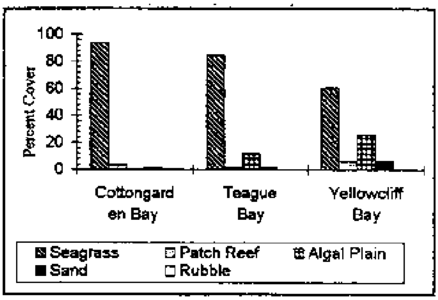


Figure 2. Proportion of habitat destribution (N = 72,000 m^2)

Monthly variations in mean fish abundance and species richness densities among sites showed different peaks throughout the year of study in all three embayments. A very distinct decline in number of individuals and species densities was observed from January to March, 1999 (Figure 3a, 3b).

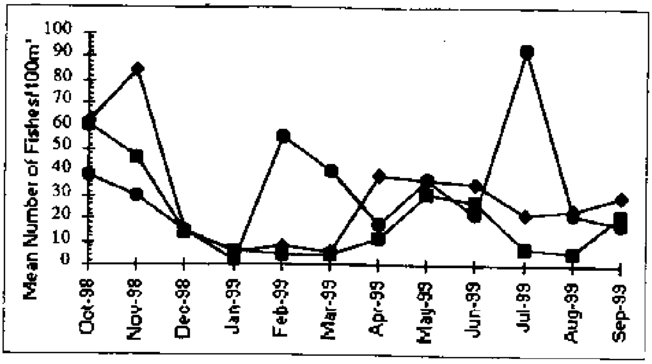
Species heterogenity and evenness mean values followed the same monthly trends as for abundance of fishes and number of species densities. Species diversity and evenness were higher in October 1998 and through the summer months and decreased during the months of January through March (1999). There were no significant differences ($p>0.05$) in number of fishes, number of species, diversity and evenness indexes among sites. Teague Bay had the highest number of fish density per site while Yellowcliff Bay had the highest number of species densities per site. Yellowcliff Bay had higher H' values while J' values where higher in Cottongarden Bay (Figure 4a, 4b).

When number of fish and number of species densities per habitat were examined, patch reef habitats had the highest number of fish and species densities except in Cottongarden Bay where rubble habitats had the highest density (Figure 5a, 5b). Kruskall-Wallis ANOVA demonstrated significant differences ($p < 0.05$) in abundance and species diversity among habitats. Patch reef and rubble habitats had significantly higher mean number of fish per transect than algal plains and sand habitats. As for species richness, patch reef and rubble habitats had significantly higher mean number of species per transect than all other habitats.

Diversity indexes (H') were also significantly higher ($p<0.05$) in patch reefs than the rest of the other habitats. Rubble and seagrass habitats had diversity indexes higher than sand and algal plain (Figure 6a). There were no significant differences ($p=0.05$) in evenness values among habitats (Figure 6b).

Table 1. Summary of visual strip transect results and the four most abundant species found.

Species Total	Fish	Percent
Cottongarden Bay		
Halichoeres bivittatus	1,687	37.73
Sparisoma radians	1,289	28.83
Thalassoma bifasciatum	218	4.88
Scarus taeniopterus	138	3.09
Total Number of Fish	4,471	
Total Number of Species	54	
Teague Bay		
Haemulon spp.	2,983	38.10
Halichoeres bivittatus	1,058	13.51
Haemulon aurolineatum	921	11.76
Sparisoma radians	496	6.34
Total Number of Fish	7,829	
Total Number of Species	66	
Yellowcliff Bay		
Haemulon spp.	2,465	33.36
Halichoeres bivittatus	1,967	26.62
Sparisoma radians	443	6.00
Acanthurus chirlurgus	281	3.80
Total Number of Fish	7,388	
Total Number of Species	74	



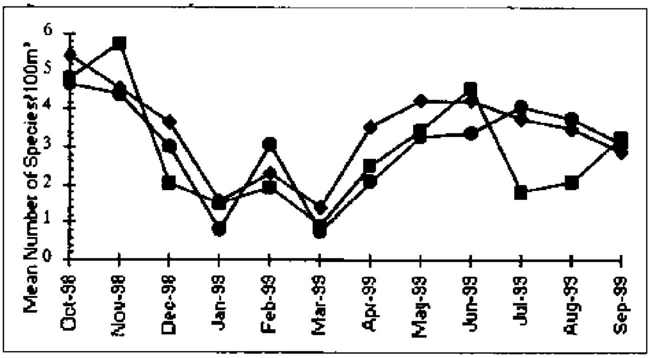


Figure 3. A) Monthly variation in mean number of fish per site; B) Monthly variation of mean number of species per site

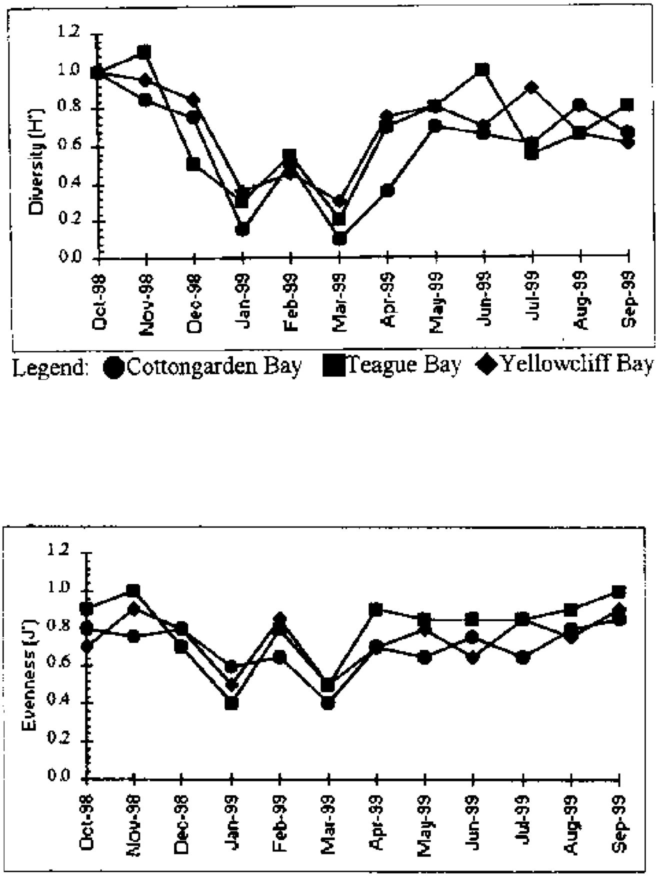
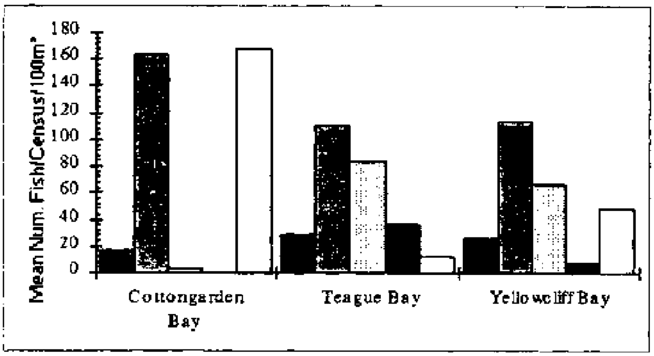


Figure 4. A) Monthly variations in Shannon Weaver Diversity per site; B) Monthly variations in Pielou Evenness per site



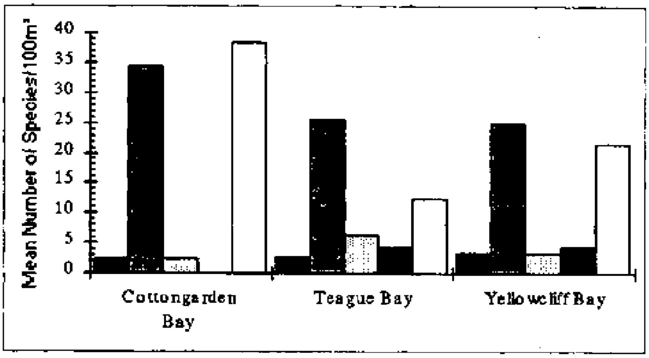


Figure 5. A) Mean number of fish per habitat; B) Mean number of species per habitat

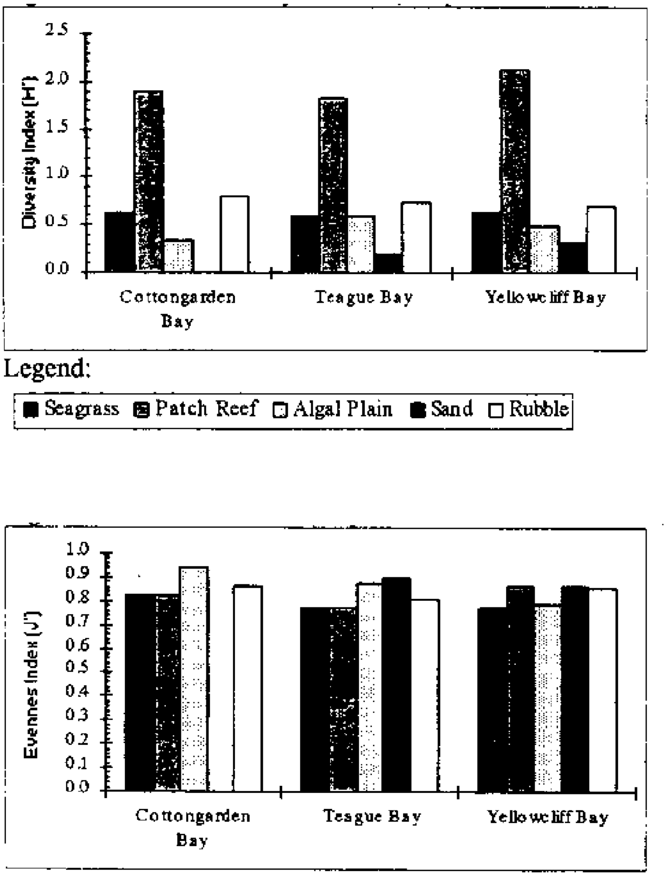


Figure 6. A) Mean Diversity Indices (H') per habitat; B) Mean Evenness (J') per habitat

Analysis of habitat and site utilization by size groups demons-trated significant differences ($p < 0.05$) in size distribution proportions among habitats and sites. Seagrass habitat had the highest proportions on all size groups among habitats followed by algal plain, except in the size group 5 - 10 cm where patch reef habitats had the highest proportion in this size group. All habitats contained recruits < 5 cm and the vast majority of recruits were distributed on seagrass, algal plain, and patch reef habitats. Examination of size-group distribution per bay showed that Yellowcliff Bay had more recruits < 5 cm while Teague Bay had larger individuals > 10 cm. (Table 2a, 2b).

Table 2. Size Class Distribution per Habitat.

Habitat	Area (m2)	Habitat Cover (%)	Species Richness	Fish Abundance				Absolute Density	
				<5 cm	5-l0cm	>10 cm	Total	A	B
Seagrass	57326	79.62	81	8492	3907	1334	13733	0.24	0.001
Algal Plain	9052	12.57	58	1662	409	1019	3090	0.34	0.006
Patch reef	2836	3.94	63	1019	832	349	2200	0.77	0.020
Sand	2462	3.42	12	207	47	25	279	0.11	0.005
Rubble	324	0.45	31	324	44	18	386	1.19	0.090
Total	72000			11704	5239	2745	19688		

A= Fish Abundance Absolute Density (Num. Fish/m^2)

B= Species Richness Absolute Density (Num. Species/m^2)

Fish Trap Results

A total of 696 individuals were caught by traps representing 16 species and 12 Families in Cottongarden Bay (Table 3). The most abundant fish was the spotted goatfish *Pseudupeneus maculatus* (38%) followed by the bucktooth parrotfish *Sparisoma radians* (38%) at Cottongarden Bay. There was a total of 760 fishes representing 26 species and 14 Families around Teague Bay. The species composition around Teague Bay was dominated by the french grunt *Haemulon flavolineatum* (31%) followed by the spotted goatfish *Pseudupeneus maculatus* (19%). A total of 438 individuals representing 19 species and 11 Families was found at Yellowcliff Bay. The spotted goatfish *Pseudupeneus maculatus* dominated the species composition (47%) followed by the yellowtail snapper *Ocyurus chrysurus* (9%) (Table 3).

Table 3. Size Class Distribution per Site.

Site	Area	Species	Fish Abundance				Absolute Density	
	(m^2)	Richness	<5cm	5-10cm	>10cm	Total	A	B
Cottongarden Bay	24000	54	1956	1873	674	4471	0.18	0.002
Teague Bay	24000	66	4687	1555	1587	7829	0.32	0.002
Yellowcliff Bay	24000	74	5061	1811	516	7388	0.30	0.003
Total	72,000		11704	5239	2745	19688		

A= Fish Abundance Absolute Density (Num. Fish/m^2)

B= Species Richness Absolute Density (Num. Species/m^2)

Table 4. Summary of traps results and the four most abundant species found. (N= 120 total traps for each site).

Species	Total Fish	Percent
Cottongarden Bay		
Pseudupeneus maculatus	269	38.65
Sparisoma radians	267	38.36
Holocentrus adscensionis	54	7.76
Ocyurus chrysurus	32	4.60
Total Number of Fish	696	
Total Number of Species	16	
Teague Bay		
Haemulon flavolineatum	239	31.45
Pseudupeneus maculatus	146	19.21
Ocyurus chrysurus	79	10.39
Haemulon aurolineatum	43	5.66
Total Number of Fish	760	
Total Number of Species	26	
Yellowcliff Bay		
Pseudupeneus maculatus	209	47.72
Ocyurus chrysurus	40	9.13
Sparisoma radians	30	6.85
Haemulon flavolineatum	24	5.48
Total Number of Fish	438	
Total Number of Species	19	

Monthly mean variations in fish abundance caught per trap among sites ranged from 1.8 to 21.8 fish per trap (Figure 7a). Mean Catch Per Unit Effort (CPUE) at Cottongarden Bay was highest in February (CPUE = 13.7 fishes/trap) and lowest in January (CPUE = 1.9). In Teague Bay CPUE was highest in April (21.8 fishes per trap) and lowest in March (CPUE = 2.1). Yellowcliff Bay monthly CPUE was highest in September (CPUE 7.7 fishes/trap) and lowest on April (CPUE=1.8).

In Cottongarden Bay, monthly mean number of species per trap ranged from 1.1 to 2.3, while mean species diversity in Teague Bay ranged from 0.9 to 4.9. Cottongarden Bay CPUE values ranged from 1.1 - 1.9 number of species/ trap. Although Teague Bay had more number of fishes and species richness per trap, there were no significant differences ($p > 0.05$) in the number of fishes, and number of species per site (Figure 7b).

Beach Seine Results

A total of 61 individuals were caught by seine net representing 9 species and 7 Families in Cottongarden Bay (Table 5). The most abundant fish was the permit *Trachinotus falcatus* (26%) followed by the slender mojarra *Eucinostomus jonensis* (19%) at Cottongarden Bay. There was a total of 263 fishes representing 29 species and 19 Families around Teague Bay. The species composition was dominated by the horseye jack *Caranx latus* (28%) followed by the slender mojarra, *Eucinostomus jonensis* (19%). The species composition at Yellowcliff Bay contributed to a total of 10 individuals representing 4 species and 4 Families. The slender mojarra *Eucinostomus jonensis* and the hound fish *Tylosorus crocodrilus* dominated the species composition accounting for 50%, and 30%, respectively, of total fish abundance.

Monthly mean variations in fish abundance caught per seine net haul among sites ranged from 0 to 29 fish per haul (Figure 8a). Mean CPUE at Cottongarden Bay was highest in May (10 fishes per haul) and lowest during the months from December 1998 to February 1999, and from August to September 1999. In Teague Bay CPUE was highest in May (29 fishes/haul) and lowest from January to February 1999. Yellowcliff Bay monthly CPUE remained low throughout the study. The highest CPUE was in December with 4 fishes/haul.

Monthly mean number of species per haul ranged from 0 to 2.5 in Cottongarden Bay while mean species diversity in Teague Bay ranged from 0.5 to 4.67. Cottongarden Bay CPUE values ranged from 0 to 2 species/haul. There were significant differences ($p<0.05$) in the number of fishes and number of species/haul among sites. Teague Bay had significantly more fishes and species/haul than any of the other sites (Figure 8b).

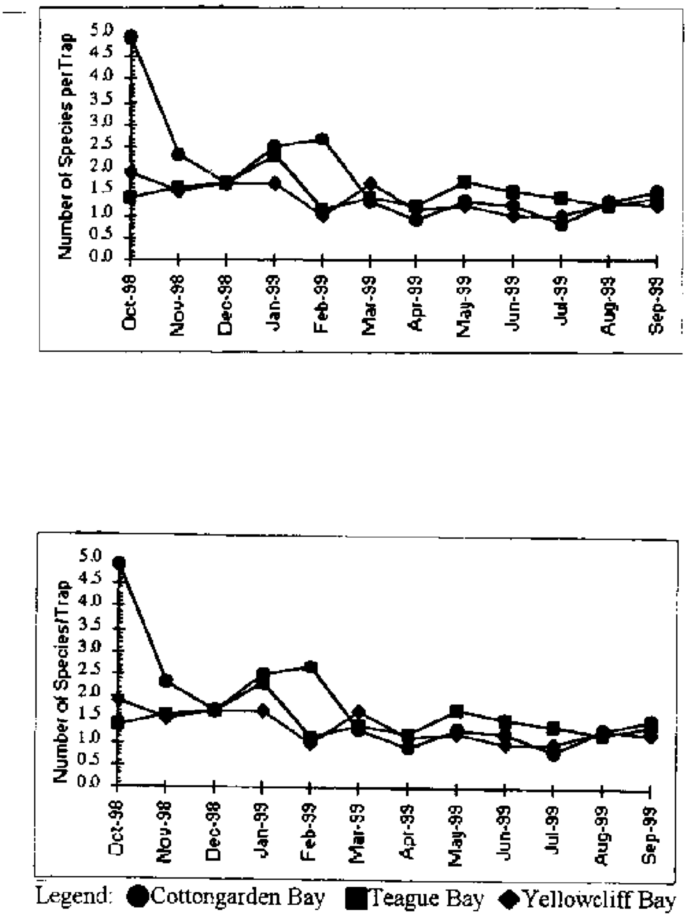


Figure 7. A) Monthly mean variation in number of fishes caught on trap per site; B) Monthly mean variation in number of species caught on trap per site.

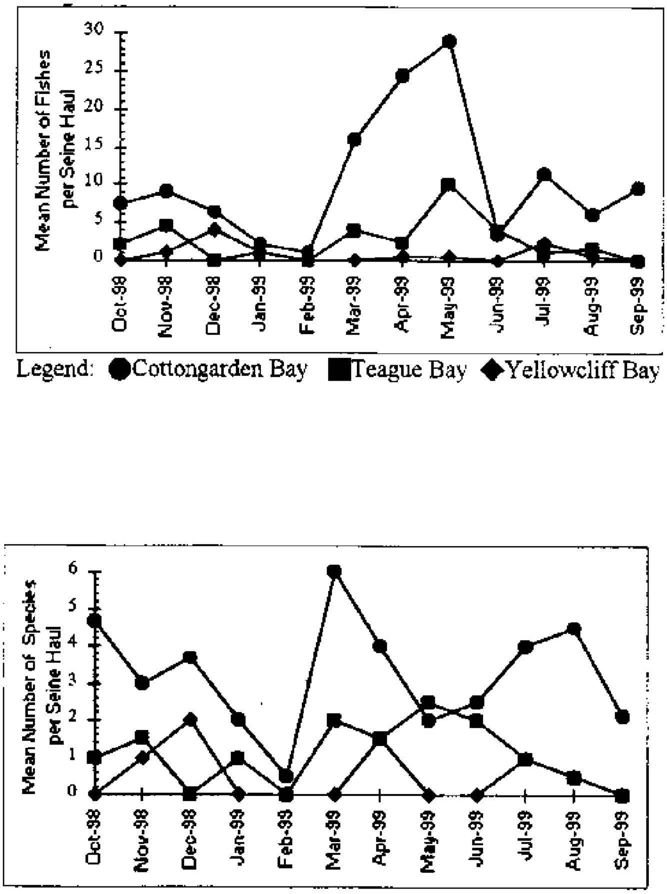


Figure 8. A) Monthly mean variation in number of fishes caught on beach seine net; B) Monthly mean variation in number of species caught on beach seine net

Table 5. Summary of beach seine net results and the four most abundant species found (N= 24 total net hauls/site).

Species	Total Fish	Percent
Cottongarden Bay		
Trachinotus falcatus	16	26.23
Eucinostomus jonesis	12	19.67
Caranx latus	10	16.39
Trachinotus goodei	7	11.48
Total Number of Fish	61	
Total Number of Species	9	
Teague Bay		
Caranx latus	74	28.14
Eucinostomus jonesis	51	19.39
Ocyurus chrysurus	27	10.27
Sphoeroides spengleri	16	6.08
Total Number of Fish	263	
Total Number of Species	29	
Yellowcliff Bay		
Eucinostomus jonesis	5	50
Tylosorus crocodrilus	3	30
Chaetodon capistratus	1	10
Sparisoma radians	1	10
Total Number of Fish	10	
Total Number of Species	4	

DISCUSSION

The role of nearshore habitats for juvenile fishes in the North-east coast of St. Croix is demonstrated to be quantifiable in terms of juvenile fish diversity and abundance by area. The data suggests that some of the function and attributes of these coastal systems such as their high productivity and structural com-plexity are conducive for a nursery function. The presence of nearshore seagrass and patch reefs habitats is likely to influence the juvenile fish assemblages in adjacent coastal areas by enhancing the numbers of micro-habitats, providing refuge from predation and foraging area. Many of the species utilizing nearshore habitats as nursery areas were found throughout the study region and in some cases in relatively high densities. These species included *Haemulon spp.*, *Haemulon aurolineatum*,*Haemulon flavolineatum*, and *Scarus taeniopterus*.

The juvenile fish assemblages in the backreef lagoonal areas of Cottongarden Bay, Teague Bay, and Yellowcliff Bay have a shared trait with those found in many other reef fish populations in that these habitats support few species in large numbers (Ogden and Zieman 1977, Brothers and McFarland 1981). Three species, *Sparisoma radians, Halichoeres bivittatus, Haemulon spp.*, accounted for >65% of all fishes observed in the visual census. The more abundant species of the fish community in this study belonged to the lower trophic level species (Table 1). Some dominant species are recreationally and commercially linked to the local nearshore fishery. The fish inhabiting the backreef areas in Cottongarden Bay, Teague Bay and Yellowcliff Bay were of smaller sizes (< 10 cm) and very few reached over 10 cm. This data indicates that backreef areas apparently plays an important role as a nursery grounds for economically important species at some point during their life cycle.

At the sites studied, fish species formed distinct communities within seagrass, sand, patch reefs, algal plains and rubble habitats. The seagrass and algal plain habitats at all three embayments were dominated by small resident species *Halichoeres spp.*, *Sparisoma radians*, and juveniles of nonresident species that are of economic importance like *Haemulon spp.* and *Ocyurus chrysurus*. Rubble and patch reef habitats harbored more species per transect and were mostly dominated by small juvenile damselfishes, parrotfishes, grunts, and doctorfishes.

Seasonality of species richness and abundance of fishes was observed in most cases, including the number of individuals of some of the most abundant species. This seasonality may be due to several phenomena which are probably not mutually exclusive. Previous studies done at Teague Bay state that during the warmer months from July to November there is a significant increase in algal cover and biomass which probably induces an increase in invertebrate populations (Rogers and Salesky 1981). This increase in food availability may favour some herbivores such as *Sparisoma radians, Scarus croicensis, Scarus taeniopterus* and some invertebrate omnivores *Halichoeres bivittatus, Haemulon spp.* and small carnivorous fishes as *Ocyurus chrysurus*. Nevertheless, seasonal variations may have another complementary origin. Several authors have produced strong evidence for the influence of settlement and recruitment processes of larvae and juvenile fishes on seasonal fluctuations of both species richness and abundance of individuals of fish communities which showed higher values in summer (Williams and Sale 1981, Doherty and Williams 1982). During this study summer pulses of recruitment were evident for some species. This fact affected mainly schooling species such as *Haemulon spp.*, although varying in magnitude between areas.

The lower abundance of fishes particularly recruits in winter which may be a cause of the decrease in number of fish individuals and species diversity in these periods may be due to several mechanisms. The most probable include natural mortality, predation and or migration to other reef areas. The natural mortality and the mortality due to predation probably affect all recruits despite the existence of preferential prey/predator relationship (Hixon 1991). Although some juveniles settle directly into adult habitat such as the pomacentrids, chaetodontids and labrids (Fowler 1990, Eckert 1984). Most juvenile reef fishes occur in shallower areas than their conspecific adults. Thus migrations of some species when the adult or subadult stage is reached from shallow to deeper waters may provide a partial explanation for the seasonal variation observed at least for fish species such as grunts and snappers. However, differences in timing and intensity of recruitment of *Haemulon spp. Halichoeres bivittatus, Sparisoma radians* and *Ocyurus chrysurus* between sites suggests different factors affecting larval supply such as spawning cycle, currents, etc. Lower overall recruitment to Cottongarden Bay may be due to lower larval supply but also shortage suitable habitat. Shortage of habitat is a possibility since habitat cover was different among sites. Yellowcliff Bay had more habitat complexity than Cottongarden Bay. Almost 95% of the habitat found in Cottongarden Bay was seagrass, and occurrence of suitable habitats for fish recruits such as patch reefs algal plain and rubble areas were rare.

Finally, it is demonstrated that the backreef lagoon areas of Cottongarden Bay, Teague Bay and Yellowcliff Bay are important nurseries for many economically important fishes. While variability among sites and months occurred, trends were similar and indicated the importance of nearshore habitats for juvenile fishes. Since the majority of juveniles in nearshore habitats are of economically important species and they appear to preferentially inhabit backreef lagoonal shallow areas, these habitats must be conserved to ensure the continued viability of fisheries resources in St. Croix.

ACKNOWLEDGMENTS

We thank W. Ventura, H. Rivera, G. Santana, C. Cortez, and D. Durant for their logistical support provided during the field survey. This study was funded by U.S. Fish and Wildlife Service Federal Aid Grant; F714 under the Dingell-Johnson Sport Fish Restoration Act.

LITERATURE CITED

Appeldoorn, R.S. C.W. Recksieck, R.L. Hill, F.E. Pagan and G.D. Dennis. 1997. Marine protected areas and reef fish movements; the role of habitat in controlling ontogenetic movements. *Proc 8th Int. Coral Reef. Symp.* **2**:1917-1922.

Austin, H.M. 1971. A survey of the ichthyofauna of the mangroves of western Puerto Rico during December, 1967 - August, 1968. *Caribb. J. Sci.* **11**(1-2):27-39.

Beumer, J.P. 1978. Feeding ecology of four fishes from a mangrove creek in north Queensland, Australia. *J. Fish. Biol.* **12**(5):475-490.

Boulon, R.H. 1986. Fisheries habitats of the Virgin Islands region of ecological importance to the fisheries resources of the Virgin islands Biosphere Reserve. St. Thomas, USVI: Island Resources Foundation.

Brothers, E.R. and W.N. McFarland. 1981. Correlations between otolith microstructure, growth and life history transitions in newly recruited French Grunts (*Haemulon flavolineatum*) Rapp. P.v. reun. *Cons. Perm. Int. Explor. Mer.* **178**:369-374.

Doherty, P.J. and D.McB. Williams. 1988. The replenishment of coral reef fish populations. *Oceanogr. Mar. Biol. Annu. Rev.* **26**:487-551.

Eckert, G.J. 1984. Annual and spatial variation in recruitment of labroid fishes among sever reefs in the Capricorn/Bunker Groups, Great Barrier Reef. *Mar. Biol.* **78**:123-127.

Fowler, A.J. 1990. Spatial and temporal patterns of distribution and abundance of chaetodontid fishes at One Tree Reef, southern GBR. *Mar. Ecol. Prog. Ser.* **64**:39-53.

Hixon, M.A. 1991. Predation as a process structuring coral reef fish communities. Pages 475-508 in: P.F. Sale (ed.) *The ecology of fishes on coral reefs.* Academic Press, London.

Macnae, W. 1974. Mangrove forest and fisheries. Indian Ocean Programme Publ., No. 34, Rome. Indian Ocean Fisheries Commission. 35 p.

Ogden, J.C. and J.C. Zieman. 1977. Ecological aspects of coral reef, seagrass beds in the Caribbean. Pages 377-389 in: *Proc. of Third International Coral Reef Symp.* R.S.M.A.S. Univ. of Miami: 656 pp.

Phillips, R.C. and E.G. Menez. 1988. Seagrassess. *Smithsonian Contributions to the Marine Sciences.* **34**. 104 pp.

Pielou, E.C. 1978. Pages 289 - 315 in: *Population and community ecology. Principles and Methods.*

Robblee, M.B. and J.C. Zieman. 1984. Diel variation in the fish fauna of a tropical seagrass feeding ground. *Bull. Mar. Sci.* **34**(3):335-345.

Rogers, C.S. and N.H. Salesky. 1981. Productivity of *Acropora Palmata* (Lamarck), Macroscopic Algae, and Algal Turf from Tague bay Reef, St. Croix, US Virgin Islands. *J. Exp. Mar. Biol. Ecol.* **49**:179-187

Rooker, J.R. and G.D. Dennis. 1991. Diel lunar and seasonal changes in a mangrove fish assemblages off southwestern Puerto Rico. *Bull. Mar. Sci.* **49**(3):684-698

Shannon, C.E. and W. Weaver. 1949. *The Mathematical Theory of communication.* Urbana III: University of Illinois Press.

Shulman, M.J. 1984. Resource limitation and recruitment patterns in a coral reef fish assemblages. *J. Exp. Mar. Biol. Ecol.* **74**:85-109.

Shulman, M.J. 1985. Recruitment of coral reef fishes: effects of distribution of predators and shelter. *Ecol.* **66**:1056-1066.

Sokal, R.R., and F.J. Rohlf. 1981. *Biometry.* W.H. Freeman and Company, New York. Pp. 959.

Springer, V.G. and A.J. Mcerlean. 1962. Seasonality of fishes on a south Florida shore. *Bull. Mar. Sci.* **12**(1):39-60

Stoner, A.W. and J.M. Waite. 1990. Distribution and behavior of queen conch, *Strombus gigas,* relative to seagrass standing crop. *Fish. Bull.* **88**:573-585.

Williams, D.McB. and P.F. Sale. 1981. Spatial and temporal patterns of recruitment of juvenile coral reef fishes to coral habitats within One Tree Lagoon, Great Barrier Reef. *Mar. Biol.* **64**:245-253.

Zieman, J. C., S.A. Mack and A.L. Mills. 1982. The ecology of the seagrassess of south Florida: a community profile. U.S. Fish. Wildlife. Serv. Biol. Serv. Prog. FWS/OBS-82-85.

Build It, But Will They Come? Preliminary Findings of Refuge Limitation Bottlenecking in Juvenile *Menippe adina* in the Mississippi Sound

VIRGINIA SHERVETTE[1], HARRIET PERRY[2], PATRICIA BIESIOT[1],
KIRSTEN LARSEN[2], AND JAMES WARREN[2].
[1]Department of Biological Sciences
University of Southern Mississippi
Hattiesburg, Mississippi 39406-5018 USA
[2]Gulf Coast Research Laboratory,
Institute of Marine Sciences
University of Southern Mississippi
Ocean Springs, Mississippi 39566-7000 USA

ABSTRACT

Many marine organisms are restricted to habitats which provide essential refuge. *Menippe adina*, the western Gulf stone crab, depends on patchy hard substrata in the otherwise soft-bottomed Mississippi Sound for individual and population survival. *Menippe adina* supports small, developing fisheries in Louisiana and Texas, and occurs as an incidental catch in the blue crab fishery in Mississippi and Alabama. Stone crab zoeae, megalopae, and small juveniles (10 - 24 mm carapace width) are relatively abundant in Mississippi Sound and the species does not appear to be recruitment limited. Larger juveniles are less common and their numbers may be related to quantity and quality of suitable habitat. There is strong evidence that refuge limitation exerts control on both population size structure and density of stone crabs. The establishment of low profile reefs in the Mississippi Sound has provided an opportunity for preliminary investigations of refuge limitation in local stone crab populations. Preliminary results indicate that suitable habitat is lacking for juvenile stone crabs and competition for available habitat may be acute between stone crabs and other xanthids (*Eurypanopeus depressus*, *Panopeus simpsoni*) as well as the toadfish, *Opsanus beta*.

KEY WORDS: Low profile artificial reef, *Menippe adina*, shelter limitation

INTRODUCTION

The western Gulf stone crab, *Menippe adina*, currently supports small, developing fisheries in Louisiana and Texas, and occurs as an incidental catch in the blue crab, *Callinectes sapidus*, fishery in Mississippi and Alabama (Guillory *et al.* 1995). *Menippe adina* can complete its life cycle within Mississippi waters; larvae are not exported offshore, but are retained in the estuary. Stuck and Perry (1992) reported that zoeae, megalopae, and small juveniles (10 - 24 mm CW) were relatively abundant in Mississippi Sound and that the species did not appear to be recruitment limited. Larger juveniles were less abundant and their numbers may be related to quantity and quality of suitable habitat. Thus, survivorship of juvenile crabs may be the prime determinant of year-class strength. There is evidence that shelter limitation exerts control on both population size structure and density in stone crabs.

Caddy and Stamatopoulos (1990) used fractal theory to examine the relationship between topographic complexity (niche availability) and the capacity of natural substrates to support individuals of different sizes. They suggested that the availability of niches, not food, determined both individual and population size and influenced timing and size at onset of migration for some organisms. The shelter bottleneck hypothesis proposes that refuge limitation exerts control on population size structure and density and that its effects are size-specific and not consistent throughout an organism's life history (Caddy and Stamatopoulos 1990). Beck (1997) noted that as individuals near the critical size, growth slows and the population is diminished by increased mortality and/or emigration from the area. Shelter bottlenecks have been identified as contributing to size-structure in several species of nephropid and palinurid lobsters (Parrish and Polovina 1994, Wahle and Steneck 1991, Howard 1980). Beck (1995, 1997) examined size-specific shelter limitation in *Menippe* and suggested that shelter bottlenecks may be important in the regulation of stone crab populations in the northern Gulf of Mexico (GOM) because this area lacks the extensive hard bottom habitats favored by these crabs. In examining shelter limitation at four sites in northwest Florida, he found three of the sites shelter limited and the size classes affected differed among the areas.

In addition to affecting size structure of stone crab populations, availability of suitable habitat may exert control on stone crab densities by providing "escape space". Jeffries and Lawton (1984) examined the concept of "enemy-free space" and identified it as an important component of many species' ecologies. Availability of structurally complex habitat offering refuge from predation was linked to blue crab production by Heck and Coen (1995), Orth and van Montfrans (1990), Wilson et al. (1990), Heck and Thoman (1984), and Heck and Wilson (1987). Heck and Coen (1995) noted that although blue crab recruitment in the GOM greatly exceeded recruitment along the Atlantic coast, numbers of surviving juveniles were similar. They attributed the geographical similarities in juvenile abundance to higher predation rates in the GOM and noted that the greater diversity of low latitude predators may limit predation-free refuges.

The establishment of low profile artificial reefs in the Mississippi Sound by the Mississippi Department of Marine Resources provided an opportunity to investigate refuge limitation in juvenile stone crabs and to examine the roles of predation and inter-specific competition in controlling population levels. These reefs provide habitat for an array of marine invertebrates, including *Menippe* *adina*, *Panopeus simpsoni*, and *Eurypanopeus depressus* and known predators of these xanthid crabs including toadfish, *Opsanus beta* (Larsen et al. 1999, Perry et al. 1999).

MATERIALS AND METHODS

The site location is shown in Figure 1. The area is characterized by small natural patch oyster reefs and a newly established low profile artificial reef.

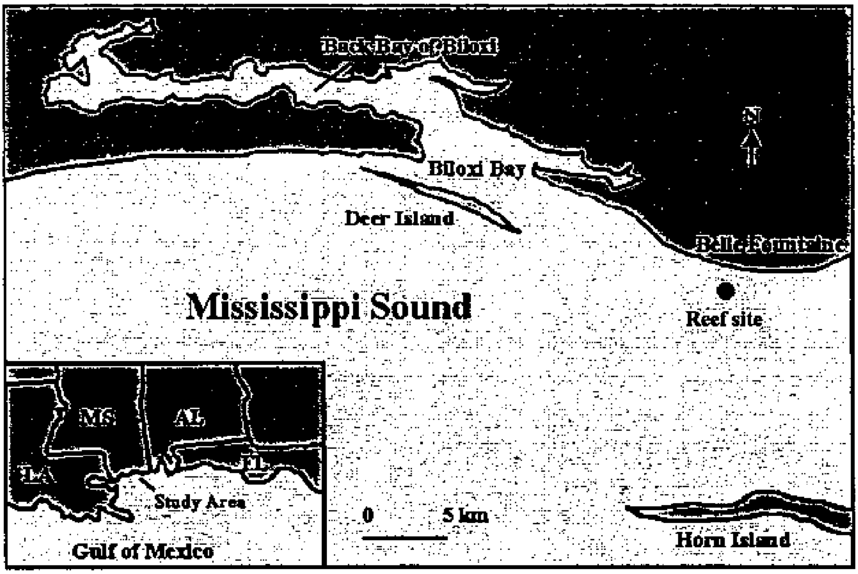


Figure 1. Location of low profile artificial reef

To test for refuge limitation in local stone crab populations, experimental niche samplers were constructed. The samplers, composed of sections of PVC pipe imbedded in a cement matrix (Figure 2), were designed to measure stone crab occupancy at size. Samplers were located on a portion of the artificial reef site at a distance of approximately 200 m from shore. Pipe diameters were selected to cover the size range of stone crab juveniles collected in the Mississippi Sound (Stuck and Perry 1992). Four sizes of pipe were used in constructing the samplers: 13 mm (0.5 in), 26 mm (1.0 in), 39 mm (1.5 in), and 52 mm (2.0 in). Sections of pipe were imbedded at a 45° angle in a 0.5 m^2 cement matrix. The lip of the pipe was flush with the cement. Pipe length was scaled in proportion to pipe diameter to accommodate stone crab morphometry. During retrieval, samplers were covered with foam-backed tops to prevent escapement of animals. Each sampler contained ten numbered pipes of a single size placed equidistant apart (one treatment). Twelve samplers, representing the four treatments with three replicates, were randomly deployed over the reef.

On retrieval, all organisms were removed from the pipes and placed in labeled containers corresponding to the hole diameter and location in the cement sampler. Samplers were returned to the water immediately after removal of organisms. The animals were transported to the Gulf Coast Research Laboratory (GCRL), in Ocean Springs, MS, for analysis. Crab species were measured to the nearest 0.1 mm carapace width (CW) and carapace length and weighed to the nearest 0.1 g. Fish were measured to the nearest 0.1 mm total length (TL) and standard length (SL) and weighed to the nearest 0.1 g. Sampling began in April 1999, with all samplers checked every two weeks. Temperature and salinity were measured in the field.

Figure 2. Shelter samplers

RESULTS

Stone crabs, *Menippe adina*, consistently began to occupy the shelters by late June (Figure 3), with frequency of occurrence ranging from 3 - 30% in the four different shelter size classes. The two smallest diameter shelter size classes (13 and 26 mm) were occupied most frequently throughout the study, but stone crabs also occurred in one or both of the two largest shelter size classes (39 and 52 mm) between 28 June and October 1.

The xanthid crab, *Panopeus simpsoni*, occupied shelters from the beginning of the study (Figure 4) but had the greatest frequency of occurrence in the two smallest shelter size classes during the first three sampling periods, when *M. adina* occupation was low or nonexistent. Between 2 September and 1 October, *P. simpsoni* occurred only sporadically in the two largest shelter size classes.

The xanthid crab, *Eurypanopeus depressus*, was present during each sampling period (Figure 5), but its percent occurrence was lowest among the three crab species, usually below 10%. This crab occurred with greatest frequency in the smallest shelter size class, but was found in all four sizes of shelter.

Toadfish, *Opsanus beta*, dominated the 52 mm diameter shelters and occurred occasionally in the 39 mm diameter shelters (Figure 6). Toadfish showed a maximum occupation rate of 70% on 28 June and a low of 7% in July and on 2 September. No toadfish occurred on the last two sampling days.

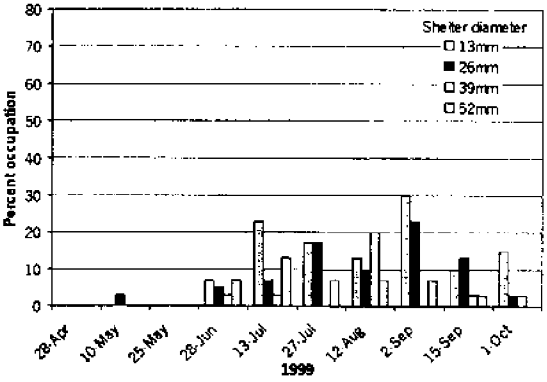


Figure 3. Percent occupation by shelter diameter for *M. adina*

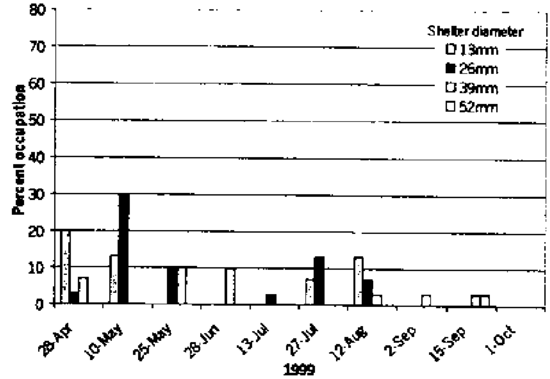


Figure 4. Percent occupation by shelter diameter for *P. simpsoni*

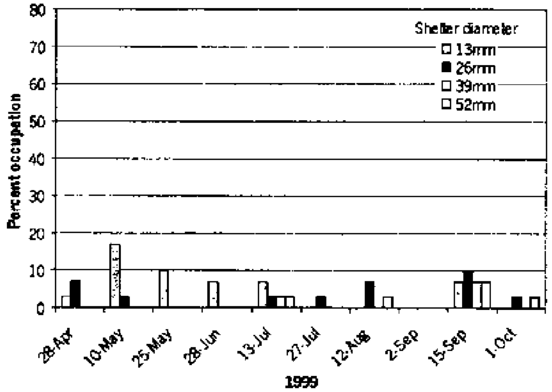


Figure 5. Percent occupation by shelter diameter for *E. depressus*

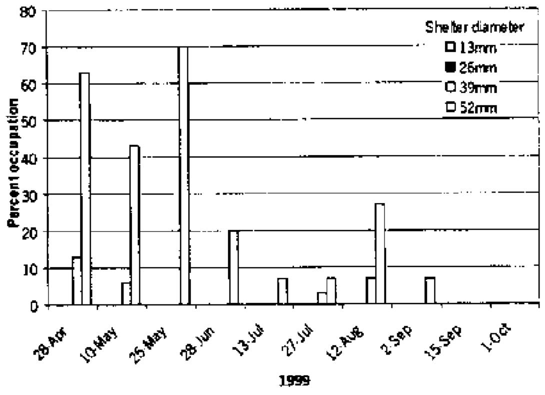


Figure 6. Percent occupation by shelter diameter for *O. beta*

In Figure 7, carapace widths of the three crab species are compared in relation to their occurrence in the different sizes of shelter. Crabs with carapace widths between 6 - 17 mm, regardless of species, occurred in the 13 mm diameter shelters (Figure 7). In terms of absolute numbers, more *Menippe adina* occupied this smallest diameter shelter than either of the two other xanthids. However, small crabs (< 20 mm CW) of each species also occurred in each of the three larger sized shelters, sometimes in combination with toadfish. The size range of crabs in the 26 mm diameter shelters was 5 - 29 mm for *M. adina*, 5 - 26 mm for *Panopeus simpsoni*, and 5 - 21 mm for *Eurypanopeus depressus*. Only two *P. simpsoni* with carapace widths greater than 30 mm were found in the 39 mm diameter shelters and none in the 52 mm diameter shelters whereas six *M. adina* with carapace widths greater than 30 mm were found in the 39 mm diameter shelters and six in the 52 mm diameter shelters.

Shelter occupation rates for crabs with carapace widths less than 25 mm are given in Figure 8. During the first two months of the study, only *Panopeus simpsoni* and *Eurypanopeus depressus* occurred in the shelters. From June onward, *Menippe adina* had a greater percent occupation rate than either of the other two xanthids.

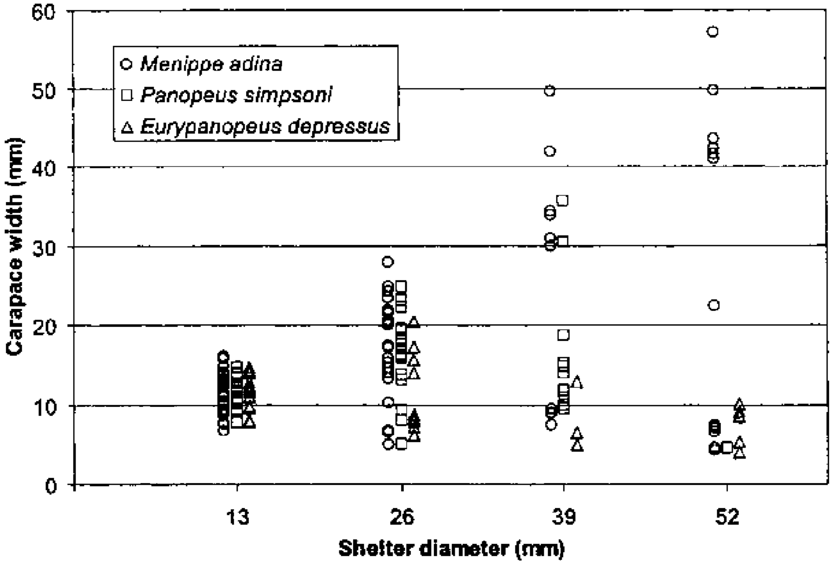


Figure 7. Size frequency distribution of crabs by shelter diameter

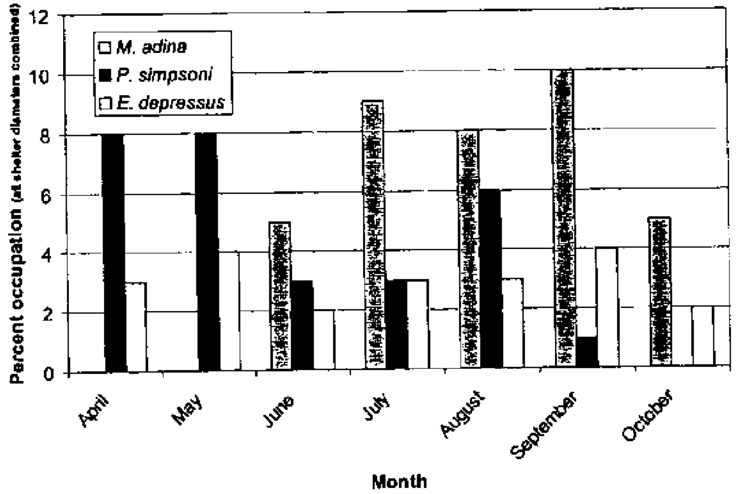


Figure 8. Monthly percent occupation of shelters by crabs under 25 mm CW

DISCUSSION

Shelter, competition, and predation are important factors in shaping community structure. For shelter-dwelling marine organisms, quantity and quality of suitable habitat may be more important in regulating population size than food availability. The role of refuge limitation in determining juvenile stone crab population levels has been addressed in several studies with size-specific shelter bottlenecks identified as being influential in setting population abundance (Gibbs 1994, Beck 1995, 1997). Although shelter occupation data from the current study are limited to the initial seven months of sampler deployment, there was no evidence of shelter limitation for the size classes of crabs observed. Occupancy rate in shelters did not exceed 30 % for any of the crab species collected to date. With one exception, the two smallest diameter shelters were the most frequently occupied throughout the study.

Selection of shelters appeared to be related to size frequency of animals occurring on the reef. Size class distributions of crabs available to occupy shelters are a function of overall species size and recruitment. During the time period covered in this study, recruitment occurred for all three xanthid species. Temporal periodicity of recruitment, however, was variable. *Menippe adina* did not occur on samplers until June. Of the three species, *Eurypanopeus depressus* was found less frequently overall. This is the smallest of the xanthids collected and it occurred primarily in the smallest shelter. Perry et al. (1999) found this crab to be highly abundant in the area of study and its low occupancy rate suggests that it is not habitat limited. After the arrival of stone crabs in June, occupancy rates for the other species declined, suggesting that some form of competition may be occurring even though the majority of available shelters were unoccupied.

Shelters were also occupied by the toadfish, *Opsanus beta*. Many of the 52 mm diameter shelters were occupied by two toadfish, one male and one female, during the first two sampling periods but after that only males, apparently guarding toadfish eggs, were found. Toadfish occupancy rates dropped in June and remained low thereafter, possibly because peak reproductive season had passed.

Stomach content analyses of *Opsanus beta* associated with the samplers demonstrated that toadfish preyed on all three species of xanthids. Predation may be an important factor in controlling population levels of these crabs.

ACKNOWLEDGMENTS

This work is a result of research sponsored in part by the National Oceanic and Atmospheric Administration, U.S. Department of Commerce, the Mississippi-Alabama Sea Grant Consortium, and the University of Southern Mississippi, Institute of Marine Sciences, Gulf Coast Research Laboratory. The U.S. Government and the Mississippi-Alabama Sea Grant Consortium are authorized to produce and distribute reprints notwithstanding any copyright notation that may appear hereon. The views expressed herein are those of the authors and do not necessarily reflect views of NOAA or any of its subagencies.

We are indebted to the following individuals who helped with field sampling: Robin McCall, Bradley Randall, Jude LeDoux, Eric Pederson, Brett Blackburn, Chuck Flowers, Todd Randall, Ronnie Bond, Glenn Hendrix, Wes Devers, John Anderson, and Monty Simmons. David Chiluiza, Lesber Salazar, Windsor Aguirre, Pam Schofield, Jason Lemus, Andres Soto also assisted with various project activities. We thank Lisa Engel for her administrative expertise.

REFERENCES CITED

Beck, M.W. 1995. Size-specific shelter limitation in stone crabs: a test of the demographic bottleneck hypothesis. *Ecology* 76(3):968-980.

Beck, M.W. 1997. A test of generality of the effects of shelter bottlenecks in four stone crab populations. *Ecology* 78(8):2487-2503.

Caddy, J.F. and C. Stamatopoulos. 1990. Mapping growth and mortality rates of crevice-dwelling organisms onto a perforated surface: the relevance of 'cover' to the carrying capacity of natural and artificial habitats. *Estuar. Coast. Shelf Sci.* 31:87-106.

Gibbs, B.R. 1994. Experiments on shelter availability and interactions on juvenile stone crabs, *Menippe adina* (Williams and Felder). Master's Thesis, University of South Alabama, Mobile, Alabama. 62 pp.

Guillory, V., H.M. Perry, and R.L. Leard. 1995. A profile of the western Gulf stone crab, *Menippe adina*. Gulf States Marine Fisheries Commission, Technical Report 3. 60 pp.

Heck, K.L., Jr. and L.D. Coen. 1995. Predation and the abundance of juvenile blue crabs: a comparison of selected east and Gulf coast (USA) studies. *Bull. Mar. Sci.* **57**(3):877-883.

Heck, K.L., Jr. and T.A. Thoman. 1984. The nursery role of seagrass meadows in the upper and lower reaches of the Chesapeake Bay. *Estuaries* **7**(1):70-92.

Heck, K.L., Jr. and K.A. Wilson. 1987. Predation rates on decapod crustaceans in latitudinally separated seagrass communities: a study of spatial and temporal variation using tethering techniques. *J. Exp. Mar. Biol. Ecol.* **107**:87-100.

Howard, A.E. 1980. Substrate controls on the size composition of lobster (*Homarus gammarus*) populations. *J. Cons. Int. Explor. Mer* **39**:130-133.

Jeffries, M.J. and J.H. Lawton. 1984. Enemy-free space and the structure of ecological communities. *Biol. J. Linn. Soc.* **23**:269-286.

Larsen, K., H. Perry, J. Warren, and C. Trigg. 2001. Seasonal colonization of artificial low profile reefs in Mississippi coastal waters: vertebrates. *Proc. Gulf Carib. Fish. Inst.* **52**:488-497.

Orth, R.J. and J. van Montfrans. 1990. Utilization of marsh and seagrass habitats by early stages of *Callinectes sapidus*: a latitudinal perspective. *Bull. Mar. Sci.* **46**:126-144.

Parrish, F.A. and J.J. Polovina. 1994. Habitat thresholds and bottlenecks in production of the spiny lobster (*Panulirus marginatus*) in the northwestern Hawaiian Islands. *Bull. Mar. Sci.* **54**:151-163.

Perry, H., K. Larsen, J. Warren, and C. Trigg. 2001. Seasonal colonization of artificial low profile reefs in Mississippi coastal waters: invertebrates. *Proc. Gulf Carib. Fish. Inst.* **52**:498-511.

Stuck, K.C. and H.M. Perry. 1992. Life history characteristics of *Menippe adina* in Mississippi coastal waters. Pages 82-98 in: T.M. Bert (ed.), *Proceedings of a Symposium on Stone Crab (Genus Menippe) Biology and Fisheries*. Fla. Mar. Res. Publ. 50.

Wahle, R.A. and R.S. Steneck. 1991. Recruitment habitats and nursery grounds of the American lobster *Homarus americanus*: a demographic bottleneck? *Mar. Ecol. Prog. Ser.* **69**:231-243.

Wilson, K.A., K.W. Able, and K.L. Heck, Jr. 1990. Predation rates on juvenile blue crabs in estuarine nursery habitats: evidence for the importance of macroalgae (*Ulva lactuca*). *Mar. Ecol. Prog. Ser.* **58**:243-251.

Dynamics of Low-profile, Inshore Artificial Reefs in The Mississippi Sound

JUDE J. LEDOUX, JAMES R. WARREN,
WESLEY DEVERS, LISA ENGEL, and MICHAEL BUCHANAN
The University of Southern Mississippi
Institute of Marine Science
Gulf Coast Research Laboratory,
P.O. 7000
Ocean Springs, Mississippi 39566-7000 USA

ABSTRACT

The popularity of low-profile reefs as fishing banks and the need to increase the potential for harvestable food from the ocean has prompted many coastal states to begin artificial reef programs. The territorial waters of Mississippi contain several open Gulf artificial reef sites and inshore, artificial, low profile reefs within the Mississippi Sound. Past studies have provided valuable information on the success of the offshore reefs and their attraction and possible support of recreationally important species. Little documented data is available about inshore, low profile reefs relative to their productivity and ability to attract fish. Mississippi has established over twenty inshore low profile reefs using oyster shell, crushed limestone, and concrete rubble. The increased number of artificial fishing reefs in Mississippi's coastal waters over the last five years has provided an immediate need in obtaining information on the association of fish populations with these reefs and their subsequent use by the fishing public. An assessment and monitoring program for four of these reefs was implemented in 1998 using entanglement gear, a 4.88meter lined otter trawl, and custom fish traps. Intercept creel surveys were also initiated to obtain usage data by the fishing public. Substrate samples from oyster and limestone were also taken to provide information on seasonality and colonization of benthic fauna on these reefs. These data may provide information on the trophic relationships between attracted finfishes and a particular substrate. The attraction of finfishes by the artificial reef varied significantly by the geographic location, substrate composition, various abiotic factors, and a variety of anthropogenic factors. *Cynoscion arenarius*, *Scomberomorus maculatus*, and *Micropogonias undulatus* were the most numerous, in decreasing abundance, recreationally important species observed from the sampled reefs. These species were more numerous on the reef site than to the adjacent off reef sites; however the differences were not statistically different because of the high variability in the catch.

KEY WORDS: Estuarine, low profile artificial reef, recreational fishing

INTRODUCTION

Artificial reefs are generally constructed in coastal waters to enhance fish populations and improve near shore fisheries as the result of fish attraction and/or production of new biomass (Fabi and Fiorentini 1994). The primary goals of artificial reefs in coastal habitats have been to enhance the production of reef-associated species (i.e., macroalgae, invertebrates, and fishes) and to increase the convenience or efficiency of harvesting reef-associated species (Seaman et al. 1989, Seaman and Sprague 1991, Pratt 1994). Inshore artificial reefs provide additional advantages for open gulf fishermen by providing proximate, economical access to fishing locations during times when inclimate weather discourages offshore travel. The territorial waters of Mississippi contain several open Gulf artificial reef sites and approximately twenty inshore, artificial, low profile reefs within the Mississippi Sound. Stroud (1965) has provided valuable information on the success of the offshore reefs and their attraction and possible support of recreationally important species. Little documented data is available about inshore, low profile reefs relative to their productivity and ability to attract fish. Mississippi, over the past twenty years, has established and maintained several inshore low profile reefs using clam shell, oyster shell, crushed limestone, and concrete rubble. With the increase over the past five years in new reefs being developed in Mississippi's coastal waters an immediate need is warranted in obtaining information on the association of fish populations with these reefs and their subsequent use by the fishing public.

The main objective of the study was to evaluate the species composition and the potential attraction of the recreationally important species by the low-profile artificial reefs placed in the Mississippi Sound. These species include, but are not limited to: Red drum (*Sciaenops ocellatus*), Spotted Seatrout (*Cynoscion nebulosus*), Silver Trout (*Cynoscion nothus*), White Seatrout (*Cynoscion arenarius*), Atlantic Croaker (*Micropogonias undulatus*), Southern Kingfish (*Menticirrhus americanus*), Southern Flounder (*Paralichthys lethostigma*), Spanish Mackerel (*Scomberomorus maculatus*), and Cobia (*Rachycentron canadum*).

MATERIALS AND METHODS

An assessment and monitoring program for four of twenty-two available reefs were implemented in March 1998. The four reefs chosen were geographically wide spread along the entire Mississippi Coast (Figure 1). Two reefs were composed of crushed lime stone and two were composed of concrete rubble. Each sample reef was located in highly accessible areas well known and utilized by the fishing public. The sample sites selected were: Hancock County Reef, located just west of the mouth of St. Louis Bay (N 30°17.298, W 89°20.831); Long Beach Reef, located just east of the Long Beach Harbor (N 30°20.852, W 89°08.016); East Deer Island Reef; located just east of Deer

Island(N 30°21.170, W 88°48.310); and Round Island Reef, located just west of Round Island (N 30°17.570, W 88°35.884).

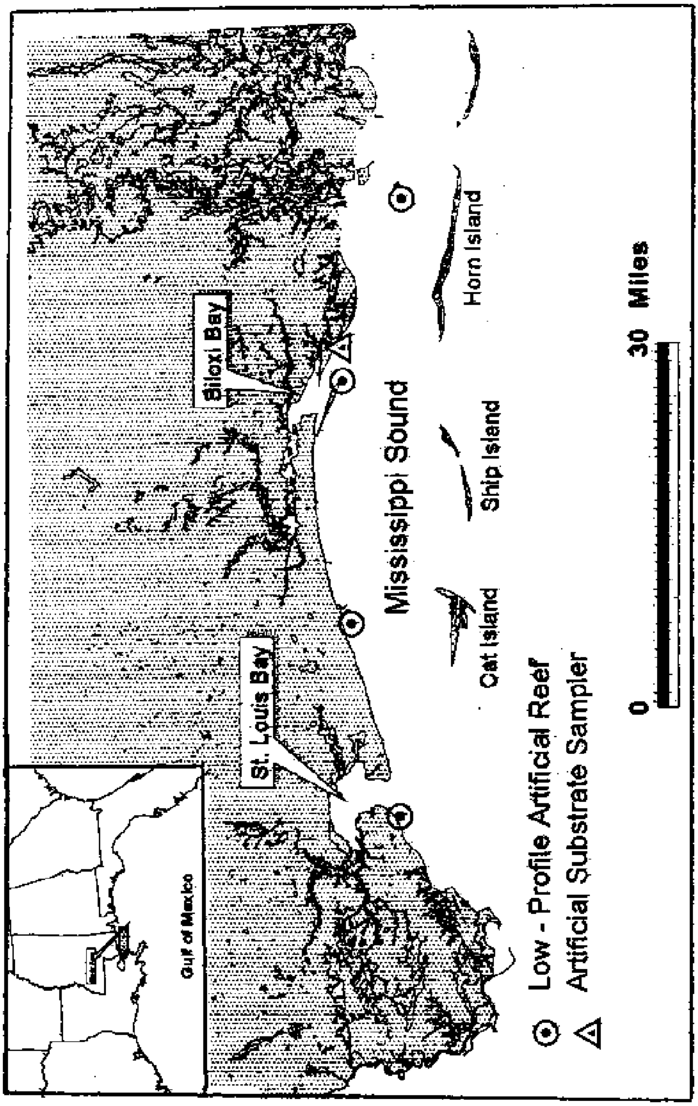


Figure 1. Low-Profile, Inshore, Artificial Reef Sample Sites in the Mississippi Sound

Each site was sampled once a month using a 228.6 meter multi-mesh size gill net, a 4.88 meter lined otter trawl, and up to five custom fish traps. The gill net was deployed across the reef and was allowed to fish for one hour. While the gill net was in the water, the five traps were randomly placed over the reef, and a ten minute trawl sample was taken on the reef. Upon retrieval of the on-reef gill net, a second gill net set was deployed at an adjacent "control" site approximately 0.8 kilometers from the reef. A ten minute trawl sample was obtained at approximately the same distance from the reef site. The fish traps were retrieved after the end of the net sampling on the reef and control site (approximately three hours). Total number and length range were recorded in the field for all non-recreational species. Total length, fork length, standard length (mm), weight (g), and sex was recorded for recreational species after returning to the Gulf Coast Research Laboratory (GCRL).

Bimonthly intercept creel surveys were conducted. The survey period was divided into weekends/holiday's and weekdays. From these two categories, dates were randomly selected: one weekday and one weekend/holiday were chosen. Survey questions included time spent fishing, and species and size range caught. At the end of each interview the captain of the vessel was given a "trip card" to be filled out at the end of the fishing day and mailed to GCRL. This card provided additional information on the numbers and size of fish that were caught.

Data was analyzed using Wilcoxon Paired Signed Rank Test.

RESULTS AND DISCUSSION

All finfish species captured can be found in Table 1. Forty-seven (47) total gill net sets and trawls were done on the four reef sites over the twelve-month sampling period, producing forty-six (46) different species of finfish.

Preliminary statistical analysis was done on five recreational species. Species included: *C. arenarius*, *C. nebulosus*, *M. americanus*, *M. undulatus*, and *S. maculatus*. The total number captured on the reef and off the reef were paired for each species. A paired sign rank test was used to determine if the number on the reef was significantly different from off the reef. A "p" value was used to determine whether or not to reject the null hypothesis that the means = 0 for an alpha value = 0.05. P Values ranged from 1.0 to 0.0119974. Of the five most abundant recreational species, *C. nebulosus*, was the only fish that was statistically more abundant (p = 0.0119974) on the reef than off the reef sites.

C. arenarius was the most abundant fish reported by recreational fishers on the artificial reefs (Figure 2). Creel catches of recreational fishers contained *C. arenarius* thirty-four (34) percent of the time. *M. americanus*, *C. nebulosus*, and *M. undulatus* were also present twenty-five (25), Eighteen (18), and 4 percent of the time, respectively.

The attraction of finfishes to the artificial reef varied significantly by the geographic location, substrate composition, various abiotic factors, and a variety of anthropogenic factors. The number of these species, generally speaking, was more numerous on the reef site when compared to adjacent off reef sites. Crushed limestone expresses itself as a suitable artificial habitat. The reefs composed of this substrate material have attracted a variety of both pelagic and bottom dwelling fishes.

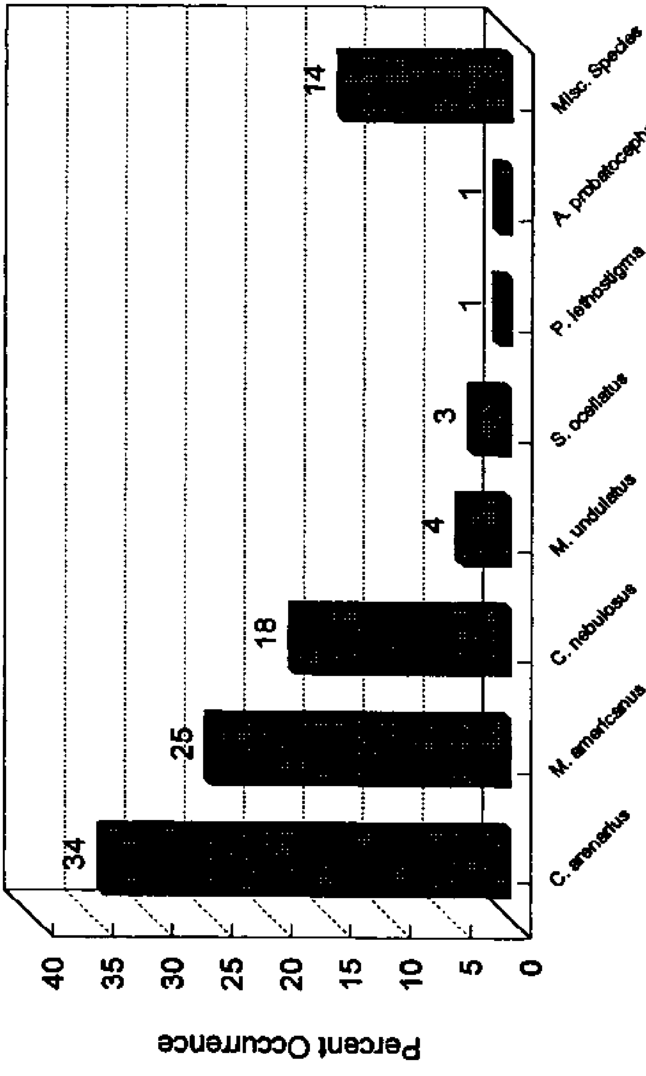


Figure 2. Species occurrence in creel catch in Mississippi coastal waters

Table 1. Total infish catch by substrate and gear type

Crushed Limestone/Oyster Shell				Concrete Rubble			
Gill Net		Trawl		Gill Net		Trawl	
Oligoplites saurus	2054	*Anchoa mitchilli*	27522	*Oligoplites saurus*	1207	*Anchoa hepsetus*	29625
Leiostomus xanthurus	245	*Bairdiella chrysura*	3686	*Leiostomus xanthurus*	278	***Micropogonias undulatus***	336
Cynoscion nebulosus	230	***Cynoscion arenarius***	1292	*Rhizoprionodon terranovae*	139	*Brevoortia patronus*	46
Cynoscion arenarius	101	*Arius felis*	146	***Cynoscion arenarius***	132	*Leiostomus xanthurus*	33
Pogonias cromis	76	***Micropogonias undulatus***	138	***Menticirrhus americanus***	81	*Trichiurus lepturus*	33
Dorosoma petenense	73	*Leiostomus xanthurus*	123	*Pogonias cromis*	69	***Cynoscion arenarius***	25
Micropogonias undulatus	67	*Peprilus burti*	96	*Larimus fasciatus*	46	*Chloroscombrus chrysurus*	22
Menticirrhus americanus	58	***Menticirrhus americanus***	54	*Dorosoma petenense*	24	*Peprilus burti*	13
Dorosoma cepedianum	24	*Lagodon rhomboides*	23	***Cynoscion nebulosus***	24	*Chaetodipterus faber*	12
Larimus fasciatus	23	*Prionotus tribulus*	12	***Micropogonias undulatus***	17	*Bairdiella chrysura*	9
Peprilus alepidotus	18	*Chloroscombrus chrysurus*	11	*Peprilus burti*	17	*Synodus foetens*	6
Rhinoptera bonasus	14	*Dasyatis sabina*	11	***Paralichthys lethostigma***	16	*Citharichthys spilopterus*	4
Peprilus burti	12	*Peprilus alepidotus*	5	*Dorosoma cepedianum*	14	*Dorosoma petenense*	4
Orthopristis chrysopterus	11	*Chaetodipterus faber*	5	*Alosa chrysochloris*	10	*Arius felis*	4
Paralichthys lethostigma	5	*Symphurus plagiusa*	4	*Citharichthys spilopterus*	8	*Lagodon rhomboides*	3
Prionotus tribulus	4	*Dorosoma petenense*	3	*Elops saurus*	3	*Symphurus plagiusa*	2
Lagodon rhomboides	3	*Brevoortia patronus*	3	*Chaetodipterus faber*	3	*Larimus fasciatus*	1
Chilomycterus schoepfi	3	***Cynoscion nebulosus***	2	*Peprilus alepidotus*	3	*Harengula jaguana*	1
Chaetodipterus faber	2	*Trinectes maculatus*	2	*Trichiurus lepturus*	2	***Menticirrhus americanus***	1
Sciaenops ocellatus	1	*Larimus fasciatus*	1	*Pomatomus saltatrix*	2	*Stellifer lanceolatus*	1
Trinectes maculatus	1	*Stellifer lanceolatus*	1	*Mugil cephlaus*	1	*Peprilus alepidotus*	1
Prionotus scitulus	1	*Trichiurus lepturus*	1	*Lepisosteus osseus*	1	*Chasmodes saburrae*	1
		Achirus lineatus	1				
		Ophidion welshi	1				
		Prionotus scitulus	1				

Bold Type - recreational species

Round Island Reef

Geographic location has played an important role in the abundance and sustainability of these reefs. The Round Island reef site was highly susceptible to various abiotic factors including weather, wave action, and ocean currents. It is the deepest of the four reefs, with an average depth of 2.43 meters. This reef is more open to the Gulf of Mexico than the others. It has produced a larger variety of fishes, including *R. canadum*, and various shark species. This area has been noted as a possible shark nursery ground. Sampling has shown that during the summer months, a large abundance of sharks can be found on or in very close proximity to this reef area. Since this site seems to be heavily utilized by sport and commercial fishermen, it has been noted that during the months of shrimp season, fish abundance decreased. Total capture of finfish from sampling during this high use period reflects this decrease in abundance.

East Deer Island Reef

The East Deer Island sample site is another heavily used fishing area utilized by recreational, commercial and charter boat interests. It has been noted in sampling that during times of heavy boat traffic in the Biloxi Ship Channel, fish were sparse in numbers. Through sampling efforts, it has been noted that excessive sedimentation due to Hurricane Georges in September 1998 has had a detrimental effect on this area. The dredging of the ship channel has also impacted the site by increasing foreign sediment deposition. The ship channel runs approximately one-half of a kilometer away from the site.

Hancock County Reef and Long Beach Reef

The remaining two reefs are protected from some of the aforementioned factors. These two reefs, Hancock County reef, and Long Beach reef are located in very close proximity to the beach or shoreline areas. Both are located approximately 300 meters from the shore. These reefs, combined, have supported the greatest number of recreationally important species.

ACKNOWLEDGEMENTS

We are greatly indebted to the Mississippi Department of Marine Resources for funding this project.

LITERATURE CITED

Fabi, G., and L. Fiorentini. 1994. Comparison between an artificial reef and a control site in the adriatic sea: analysis of four years of monitoring. *Bull. Mar. Sci.* 55(2-3):538-558.

Pratt, J.R. 1994. Artificial habitats and ecosystem restoration: managing for the future. Bull. Mar. Sci. 55:268-275

Seaman, W., and L.M. Sprague. 1991. *Artificial habitats for marine and freshwater fisheries.* Academic Press, San Diego, CA.

Seaman, W., R.M.Buckley, and J.J. Polovina. 1989. Advances in knowledge and priorities for research, technology and management related to artificial aquatic habitats. *Bull.Mar.Sci.* **44**:527-532.

Stroud, R. H. 1965. Artificial reefs as a tool of sport fishery management in coastal marine waters. Proceedings of International Game Fish Conference, Miami, Fla. November 12, 1965

Efficacy of Two- and Three-Chamber Light-traps for Presettlement Fishes and Invertebrate Plankton from Mangrove and Coral Reef Habitats at Key Largo, Florida

DAVID G. LINDQUIST[1], FRANK J. HERNANDEZ, Jr.[2], ILEANA E. CLAVIJO[1] and MAUREEN E. WHITTAKER[1]
[1]Department of Biological Sciences
and Center for Marine Science Research
University of North Carolina at Wilmington
Wilmington, NC 28403 USA
[2]Department of Oceanography and Coastal Sciences
Louisiana State University
Baton Rouge, LA 70803 USA

ABSTRACT

Light-traps have been used in ichthyo- and invertebrate plankton studies to sample the larval and juvenile stages of fishes and planktonic invertebrates that are often inadequately sampled with conventional nets. Light-traps can be used in shallow areas where the use of towed nets is difficult, such as in mangroves and over coral reefs. Our purpose was to field test and compare the efficacy of two- and three-chamber light-traps in these two habitats. Three-chamber traps significantly out-performed two-chamber traps in both the mangroves and over the reef, collecting 7 and 9.5 times the number of fish larvae per hour, respectively. The three-chamber traps also sampled a greater diversity and abundance of invertebrate plankton.

KEY WORDS: Ichthyoplankton, invertebrate plankton, light attraction, sampling gear

INTRODUCTION

Light-traps have been used recently in ichthyoplankton studies to sample the older, post-flexion pelagic stages and presettlement juveniles that are not sampled adequately with conventional net tows (Brogan 1994, Choat et al. 1993, Hernandez and Lindquist 1999). The traps have demonstrated certain biases, including being size and taxon selective (Doherty 1987), but they do offer several advantages as a sampling device. They can be used in shallow areas where the use of towed nets can be difficult (Brogan 1994). Larvae are collected live and can be placed in aquaria for further studies and identification purposes (Doherty 1987) or released by divers and tracked for behavioral analyses (Leis et al. 1998, 1999). Several designs, all modeled after Doherty's (1987) design, have been used extensively around different reef systems, particularly the Great Barrier Reef, and have demonstrated the ability to sample individuals from many different families. However, light-traps have not specifically been used to assess the relative abundance of all captured marine invertebrate plankton (see Thorrold 1992 and Moltschaniwskyj and Doherty 1994, 1995, and Kraemer 1996 for juvenile squid catches). In May, 1994, we were able to collect zooplankton using two different light-trap designs in Key Largo, Florida in a mangrove lagoon and above a coral reef. Our purpose was to field test and compare both designs in terms of their efficiency as presettlement larval fish and invertebrate plankton samplers.

METHODS and MATERIALS

Samples were collected from a shallow (< 2 m deep) mangrove lagoon channel located near an inlet on the southwest end of Largo Sound (25°6.5'N, 80°24.2'W) on May 9 - 11, 1994, and from Three Sisters, a reef (< 7 m deep) located approximately 7.5 km from shore (25°1.6'N, 80°23.7'W), on May 12, 1994 (Figure1).

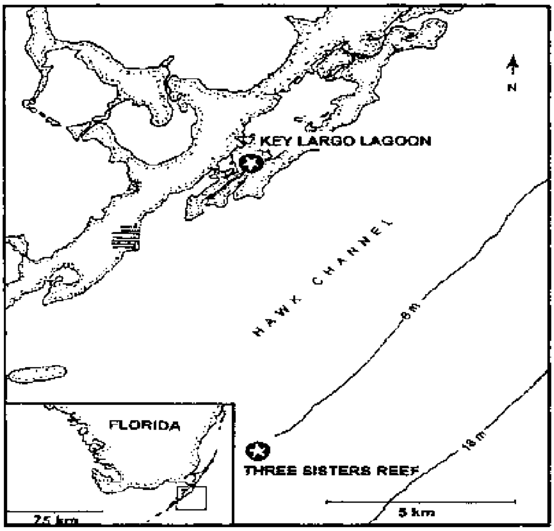


Figure 1. Chart of study area showing the two study sites. Depth contours shown are 6 and 18 m.

Two different light-trap designs were utilized to collect fish larvae and presettlement juveniles. One design, constructed by Progressive Technologies, Inc. (PTI) in Plantation, Florida, and modeled after Doherty (1987), had three chambers and a timer, which alternately turned the bulbs in the two upper chambers on and off, drawing organisms into the trap. The other, simpler design, constructed at UNCW's Center for Marine Science Research (CMSR), and modeled after Brogan (1994), had only two chambers with the two 8 watt florescent bulbs in each chamber remained on continuously. Traps were deployed on site beginning at sunset using a 7 m outboard (*R/V Tomtate*) from UNCW's National Undersea Research Center facilities in Key Largo, Florida. Each night in the lagoon, three sets of samples were completed using 3 three-chamber models (n = 27) and 1 two-chamber model (n = 9). Above the reef, three sets were completed using 2 three-chamber light traps (n = 6) and 1 two-chamber light trap (n = 3). In the mangrove, traps were moored to the bottom with anchors. Above the reef, the traps were linked together at 50 m intervals with line and attached to a mooring. Buoys were attached to each trap above the reef, allowing them to remain in the water column near the surface. Traps were allowed to fish for approximately 1 hour before being collected. The duration that each trap fished was recorded. Upon retrieval, the contents of each trap were rinsed with seawater into a sieve. Organisms were then rinsed into a collection jar and fixed with 95% ethanol. Samples were sorted in the laboratory for larval and juvenile fishes and zooplankton. Fish larvae were counted and measured to the nearest 0.5 mm. Fish identifications were made to the lowest possible taxon using many resources, primarily Fahay (1983), Leis and Rennis (1983), and reference materials provided by Drs. A.B. Powell and R.E. Robbins at a larval fish workshop conducted at the National Marine Fisheries Service laboratory in Beaufort, North Carolina. Invertebrate plankton were identified to the lowest possible taxon and enumerated, except in eight cases where high volumes were subsampled, using a volumetric technique to extrapolate total numbers in the sample.

RESULTS

The Doherty three-chamber model significantly out-performed the Brogan two-chamber model in both the mangrove and on the reef, collecting 7 and 9.5 times the number of fish larvae/hr, respectively (Table 1). A total of 11 Doherty samples and seven Brogan samples contained no fish larvae. A total of 60 individuals representing at least five families was sampled using the Doherty model in the mangrove, while the Brogan model sampled only four individuals representing two families. Above the reef, the Doherty model collected 29 fish from at least nine families, and the Brogan design collected two individuals from two families. Overall, individuals from at least 12 families were collected, and the combined capture rate was 1.9 fish/hr. Clupeids (primarily *Jenkinsia* spp.) were the most abundant larvae collected by the Doherty trap, dominating both the mangrove (72%) and reef (55%) samples (Table 2).

Table 1. Number of fish larvae and fish juveniles sampled per hour in the mangrove lagoon and above the reef.

Location	Trap Type	Number of Fish Sampled per Hour
Mangrove	Doherty	2.1
Mangrove	Brogan	0.3
Reef	Doherty	5.7
Reef	Brogan	0.6

Table 2. Total number, size range, and percent total of fish larvae and fish juveniles sampled with light-traps in the mangrove and above the reef. **=no data.

	MANGROVE						REEF					
	DOHERTY			BROGAN			DOHERTY			BROGAN		
Family	n	SL	%	n	SL	%	n	SL	%	n	SL	%
Atherinidae	7	19 - 49.5	12	1	5.0	25	0	**	0	0	**	0
Clupeidae	44	14.5 - 57	72	3	16 - 17	75	16	4.5 - 18	55	0	**	0
Cyprinodontidae	1	7.0	2	0	**	0	0	**	0	0	**	0
Engraulidae	4	24 - 38	7	0	**	0	0	**	0	0	**	0
Gobiidae	0	**	0	0	**	0	1	4.5	3	0	**	0
Haemulidae	0	**	0	0	**	0	2	5 - 5.5	7	1	5.0	50
Labrisomidae	0	**	0	0	**	0	1	7.5	3	0	**	0
Ophidiidae	0	**	0	0	**	0	1	5.0	3	0	**	0
Pomacentridae	0	**	0	0	**	0	1	8.0	3	0	**	0
Synodontidae	0	**	0	0	**	0	2	19 - 20	7	1	35.0	50
Tetraodontidae	0	**	0	0	**	0	1	3.5	3	0	**	0
Unidentified	4	2.5 - 3	7	0	**	0	4	5 - 7.5	14	0	**	0
Totals	60	2.5 - 57.0	100	4	5.0 - 17.0	100	29	3.5 - 20	100	2	5.0 - 35.0	100

The Doherty traps sampled individuals from a larger size range (2.5 mm-57.0 mm) than the Brogan traps (5.0 mm-35.0 mm), although the number of individuals sampled with the latter model was too low for comparison. The majority of individuals sampled with both trap designs on the reef and in the mangrove lagoon were post-flexion larvae and juveniles (Table 3).

Table 3. Summary of the different stages of fish larvae and juveniles sampled with light-traps.

Site/Trap	% Preflexion	% Flexion	%Postflexion/ Juveniles
Mangrove			
Doherty	5	2	93
Brogan	25	0	75
Reef			
Doherty	17	17	66
Brogan	50	0	50

Both trap types sampled diverse invertebrate plankton, including holoplankters such as copepods, chaetognaths, and polychaetes as well as meroplankters such as stomatopods, zoea, and megalopae. The total number of individuals sampled varied for each trap type, but the three-chamber trap caught an average of four to seven times more invertebrate plankton than the two-chamber trap. Differences in total catches were observed at the two habitats. The mangrove lagoon had a mean of 768 individuals per two-camber trap and 5,842 per three-chamber trap while these numbers for the coral reef habitat were 188 and 809, respectively. The composition of invertebrate plankton also differed at the two habitats (Figure 3). Harpacticoid copepods comprised 68 percent of the mean total at the mangrove lagoon, but this is biased by a single three-chamber trap catch of 37,404 individuals. Cumaceans were the second most abundant taxon at the mangrove lagoon with nearly 19 percent of the mean total. Amphipods and calanoid copepods were the abundant taxa caught at the coral reef with 22 and 18 percent of the mean totals, respectively, for the two- and three-chamber traps.

DISCUSSION

The Brogan two-chamber model did not sample as well as the Doherty three-chamber model in terms of number of individual fish sampled, number of fish taxa sampled, and size-range of fish sampled. However, only one Brogan trap was available for use in the study, and did not get field tested as extensively as the Doherty model. Both traps appear to collect the "target" size-range they are designed to sample, the post-flexion larvae and juvenile stages. With the exception of family Cyprinodontidae, all families represented in our studies have been sampled in other light trap studies. The capture rate for the Doherty trap above the reef is slightly higher than that reported in Doherty's original study (1987), but lower than subsequent studies, which range from 11.8 - 313.5

larvae/hr. It must be noted, however, that our site was a small patch reef and not comparable to the Great Barrier Reef sites in terms of larval abundance and diversity.

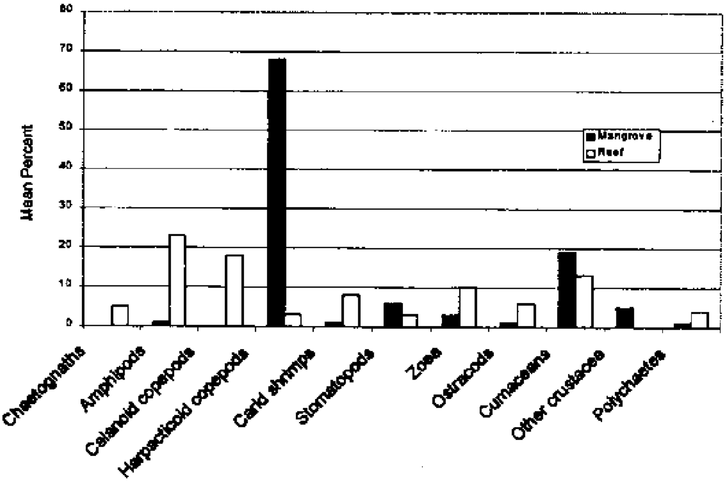


Figure 3. Comparison of major invertebrate zooplankton taxa at the two habitats captured in both two- and three-chamber light-traps.

Our results indicate that the three-chamber trap is much more efficient in the capture of invertebrate plankton than the two-chamber trap. The timer light system of the three-chamber trap helps attract and retain a greater number of individuals compared to the continuous light system of the two-chamber trap. Moreover, the three-chamber trap has twice the number of entry vents compared to the two-chamber trap (six vs. three) and thus allows a greater number of individuals to enter the upper collecting chamber. The main disadvantages of the three-chamber trap are cost ($3,000 in 1994) as well as greater bulk and weight. Both light-trap designs sampled the diversity of invertebrate plankters almost equally. Only cyclopoid copepods were entirely absent in the two-chamber trap but they were rare in the three chambered trap, as well.

Differences in the composition and abundance of invertebrate plankters at the two sites were observed in our study. These differences may be due to distributional patterns in planktonic taxa that may be controlled by prevailing current patterns at the two sites. Differences may also be due to biological factors since some organisms migrate at night from the soft substrates of the mangrove sediments to the water column (e.g., harpacticoid copepods) and are thus differentially available depending upon temporal and spatial activity patterns.

ACKNOWLEDGEMENTS

We gratefully acknowledge NOAA's National Undersea Research Center at UNCW for providing facilities and logistic support for the field work at Key Largo. Funding was provided by NOAA's Cooperative Institute for Fishery Oceanography and a UNCW Faculty Development grant. D. Aten at CMSR was instrumental in the designing and manufacturing of the two-chamber light-trap. M. Boris, F. Chapman, A. Henrikson, F. Leyva, B. Spudich, and D. Swearingen helped with the field collections. Dr. A. McCrary kindly helped with the invertebrate zooplankton identifications. This paper is contribution No. 125/6, Center for Marine Science Research, University of North Carolina at Wilmington.

LITERATURE CITED

Brogan, M.W. 1994. Two methods of sampling fish larvae over reefs: a comparison from the Gulf of California. *Mar. Biol.* **118**:33-44

Choat, J.H., Doherty, P.J., Kerrigan, B., Leis, J.M. 1993. A comparison of towed nets, purse seine, and light-aggregation devices for sampling larvae and pelagic juveniles of coral reef fishes. *Fish. Bull. U.S.* **91**:195-209

Doherty, P.J. 1987. Light-traps: selective but useful devices for quantifying the distributions and abundances of larval fishes. *Bull. Mar. Sci.* **41**:423-431

Fahay, M.P. 1983. Guide to the early stages of marine fishes occurring in the Western North Atlantic Ocean, Cape Hatteras to the Southern Scotian Shelf. *J. Northw. Atl. Fish. Sci.* **4**: 423 pp.

Hernandez, F.J., Jr. and D.G. Lindquist. 1999. A comparison of two light-trap designs for sampling larval and presettlement juvenile fish above a reef in southern Onslow Bay, North Carolina. *Bull. Mar. Sci.* **64**:173-184.

Kraemer, J.J. 1996. Size frequency and light trap catch efficiencies of the squid, *Loligo* sp., during the hatchling and juvenile stages in the coastal waters of Onslow Bay, North Carolina. Honors thesis, University of North Carolina at Wilmington, 30 pp.

Leis, J.M., Rennis, D.S. 1983. *The Larvae of Indo-Pacific Reef Fishes*. University of Hawaii Press, Honolulu. 371 pp.

Leis, J.M. and B.M. Carson-Ewart. 1998. Complex behaviour by coral-reef fish larvae in open-water and near-reef pelagic environments. *Env. Biol. Fish.* **53**:259-266.

Leis, J.M. and B.M. Carson-Ewart. 1999. *In situ* swimming and settlement behavior of larvae of an Indo-Pacific coral-reef fish, the coral trout *Plectropomus leopardus* (Pisces:Serranidae). *Mar. Biol.* **134**:51-64.

Moltschaniwskyj, N.A. and P.J. Doherty. 1994. Distribution and abundance of two juvenile tropical *Photololigo* species (Cephalopoda: Loliginidae) in the central Great Barrier Reef Lagoon. *Fish. Bull. U.S.* **92**:302-312.

Moltschaniwskyj, N.A. and P.J. Doherty. 1995. Cross-shelf distribution patterns of tropical juvenile cepahlopods sampled with light-traps. *Mar. Freshwater Res.* **46**:707-714.

Thorrold, S.R. 1992. Evaluating the performance of light-traps for sampling small fish and squid in open waters of the central Great Barrier Reef lagoon. *Mar. Ecol. Prog. Ser.* **89**:277-285

Monitoreo de Grandes Peces Pelagicos en el Mar Caribe y el Atlantico Centro-Occidental Mediante un Programa Integrado de Monitoreo en Venezuela

FREDDY AROCHA[1] and LUIS A. MARCANO[2]
[1]*Instituto Oceanográfico de Venezuela*
Universidad de Oriente
Cumaná 6101, Venezuela
[2]*Centro de Investigaciones Pesqueras*
FONAIAP
Cumaná 6101, Venezuela

RESUMEN

El monitoreo de grandes peces pelágicos en el Mar Caribe y aguas adyacentes en puertos venezolanos, comenzó en 1987 con el apoyo financiero de la Comisión Internacional para la Conservación del Atún Atlántico (CICAA) a través del Programa de Investigación para Peces de Pico (PIPP). El propósito del PIPP era y sigue siendo, determinar el estatus de los stocks de los peces de pico èn el Océano Atlántico. En los primeros años del PIPP (1987-1990), el esfuerzo se concentró en los muestreos en puerto de las capturas que se realizaban mediante la pesca deportiva que dirigía su esfuerzo hacia la captura de los peces de pico; mientras que la cobertura con observadores a bordo de la flota palangrera industrial se limitó a un promedio de 3 viajes/año resultando en un promedio de 30 lances/año. A partir de 1991, el PIPP de la CICAA fomentó un programa de observadores a bordo de embarcaciones palangreras que dirigían su esfuerzo a túnidos y pez espada. Se incrementó el esfuerzo para cubrir los puertos claves donde se desembarcaban peces de pico y pez espada capturados por la pesquería artesanal. El PIPP entrenó observadores científicos para que registraran información detallada sobre las características del arte de pesca, la posición y hora de los lances, el estatus de los especímenes capturados (vivo o muerto, embarcado, descartado o marcado), medidas de talla y peso, así como identificación del sexo de las diferentes especies. Adicionalmente, los observadores recolectaban muestras biológicas dirigidas a apoyar la investigación sobre estudios que respondieran interrogantes sobre la historia de vida de las especies (*e.g.*, edad, crecimiento, reproducción, hábitos alimenticios). También se registra la interacción con el arte de pesca de tortugas marinas, mamíferos marinos y aves marinas. A partir de 1991 se presenta un incremento sostenido en la cobertura de viajes con observadores científicos en las embarcaciones industriales de palangre, pasando de 16 viajes/año (99 lances) a 35 viajes/año (320-488 lances) a partir de 1993. Luego de siete años de existencia del PIPP con el nuevo esquema, el Programa ha registrado el cambio en la estrategia de pesca (*i.e.*, el uso de carnada viva y la eliminación de viaje-especie objetivo). La información biológica ha contribuido al conocimiento de la estrategia reproductiva del pez espada en el Atlántico centro-occidental, en la distribución del stock y biología reproductiva del dorado, a la dirección del movimiento desde y hacia Venezuela de las especies de peces de pico. Adicionalmente, se ha desarrollado una base de datos sobre la captura incidental de tiburones oceánicos capturados en los lances dirigidos al pez espada. Este tipo de información y la calidad de la misma no habría sido posible de obtener sin contar con le apoyo de un programa de monitoreo para grandes peces pelágicos como el que existe actualmente en Venezuela.

PALABRAS CLAVES: Peces de Pico, Programa de observadores, Venezuela.

Monitoring Large Pelagic Fishes in the Caribbean Sea and the Western Central Atlantic by an Integrated Monitoring Program from Venezuela

ABSTRACT

Monitoring of large pelagic fishes in the Caribbean Sea and adjacent waters from Venezuelan ports started in 1987 with the support of the International Commission for the Conservation of Atlantic Tunas (ICCAT) Enhanced Research Program for Billfish (ERPB). The purpose of the ERPB was to aid in determining the status of the billfish stocks in the western Atlantic Ocean. In the early years of the ERPB in Venezuela (1987 – 1990), effort was concentrated on port sampling and monitoring recreational fisheries targeting billfish, at-sea-observer covered trips on commercial longlines were limited to an average of 3 trips/year, comprising on average 30 sets/year. In 1991, ICCAT's ERPB sponsored a pelagic longline observer program in Venezuela on vessels targeting tuna and swordfish, and on artisanal longliners targeting billfish. Efforts were made to increase port sampling by monitoring key localities where billfish and swordfish were being landed by artisanal fishers. The program trains scientific observers to record detailed information concerning gear characteristics, position and time the gear is set and hauled, status and action of the specimens caught (alive or dead, kept, discarded or tagged), length and weight measurements, and sex identification of specimens. In addition, observers collect biological samples to support research studies directed to answer questions about the species life history (age, growth, reproduction, etc.). Interaction of sea turtles, marine mammals and sea birds is also recorded. Since 1991, the observed covered trips showed a steady increase, from 16 covered trips (99 sets) in 1991 to leveling around 35 trips/year since 1993 (320 – 488 sets). After seven years, the overall program has provided information on shifts in fishing strategy (use of live bait and elimination of target species trips). The biological data provided information on the reproductive strategy of swordfish in the western central Atlantic, information on stock distribution and reproductive biology on dolphinfish, movement direction on billfish from and to Venezuelan waters. In addition, a shark database was developed from by-catch associated with swordfish catches. None of this information would have been possible to collect without the support of the integrated monitoring program for large pelagic fishes in Venezuela.

KEY WORDS: Billfish, Observer program, Venezuela

INTRODUCCION

El monitoreo de grandes peces pelágicos en el Mar Caribe y aguas adyacentes en puertos venezolanos comienza con el Programa Expandido de Investigación en Peces de Pico, auspiciado y coordinado por la Comisión Internacional para la Conservación del Atún del Atlántico (ICCAT), en 1987. Durante los primeros años (1987-1990) del programa de monitoreo, sus funciones estaban dirigidas a los muestreos de peces de pico en el puerto pesquero de Cumaná y al monitoreo de la pesca deportiva en el área de La Guaira. El esfuerzo que se dirigía al monitoreo de la pesca deportiva en el área de La Guaira se debía a que la serie histórica más larga sobre índices de abundancia provenía del sector deportivo que pescaba en esa área. Sin embargo, a comienzo de la decada de los años 90, la situación económica en Venezuela afectó la pesca deportiva y como consecuencia, la serie histórica de índices de abundancia se vieron afectados. Si bien se continúan los esfuerzos para monitorear la pesca deportiva y mantener la serie histórica de índices de abundancia, los resultados obtenidos de la recolección de información estadística no reflejaban la situación real de las pesquerías de grandes pelágicos en aguas venezolanas. Se requería de un programa que cubriera los distintos sectores que explotaban los peces pico.

Es en 1991 cuando el Programa adquiere mayor relevancia y fortalecimiento debido a la mayor cobertura de las actividades y a la estrecha colaboración entre el Centro de Investigaciones Pesqueras del FONAIAP-Sucre (coordinador de la parte administrativa y logística) y el Instituto Oceanográfico de Venezuela de la Universidad de Oriente (coordinador de la parte científica y calidad de información). La amplición de las actividades estaban dirigidas a todas las pesquerías que capturaban peces de pico en el Caribe venezolano y Atlántico centro-occidental. Entre las que se incluía, además de la pesca deportiva, a la pesca palangrera industrial que dirigía su esfuerzo a los atunes, y a las pesquerías artesanales (tanto de palangre como de redes de ahorque) que dirigían su esfuerzo hacia la captura estacional de peces de pico. También se incluyó en el Programa, el monitoreo de palangreros que dirigían su esfuerzo a la pesca del pez espada, dada la importancia que estaba adquiriendo este recurso a finales de los 80 y comienzos de los 90. Este tipo de monitoreo, tanto en puertos de desembarque como con observadores científicos embarcados, dió la oportunidad para monitorear la captura, efectuar mediciones y tomar muestras biológicas de otras especies pelágicas de importancia en las pesquerías venezolanas, como lo son los atunes, el dorado y los tiburones, entre otras.

A medida que el Programa fue avanzando, se incorporaron nuevas actividades para responder a interrogantes científicas sobre algunas especies que son capturadas por las pesquerías que dirigen su esfuerzo a grandes peces pelágicos.

En este documento se presentan los resultados de las actividades de la pesca comercial monitoreada por el Programa durante el período comprendido entre 1991 y 1998.

METODOS

La información obtenida por el Programa procede del monitoreo de tres tipos de pesquerías que operan en áreas comunes según la especie objetivo. La información que se obtiene de la pesquería de palangre industrial, proviene de observadores científicos embarcados en barcos que dirigen su esfuerzo hacia la captura de atún o pez espada. La mayoría de estas embarcaciones suelen ser de 20 – 24 m de eslora que utilizan palangres de 30 - 50 km de largo y pescan de 600 – 900 anzuelos. También hay cuatro embarcaciones de tipo coreano, que son de 27 - 29 m de eslora que utilizan palangres de 50-120 km de longitud y pescan de 500 – 1.800 anzuelos. Estas últimas, dirigen sus operaciones de pesca principalmente hacia el área exterior del arco de las Antillas y sur de Puerto Rico (Figura 1). Mientras que el resto de las embaiciones de palangre industrial opera alrededor del cinturón de islas oceánicas venezolanas, norte de la Isla de Margarita y al noreste de las Guayanas (Figura 1).

Otra información que recoge el Programa es la procedente del monitoreo de las pesquerías artesanales que dirigen su esfuerzo hacia la captura de peces pico de manera estacional. La información que recibe el programa de estas pesquerías proviene de dos fuentes: la pesquería artesanal con redes de ahorque que opera en el área al norte de La Guaira y la pesquería artesanal de palangre que opera mayormente en el corredor sur de las islas oceánicas venezolanas (Figura 1). La información de la pesquería artesanal con redes de ahorque que opera al norte de La Guiara, se recoge por el monitoreo de los desembarques en su puerto base, que es la localidad de Playa Verde. Mientras que la información que proviene de la pesquería artesanal de palangre se recoge en la localidad de Juangriego – Isla de Margarita con monitoreos de los desembarques en las diferentes rancherías que compran la pesca. En cuanto al arte utilizado por estas pesquerías artesanales y modo de operación se tiene lo siguiente: Las redes de ahorque que se emplean en la pesqueria artesanal que opera en La Guaira son de 1.000 a 1.200 m de longitud y 10 m de alto, con una luz de malla de 14 cm, las cuales se operan desde embarcaciones de 8 a 13 m de eslora con dos moters fuera de borda (maxima potencia 120 hp), este tipo de embarciones realizan una operación de pesca por noche; mientras que las embarcaciones con palangre artesanal realizan operaciones que duran 8 - 12 días, con un palangre de 12 - 15 km de longitud y pescan 450 - 600 anzuelos. Este tipo de embarcaciones son de madera que miden entre 7 y 14 m de eslora.

Los datos utilizados en el presente trabajo provienen de la información tomada por observadores científicos a bordo de embarcaciones palangreras industriales y artesanales y de los muestreos realizados en los puertos pesqueros de Cumaná, Playa Verde (área de La Guaira) y Juangriego (Isla de Margarita). Las medidas morfométricas tomadas de las diferentes especies siguen los patrones estándar del manual de campo de la Comisión Internacional para la Conservación del Atún Atlántico (Miyake 1990).

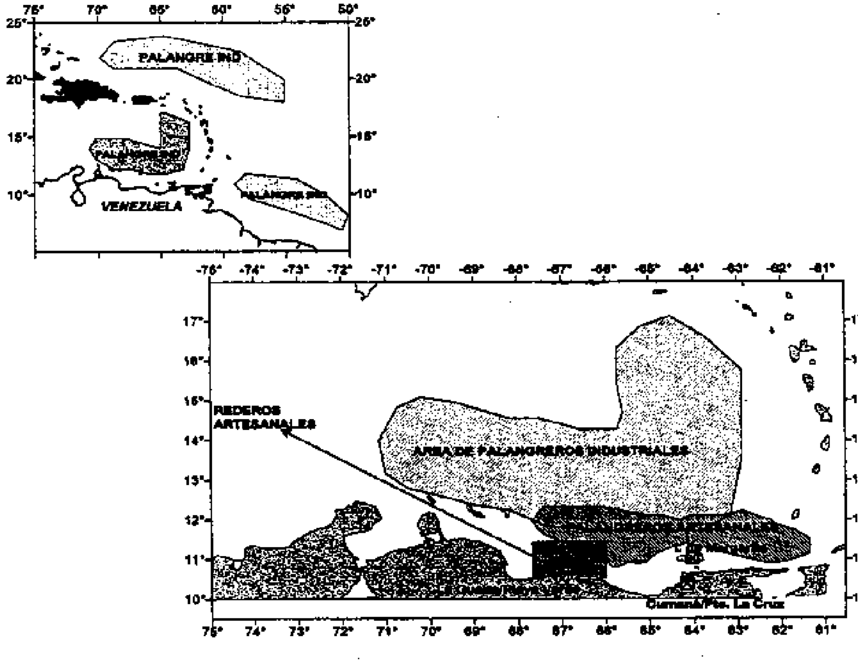


Figura 1. Area de distribución de la pesca de grandes peces pelágicos. APM es el Area de Protección Marina alrededor de La Guiara.

RESULTADOS Y DISCUSION

El número de viajes cubierto por observadores científicos en palangreros industriales se duplicó a partir del segundo año del programa (1992), pasando de 15 viajes para 1991 a 30 en 1992 (Figura 2). A partir de 1993, el número de viajes cubierto por observadores se encuentra entre 32 y 35, con la excepción de 1994 cuando la cobertura de viajes bajó a 22 viajes. La cobertura de viajes por observadores científicos con respecto al total de los viajes realizados por la flota palangrera industrial ha fluctuado entre un 8,1% en 1991 y un 18,7% para 1998 (Tabla 1). En líneas generales la cobertura promedio ha estado alrededor de un 13%. La duplicación del número de viajes de 1991 a 1992, así como la baja cobertura, se debió a que el programa comenzó en mayo de 1991, lo que implicaba que solo se registraron la mitad del número de viajes (15) de lo que se cubriría e un año completo (30 viajes). Sin embargo, la abrupta caída en la cobertura de viajes en 1994 se originó por la falta de disponibilidad de embarcaciones, ya que durante ese año hubo embarcaciones que dejaron de pescar en aguas venezolanas o fueron transformadas para otras actividades. En los años siguientes a 1994 se ha logrado cubrir entre 2 y 5 viajes por encima del estimado promedio de 30 viajes/año.

En cuanto al número de lances observados por viaje durante el período de cobertura, los mismos han ido en aumento desde que se amplió la cobertura. En promedio, el número de lances por año supera los 300 lances (Figura 2). Sin embargo, se observa que en los años 1993, 1995 y 1998, el numero de lances superó la cifra de 400 lances/año. Ello se debió a la cobertura de cuatro viajes anuales en embarcaciones de tipo coreano que pescan al noreste del arco de las Antillas Menores, realizando viajes de 22-29 días con desembarques y re-abastecimiento en la isla de St. Martin.

Tabla 1. Cobertura de viajes con observadores científicos en la pesca industrial con palangre durante el periodo de 1991 a 1998.

	1991	1992	1993	1994	1995	1996	1997	1998
Viajes observados	17	32	34	22	37	35	37	43
Viajes totales	209	197	239	252	298	299	279	230
% observado	8.1	16.2	14.2	8.7	12.4	11.7	13.3	18.7

La presencia de observadores científicos a bordo de embarcaciones comerciales ofrece la oportunidad de registrar la información necesaria para monitorear la pesquería y los stocks de las diferentes especie explotadas comercialmente. La información registrada en planillas científicas para tal fin (Miyake 1990) provee a los científicos de información básica sobre la configuración del arte y la información sobre las especies comerciales y no comerciales capturadas en cada lance. La información que se registra sobre las especies capturadas por los observadores a bordo son sus medidas de longitud, sexo y su condición de vivo o muerto al momento del embarque. Mientras que la información que se recoge en los muestreos en puerto son básicamente las medidas de longitud que se tomen, ya que la totalidad de los ejemplares desembarcados vienen eviscerados y en muchas ocasiones sin cabeza.

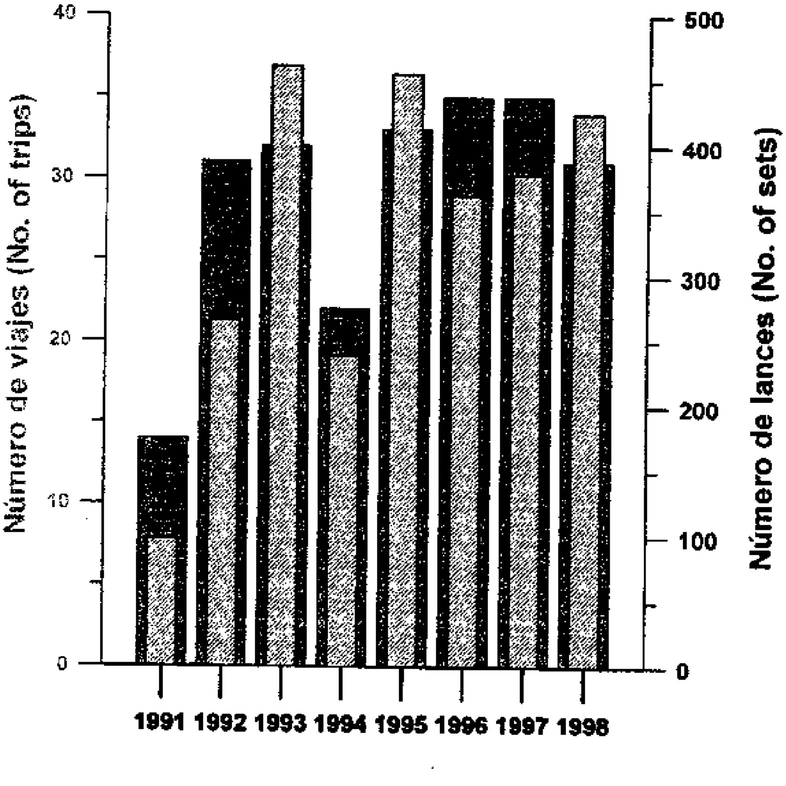


Figura 2. Número de viajes y número de lances controlados por observadores científicos del PIPP durante el período de 1991-1998.

Durante el período examinado se han podido observar diversas especies que son capturadas por estas pesquerías, mas solo se han registrado las que presentan valor comercial al momento de los desembarques y de su volumen en las capturas. En total se han registrado 12 especies y un grupo de especies (tiburones) que merecerá una explicación mas adelante. Dentro de las 12 especies más observadas, solo cuatro conforman un poco más del 80% del total de las especies observadas (Figura 3). La composición porcentual en numero observado por especie del grupo más importante, presenta al atún blanco o albacora (*Thunnus alalunga*-ALB) como la especie más observada (37.7%) durante 1991-1998 por las pesquerías de grandes peces pelágicos. Le sigue en importancia el atún aleta amarilla (*Thunnus albacares*-YFT) con un 22.4%, mientras que la otra especie de atún de gran valor comercial, como lo es el atún ojo gordo (*Thunnus obesus*-BET), se presenta en un cuarto lugar con un 10.7%. El pez espada (SWO) fue la especie que se observó más después del atún aleta amarilla, con un 13.2%. Entre las especies de menor importancia y que se pueden considerar como capturas secundarias están los peces de pico con un 9.8% (aguja azul-BUM con 2.4%, aguja blanca-WHM con 4.0%, pez vela-SAI con 2.0% y la aguja corta o pez lanza-SPF con 1.4%), seguido por el grupo de los tiburones (SHK) que conforman un 3% del total observado. El resto de las especies, entre las que se incluyen al dorado (DOL), el peto (WAH), el atún aleta negra (*Thunnus atlanticus*-BFT) y el listado (*Katsuwonus pelamis*-SJK), conforman el porcentaje restante observado (3.2%).

Si bien la albacora aparece como la especie más observada en número de ejemplares, no es la más importante en las capturas ni en los desembarques. Ello se debe a un artificio del sistema de muestreo originado por las observaciones en los palangreros de tipo coreano. Como se ha explicado anteriormente, el número de lances en estas embarcaciones supera notablemente al de los palangreros más comunes, a la vez que supera también el número de días observados, así como el número de anzuelos pescados. Es el atún aleta amarilla la especie de mayor importancia comercial con un 70% en los desembarques en peso de la flota palangrera, mientras que los desembarques de albacora no sobrepasan el 2% (SARPA y FONAIAP 1999). Sin embargo, es de hacer notar que las capturas de albacora registradas por las embarcaciones de tipo coreano con bandera venezolana no son desembarcadas en su totalidad en Venezuela, ya que la mayoría de las capturas se desembarcan en las islas antillanas.

Dado el interés que ha suscitado en el ámbito mundial las pesquerías de tiburones, a partir de 1994 el Programa comenzó a registrar las especies de tiburones capturadas como pesca incidental en las pesquerías de palangre industrial monitoreadas por observadores científicos. Se pudo observar que la mayor proporción (61.4%) la conforman tres especies, el tiburón azul (*Prionace* *glauca*), el tiburón macuira (*Carcharhinus limbatus*) y el tiburón bobo (*Cracharhinus porosus*) (Figura 4). El segundo grupo de especies en importancia (19.2%) le conforman tres especies, el tiburón cornuda (*Sphyrna zygaena*), el tiburón gran martillo (*Sphyrna mokarran*) y el tiburón mako (*Isurus oxyrinchus*). El resto de las especies de tiburones observadas no superan el 4% según se desprende de la Figura 4. En años recientes, el registro de tiburones por parte de observadores científicos a descendido debido a la disminución en la cobertura de viajes cuyo objetivo de pesca era el pez espada, ya que era en esos viajes cuando se incrementaban las observaciones de tiburones. Este descenso en la cobertura de viajes con objetivo la pesca del pez espada se debe a que la flota palangrera industrial actualmente dirige más su esfuerzo hacia la captura de atunes.

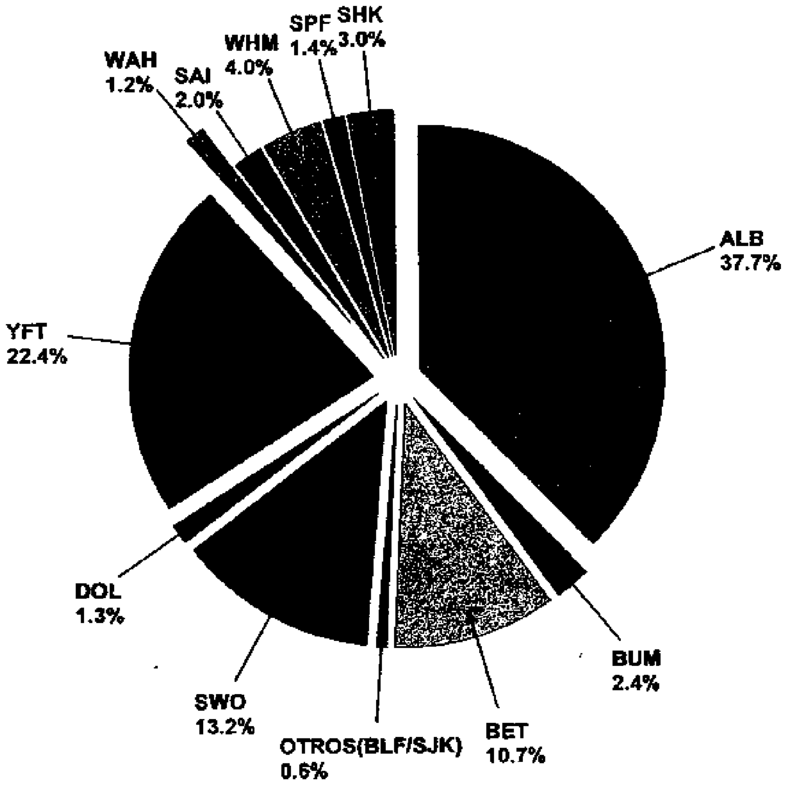


Figura 3. Composición porcentual por especies capturadas (en número) por palangreros industriales Venezolanos en el Caribe y el Atlántico centro-occidental.

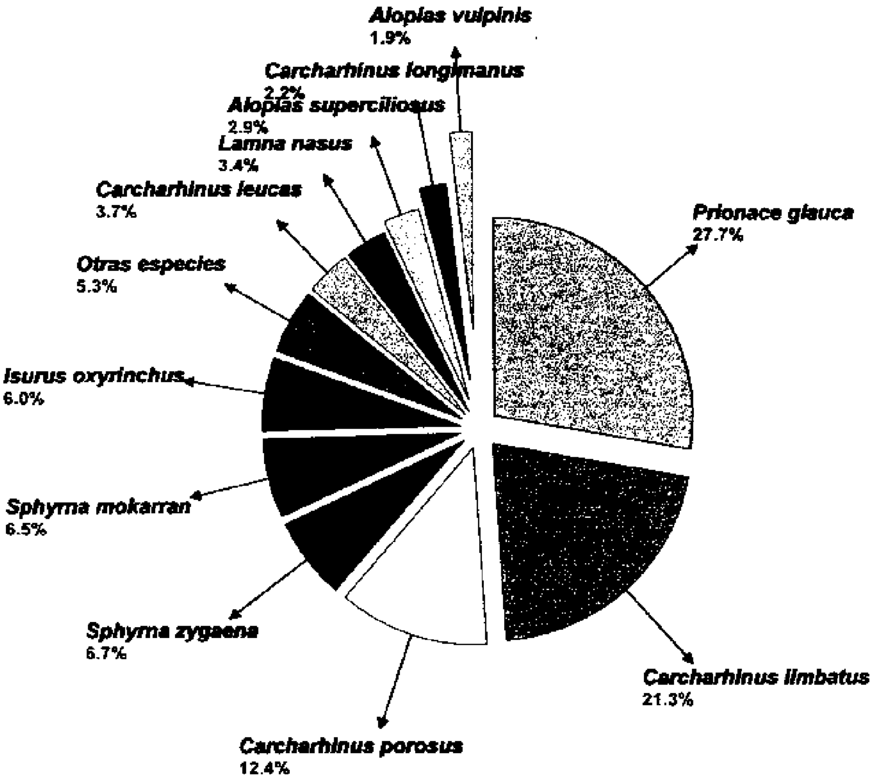


Figura 4. Proporción porcentual de especies de tiburones capturados por embarcaciones palangreras venezolanas el Caribe y el Atlántico centro-occidental.

Una información valiosa que genera el Programa son las mediciones de tallas de las diferentes especies observadas. Elto permite conocer la estructura de tallas de la porción de la población que se captura en el Caribe y en el Atlántico centro-occidental. Comenzando con las especies de atunes explotadas por la pesquería palangrera industrial, se observa que la distribución de tallas en el atún ojo gordo presenta una distribución con tres modas, una muy pronunciada alrededor de los 130 cm LH y otras dos menos pronunciadas alrededor de los 110 cm LH y 75 cm LH (Figura 5). Mientras que las otras dos especies de atún, la albacora y el aleta amarilla, presentan una distribución de tallas unimodal. Una posible explicación para que se observen varias modas en la distribución de tallas del atún ojo gordo es que esta especie se captura de manera incidental por las pescas dirigidas al atún aleta amarilla y las dirigidas a la albacora por los barcos de tipo coreano. Esto implica que la configuración del arte, que es distinto para la pesca de aleta amarilla y albacora, afecta la capturabilidad de juveniles y adultos medianos, originando así una distribución de tallas con varias modas. En el caso de la estructura de tallas observadas de la albacora y el aleta amarilla nos indica que los ejemplares capturados son mayormente adultos maduros que son capturados en áreas de reproducción (Bard 1981, Arocha et al. 1999), ya que en ambos casos la distribución de tallas es la de ejemplares sexualmente maduros.

El pez espada es la especie más importante después de los atunes, se captura por los palangreros industriales y por la pesquería artesanal con redes de ahorque que opera al norte del área de La Guiara. En ambas pesquerías se observa una distribución de tallas unimodal (Figura 6), la moda se encuentra alrededor de los 115 –120 cm MILH, la cual está por debajo de la talla mínima recomendada por la CICAA desde 1991 (que es de 125 cm MILH). Ello implica que la mayoría de las capturas de pez espada en Venezuela estén por debajo de la talla mínima permitida. Esto trajo como consecuencia que los viajes dirigidos a la pesca del pez espada disminuyeran drásticamente a partir de 1993 cuando las exportaciones de pez espada se vieron severamente afectadas por no cumplir con las tallas mínimas.

Los peces de pico conforman una parte importante en las capturas incidentales de las pesquerías palangreras industriales, pero en Venezuela son objeto también de una pesquería dirigida por parte de dos tipos de pesquerías artesanales, la de redes de ahorque y palangre. En este grupo de especies, conformado por la aguja azul, la aguja blanca, el pez vela y el pez lanza, se observa que la estructura de tallas varia según la pesquería y el área donde se capturen las distintas especies.

Así se observa que la aguja azul (Figura 7), es más frecuente en las capturas de la pesca artesanal con redes de ahorque y presenta una distribución de tallas con una moda alrededor de los 200 cm MILH; mientras que en los palangreros artesanales es poco comun. Sin embargo, en los palangreros industriales aparece con marcada frecuencia, pero con una estructura de tallas con una moda inferior (190 cm MILH) a la de las pesquerías con redes. Es muy posible que ello se deba al área de pesca de las diferentes pesquerías. La artesanal con redes de ahorque están ubicadas alrededor de un monte submarino conocido como el "Placer de la Guaira", que muy conocido por la frecuencia de especies de peces pico en el área. Mientras que las otras dos pesquerías están principalmente fuera del área de La Guaira. La aguja blanca, presenta una mayor frecuencia en las pesquerías artesanales palangreras (Figura 8); mientras que las otras dos pesquerias la frecuancia es similar. Se observa que la distribución de tallas en las pesquerfas artesanales es muy similar, pero distintas a la de la pesquería de palangre industrial. Una explicación podría ser que las pesquerías artesanales operan mas cerca de las islas oceánicas y montes submarinos que la pesquería de palangre industrial, lo cual pudiera ser que solo capturasen un grupo de individuos de tallas similar.

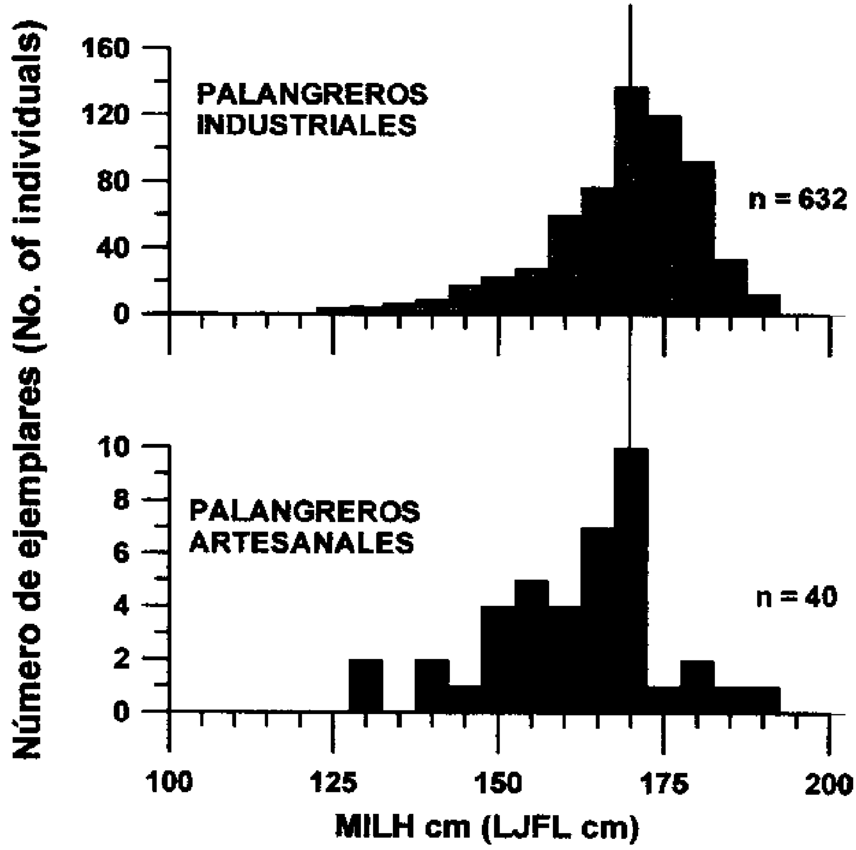


Figura 5. Distribución de frecuencia de tallas del atún ojo gordo (superior), el atún albacora (centro) y el atún aleta amarilla (inferior) procedentes de embarcaciones palangreras industriales venezolanas.

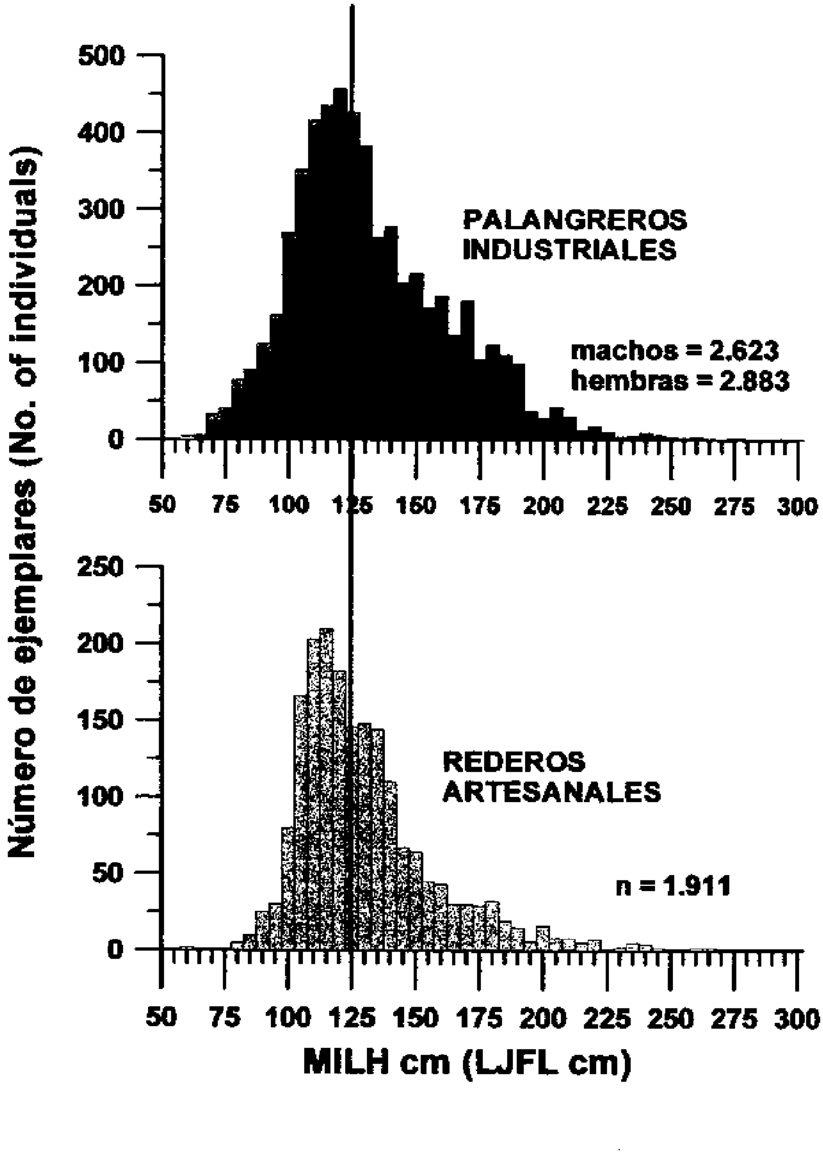


Figura 6. Distribución de frecuencia de tallas del pez espada pescado por embarcaciones palangreras industriales venezolanas (superior) y por embarcaciones artesanales con redes de ahorque (inferior). La línea vertical indica la talla mínima de captura para el pez espada del Atlántico.

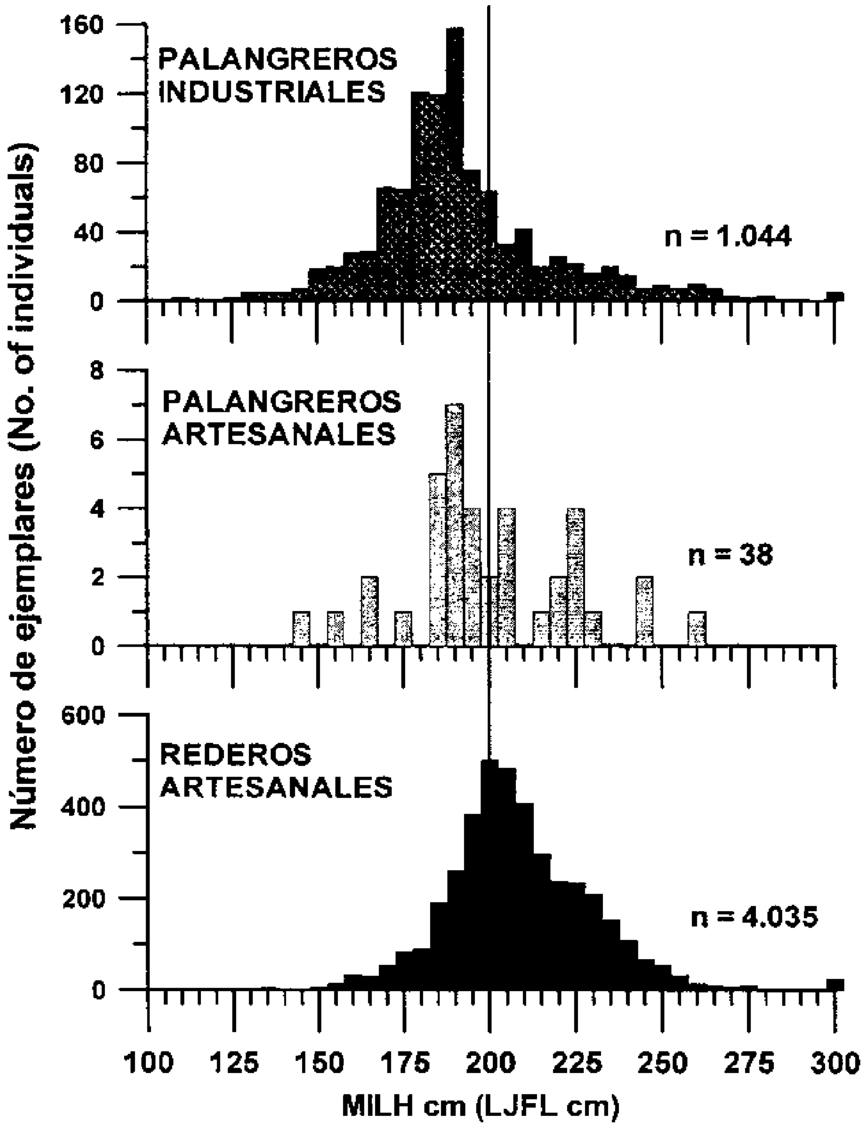


Figura 7. Distribución de frecuencia de tallas de la aguja azul pescada por embarcaciones palangreras industriales venezolanas (superior) por palangreros artesanales (centro) y por embarcaciones artesanales con redes de ahorque (inferior). La línea vertical indica la moda de 200 cm MILH.

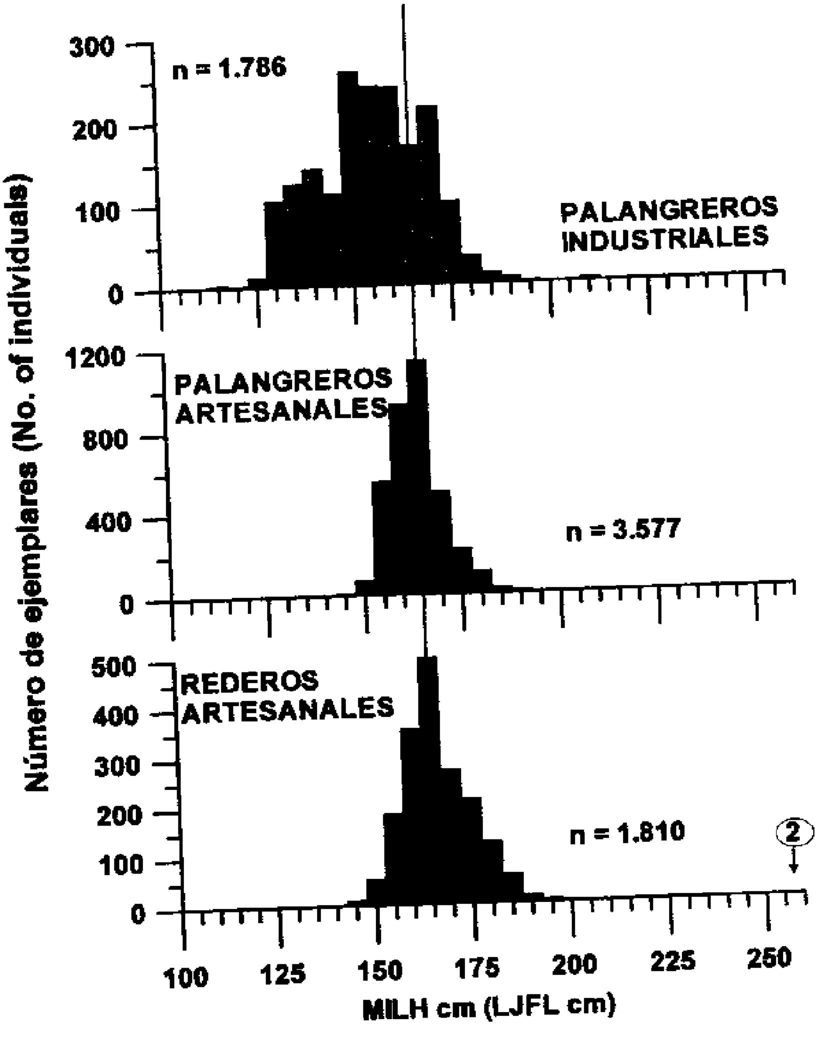


Figura 8. Distribución de frecuencia de tallas de la aguja blanca pescada por embarcaciones palangreras industriales venezolanas (superior) por palangreros artesanales (centro) y por embarcaciones artesanales con redes de ahorque (inferior). La línea vertical indica la moda de 165 cm MILH.

El pez vela es la especie más común capturada por la pesquería de redes de ahorque en el área de La Guaira (Figura 9), seguida por la pesca artesanal de palangre según se desprende de los monitoreos de esta especie realizados por el Programa. Su distribución de tallas parece uniforme en las tres pesquerías, lo cual se debe a su carácter más costero con respecto a las agujas. En cambio el pez lanza se presenta principalmente en los palangreros industriales según el numero de ejemplares observados y medidos (Figura 10), mientras que en las pesquerías artesanales solo aparece en la de palangre. Ello se debe a que esta especie es la más oceánica de todos los peces pico.

Otra especie que ha sido observada con detenimiento recientemente es el dorado (*Coryphaena hippurus*), debido a su importancia en los desembarques en años recientes. El dorado capturado por el palangre presenta una estructura de tallas diferente al dorado capturado por redes de ahorque (Figura 11). En redes de ahorque, la distribución de tallas esta sesgada hacia las tallas mayores (110 - 120 cm LH), mientras que en el palangre la distribución es mas o menos uniforme. Sin embargo, se presentan unos picos marcados entre las tallas de ambos sexos para las distintas pesquerías, lo cual según Arocha et al. (1999) obedece a movimientos estacionales de dos grupos de dorados que ocurren en temporadas distintas en aguas del Caribe venezolano.

Entre otras actividades registradas por el Programa estan notas sobre la captura incidental de tortugas, mamíferos y aves marinas. Este tipo de información aun no ha sido analizada cuantitativamente; sin embargo una apreciación cualitativa según la información de los observadores científicos es que el 95% de las capturas incidentales de tortugas marinas son de la tortuga cardón (*Dermochelys coriacea*), la cual siempre se enreda con el rendal. Aun no ha habido observaciones de tortugas con un anzuelo tragado. Se puede considerar que la incidencia de tortugas en los palangres que pescan en aguas del Caribe venezolano es baja. Aun mas baja es la incidencia de mamíferos y aves marinas en los palangres, solo se han registrado dos enredos con rendales de delfines y otros pocos de aves marinas en todo el período de monitoreo del Programa (1991 - 1998).

El Programa también ha contribuido a la recolección de material biológico, entre los cuales se destacan, la recolección de gónadas de pez espada, dorado y atún aleta amarilla para estudios de dinámica reproductiva, la recolección de estructuras óseas de peces pico y pez espada para estudios de crecimiento y determinación de edad, el marcado y liberado de ejemplares juveniles (<125 cm MILH) de pez espada para estudios de movimiento migratorio. El resultado de la información obtenida de este Programa ha servido para responder algunas interrogantes sobre la dinámica reproductiva en el pez espada (Arocha,1997), para estimar y validar las estimaciones de edad del pez espada en el Atlántico noroccidental a partir de estructuras óseas (Ehrhardt et al. 1996), para proponer una estrategia reproductiva en el dorado basado en movimientos de esta especie en el Caribe venezolano y su desarrollo gonadal (Arocha et al. 1999a), y para establecer una temporada de desove del atún aleta amarilla en el Caribe y Golfo de México (Arocha et al. 1999b).

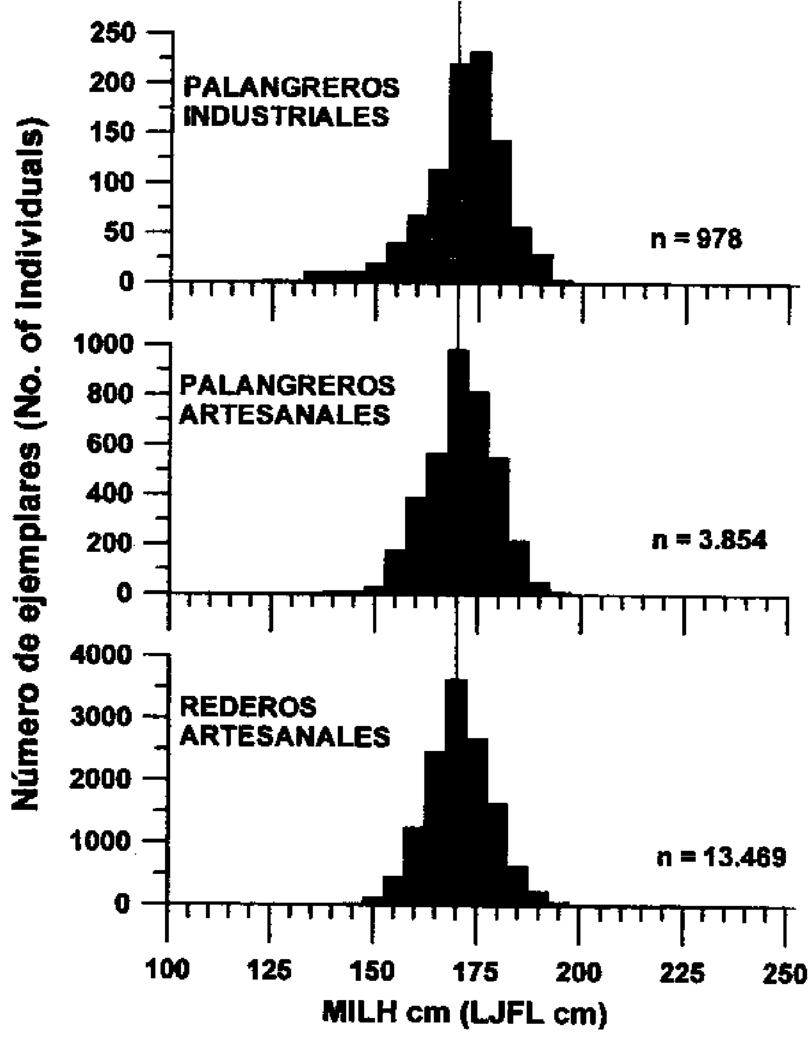


Figura 9. Distribución de frecuencia de tallas del pez vela pescado por embarcaciones palangreras industriales venezolanas (superior) por palangreros artesanales (centro) y por embarcaciones artesanales con redes de ahorque (inferior). La línea vertical indica la moda de 170 cm MILH.

También se ha contribuido con la recolección de tejido biológico de peces de pico, pez espada, peto y dorado para estudios de diferenciación de stocks entre áreas geográficas mediante el análisis de variabilidad genética. Hasta la fecha, solo se conocen resultados sobre la estructura del stock del pez espada del Atlántico, cuyos estudios indican que existe una separación de unidades reproductivas genéticas entre el Atlántico noroccidental y el Atlántico sur (Alvarado Bremer et al. 1999).

Los resultados obtenidos por el Programa y presentados en este trabajo han ido mas allá de las expectativas de monitorear los peces de pico para obtener índices de abundancia más confiables de los obtenidos hasta 1990. Ello obedeció al esfuerzo entre las instituciones locales y al apoyo constate de la CICAA y otras organizaciones no gubernamentales. También al deseo de participar de algunos empresarios y pescadores interesados por el estado de los recursos que ellos explotan. Programas de este tipo se hacen cada vez más necesarios en la zona del Caribe ya que es la única manera de conseguir información confiable que permita evaluar los stocks de grandes peces pelágicos los cuales son compartidos por otros estados ribereños.

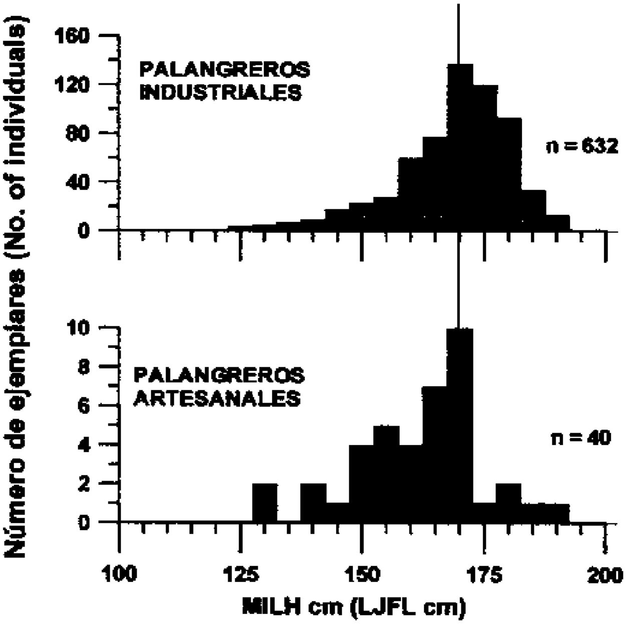


Figura 10. Distribución de frecuencia de tallas del pez lanza pescado por embarcaciones palangreras industriales venezolanas (superior) y por palangreros artesanales (inferior). La línea vertical indica la moda de 170 cm MILH.

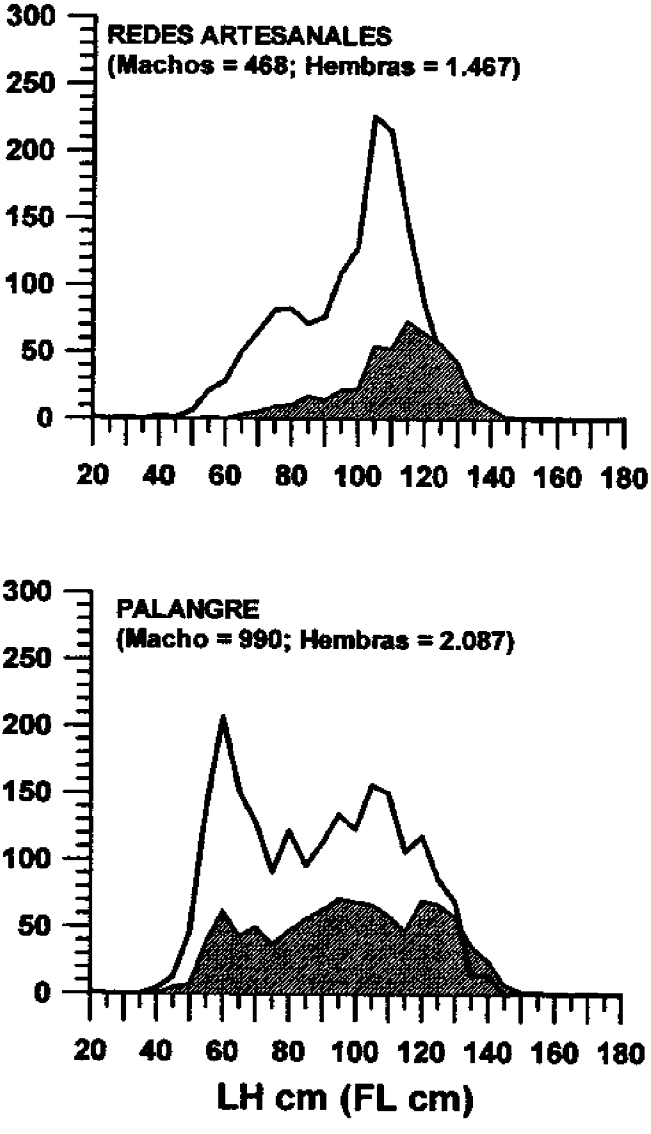


Figura 11. Distribución de frecuencia de tallas del dorado pescado por embarcaciones palangreras venezolanas y por embarcaciones artesanales con redes de ahorque. La distribución rayada indica la distribución de tallas de machos y la transparente de hembras.

LITERATURA CITADA

Alvarado Bremer, J. , J. Mejuto, J. Gomez-Marquez, J. Boan, P. Carpintero, J.M. Rodriguez, C. Plà, J.M. de la Serna y B. Ely. 1999. Heirarchial analysis of swordfish mitochondrial DNA sustructure within the Atlantic Ocean. *ICCAT Coll. Vol. Sci. Pap.* **49** (1).

Arocha, F. 1997. *The reproductive dynamics of swordfish Xiphias gladius L. and management implications in the northwestern Atlantic.* PhD Dissertation, University of Miami, Miami. 383 pp.

Arocha, F., L. Marcano, A. Larez, D. Altuve y J. Alio. 1999a. The fishery, demographic size structure and oocyte development of dolphinfish, *Coryphaena hippurus*, in Venezuela and adjacent waters. *Scientia Marina:* en prensa.

Arocha, F., D.W. Lee, L. Marcano y J. Marcano. 1999b. Preliminary studies on the spawning of yellowfin tuna, *Thunnus albacares*, in the western central Atlantic. ICCAT SCRS/99/79, 7pp.

Bard, X. 1981. Le thon germon *Thunnus alalunga* (bonnaterre, 1788) de l'océan Atlantique: de la dynamique des populations à la stratégie démographique. Thèse Doctorat des Sciences Naturelles, Université Pierre et Marie Curie, Paris VI. 330 pp.

Ehrhardt, N.M., R. Robbins y F. Arocha. 1996. Age validation and growth of swordfish, *Xiphias gladius*, in the Northwestern Atlantic. *ICCAT Coll. Vol. Sci. Pap.* **45**(2)358-367.

Miyake, M. 1990. Field manual for statistics and sampling of Atlantic tunas and tuna-like fishes. 3rd Ed. ICCAT, Madrid. 184 p.

SARPA y FONAIAP. 1999. Informe Nacional de Venezuela, 1998. ICCAT SCRS/99/139, 9 pp.

Attempts to Enhance the Visibility and Contrast of Presumed Growth Marks on Sagittal Otoliths from Wahoo, *Acanthocybium solandri*, from the Northern Gulf of Mexico and Bimini, Bahamas

JAMES S. FRANKS, JAIME L. SHEA,
NANCY J. BROWN-PETERSON, MELANIE S. GRIGGS,
and KIRSTEN M. LARSEN
The University of Southern Mississippi
Institute of Marine Sciences
Gulf Coast Research Laboratory
P.O. Box 7000
Ocean Springs, Mississippi 39566-7000 USA

ABSTRACT

In the western Atlantic Ocean wahoo, *Acanthocybium solandri*, occur from New Jersey to Columbia, including Bermuda, the Bahamas, the Caribbean Sea, and the Gulf of Mexico. Although wahoo support recreational and commercial fisheries throughout the Western Central Atlantic, adequate life history data, including estimated size-at-age, for stock assessments are lacking. Sagittal otoliths from wahoo (935 - 1,803 mm FL) caught in the recreational fishery from the northern Gulf of Mexico during May - September 1997 and 1998 and at Bimini, Bahamas in November 1997 and 1998 were examined to determine their potential use for age estimation. Whole sagittae and transverse thin-sections of sagittae viewed under transmitted and reflected light at 25 - 40x magnification generally revealed a series of ridges and faint opaque marks on the posterior portion of the distal surface of all whole sagittae and vague opaque marks on thin-sections from some sagittae. Attempts to enhance the visibility and contrast of the vague opaque marks (presumed growth zones) on thin-sectioned sagittae included application of histological stains (Toluidine Blue, Eosine Y, Neutral Red, and Analine Blue), etching with 5% EDTA followed by application of histological stains, etching and staining with mixtures of 1% acetic acid and histological stains, and digestion with proteinase K buffer. Etching and staining techniques did not delineate growth zones nor did they expose any "concealed" annual marks. Distinct opaque bands were observed only on sectioned sagittae from the two largest fish in the sample prior to the application of etching and staining agents, and a 2-h application of acidified Neutral Red provided the greatest visibility and contrast between presumed growth zones on those two sagittae.

KEY WORDS: *Acanthocybium solandri*, age, otoliths

INTRODUCTION

Wahoo, *Acanthocybium solandri*, are large, oceanic, pelagic fish of the family Scombridae and are distributed worldwide in tropical and subtropical waters (Collette and Nauen 1983). Wahoo seasonally extend into temperate waters. In the south Atlantic Ocean wahoo occur from New Jersey to Columbia, including Bermuda, the Bahamas, the Caribbean Sea and the Gulf of Mexico (Robins et al. 1986). Wahoo are considered a migratory species throughout much of the Western Central Atlantic (WCA) region (Hunte and Mahon 1985), although little is actually known about its movements and seasonal migratory patterns. Rivas (1951) reported that wahoo migrate seasonally through the Florida Straits and along the Gulf Stream and are particularly abundant along the north coast of Cuba during the winter. Wahoo are caught year-round in the northern Gulf of Mexico (Gulf), but apparently are most abundant there during spring through fall. Wahoo appear to be most abundant off Bimini, Bahamas during fall through winter.

Wahoo support important commercial (Luckhurst et al. 1997) and recreational fisheries in the Gulf of Mexico, south Atlantic Ocean, and Caribbean Sea (Franks et al. In press). Wahoo are a highly prized game fish at Bimini where prestigious wahoo fishing tournaments are held annually. Wahoo are not managed in the territorial waters of the United States or the Bahamas.

Despite their importance, little is known about the age and growth of wahoo from the WCA. Hogarth (1976) examined various life history aspects of wahoo off North Carolina and reported estimated ages for his specimens based upon examination of whole sagittal otoliths. Luckhurst et al. (1997) examined the microstructure of sagittal otoliths from wahoo collected off Bermuda and reported the presence of presumed daily growth increments. Franks et al. (In press) examined dorsal fin spines from wahoo captured in the northern Gulf and at Bimini, Bahamas to determine their potential as ageing structures.

Preliminary evaluation by us of transverse, thin-sectioned sagittal otoliths from adult wahoo revealed vague, opaque areas that potentially represented annular marks. Recent advances in otolith research have demonstrated that chemical treatment such as etching and digestion enhance the visibility of growth marks in difficult to interpret whole and sectioned otoliths (Richter and McDermott 1990, Shiao et al. 1999). Thus, in an effort to further examine the potential of wahoo sagittal otoliths as ageing structures, we conducted a small-scale experimental study to determine the effectiveness of specific etching, digestion, and staining techniques to enhance the visibility and contrast of vague, opaque marks (possible annual growth bands), as well as possibly expose "concealed or obscured" annual growth marks, on transverse thin-sections of wahoo sagittal otoliths.

MATERIALS AND METHODS

Sample collection

We sampled wahoo caught by recreational hook-and-line gear in the northcentral Gulf (June - September 1997, and May - June 1998) and, to a lesser extent, at Bimini, Bahamas (November 1997 and 1998). Wahoo from the northern Gulf were caught off the states of Louisiana, Mississippi, Alabama, and Florida in an area located north of lat. 29° and between long. 86°W and long. 89°W from waters that ranged 100 to 600 meters deep. Specimens from Bimini were caught in nearby Atlantic waters. We sampled specimens dockside and at fishing tournaments and recorded pertinent biological data for most specimens. Sagittal otoliths were extracted from specimens (57 males; 110 females), cleaned with water, air-dried, placed in vials with a collection number, and archived until processed.

Selection and Processing of Sagittal Otoliths

Left sagittae were selected from a sample of adult males (*n* = 20, 1,100 - 1,390 mm fork length) and females (*n* = 20, 1,000 - 1,803 mm fork length). Otoliths were embedded in epoxy resin and thin-sectioned transversely through the core at thicknesses ranging from 0.3 - 0.6 mm using a Buehler Isomet low-speed saw with a diamond blade. Sections were viewed under a dissecting microscope at 25 - 40x magnification using transmitted and reflected light to document any opaque marks.

Treatment of Otoliths

Based largely upon studies by Richter and McDermott (1990), Deree (1999), and Shiao et al. (1999), we selected two etching agents, one digestion agent, and four histological stains of various characteristics for our work. Our study consisted of five separate experiments with sectioned sagittae:

i) staining with a 1% solution of four histological stains only (Table 1);
ii) etching and staining with an acidified solution (1% acetic acid) of each of the four histological stains;
iii) etching with 5% ethylenediaminetetraacetate (EDTA);
iv) etching with 5% EDTA, followed by applications of each of the four stains, and
v) digestion by proteinase K buffer (PKb) (Table 2). Times of exposure to etching agents and stains varied somewhat, depending upon the particular experiment, and ranged from a few minutes to several days.

Proteinase K (buffered) was selected for its potential to degrade the fibrous proteins within thin-sections of sagittae. Components of the PKb mixture were diluted from stock solution (Table 2). Sagittae were digested with 0.2 ml PKb in 1.5 ml Eppendorf vials at 45°C with gentle shaking for periods of time which ranged from 0.25 - 48 hours. Samples were periodically removed from the PKb to assess the progress of digestion. Freshly prepared PKb was used for each digestion.

Otolith sections were examined with both transmitted and reflected light following staining, etching, and digestion techniques. Appraisals and general conclusions on the appearance and general morphology of sectioned sagittae were based upon the observations of three readers.

RESULTS

Untreated Otoliths

Observations of whole sagittal otoliths from adult wahoo under a dissecting microscope at 24 - 40x magnification using both transmitted and reflected light generally revealed a series of "ridges" (or "steps") and other features on the posterior region (postrostrum) of the distal surface. The proximal surface, which contained the sulcus, displayed a smoother exterior. The rostrum and antirostrum displayed a series diminutive marks (some conspicuous, some vague) along the surface of their entire length. Transverse sections typically revealed intensely opaque dorsal and ventral lobes with faint, opaque marks. Some sections appeared exceedingly crystalline. With the exception of two otoliths, "distinctive" opaque bands were not observed on any sections.

Table 1. Histological stains and staining techniques.

Stain	Color	Specificity
Aniline Blue	Blue	Collagen, cartilage
Eosin Y	Red (Orange)	Cytoplasm
Neutral Red	Red	Embryonic tissues
Toluidine Blue	Blue	Metachromic nuclear stain

Methods applied to each of the stains tested:

1. Sectioning and staining in 1% stain
2. Sectioning and staining in 1% stain in 1% acetic acid
3. Sectioning, etching with EDTA, and staining in 1% stain

Source: Richter and McDermott (1990); Deree (1999)

Treated Otoliths

Histological stains only — Otoliths from 10 fish were used in this experiment. Application of the four stains to mounted and unmounted (not on slides) sections did not enhance vague opaque bands and did not reveal any obscured or concealed growth marks. Sections stained for < 0.5 hours appeared faintly stained, and those stained for 0.5 - 2.0 hours became darkened without any observable contrast in cross-section features.

Table 2. Composition of Proteinase K buffer.

	Stock solution concentration	Usage	Volume final Concentration
Proteinase K		10 mg	1 mg/ml
Tri-HCl (pH = 8.0)	1 M	0.1 ml	0.01 M
SDS (Sodium dodecyl sulphate)20%	0.5 ml	1%	
NaCl	5 M	0.02 ml	0.01 M

Total volume of PKb is brought to 10 ml using distilled water.
Source: Shiao et al. (1999)

Acidified histological stains — Otoliths from 20 fish were used in this experiment. Application of the four acidified (1% acetic acid) histological stains to mounted and unmounted sections produced poor results. Applications < 1 hour resulted in minimal etching and faint staining which did not enhance the opaque marks and showed no contrast in any section features. Applications >1 hour resulted in the progressive destruction of sections, characterized by darkened, ambiguous cross-section features.

EDTA — Otoliths from 10 fish were used in this experiment. Etching of sections with EDTA from 0.1 - 1 hours produced no discernable enhancement of cross-section features, and presumably "obscured" bands were not revealed. Etching for >1 hour essentially destroyed the sections.

EDTA and histological stains — Otoliths from 10 fish were used in this experiment. Etching with EDTA from 0.1 - 1 hours, followed by staining with the four histological stains from 1 - 3 hours did not enhance the contrast or visibility of features on mounted sections. Unmounted sections were not treated with EDTA and histological stains.

Proteinase K buffer — Otoliths from nine fish were used in this experiment. Digestion by PKb did not enhance any opaque marks on sections, nor did the treatment reveal any "obscured" annular bands. Digestion with PKb for 48 hours radically altered the surface morphology of sections and rendered them unreadable.

Enhancement of bands: the exceptions — The exceptions to all sagittae examined in our study were those from the two largest fish in our sample (females; 1,780 and 1,803 mm FL). Those otoliths provided thin-sections that displayed narrow, moderately conspicuous opaque bands on their ventral lobe that were not observed on any other sagittae. These bands were distinctly different in their morphology and physical location on sections than were the faint, opaque markings on sections that we originally considered as possible age marks.

Although the bands on the two sagittae were discernable prior to any chemical treatments, etching and staining did moderately enhance their visibility and contrast against the background opaqueness of the ventral lobe. Among the treatments used with the two otoliths, application of acidified Neutral Red for a period of two hours produced the most obvious enhancement of opaque bands. Sectioned sagittae indicated the existence of 5 and 6 opaque bands for the 1,780 and 1,803 mm FL specimens, respectively. Since all sagittae were processed identically, we are puzzled by the appearance of distinctive opaque bands only on sagittae from the two large fish and by their absence on all other sagittae examined. We questioned: were the opaque marks representative of annular growth, and why were they not visible on thin-sections from all sagittae examined?

DISCUSSION

Etching and digestion agents and histological stains used in our study did not enhance the visibility of vague, opaque features observed in cross-sections of most wahoo sagittae, nor did the treatments reveal the existence of possibly obscured growth zones in any sections. Distinctive opaque bands on sagittal sagittae from the two largest wahoo in our sample were enhanced somewhat when treated with acidified Neutral Red, and it is presumed that the etching agent (acetic acid) reacted with surface $CaCo_3$, leaving the protein to react with the stain.

Shiao et al. (1999) reported positive results using PKb on cross-sectioned otoliths from two marine fish. They observed that enhancement of large otolith increments, including annuli, often required several hours exposure to PKb to ensure visual enhancement, at least for otoliths which are amenable to PKb treatment. Shiao et al. (1999) further noted that PKb may not be suitable for use with highly calcified otoliths with scanty amounts of proteins. Perhaps wahoo sagittae qualify for this category.

Richter and McDermott (1990) observed that different types of otoliths require different methods of etching and staining. The chemical agents and procedures used in our study were not effective for use with wahoo otoliths, even though essentially identical techniques greatly enhanced the contrast and visibility of vague daily marks and annular bands on otoliths from species examined by Richter and McDermott (1990), Deree (1999), and Shiao et al. (1999). Admittedly, those researchers did observe growth bands, albeit vague, on most otoliths prior to chemical treatments, and, in retrospect, this is probably a prerequisite for the effective use of etching agents to enhance age-growth features. However, etching is common practice for revealing growth marks, some often totally obscured, on difficult to age otoliths (Haake et al. 1982, Secor et al. 1991).

Perhaps further experimentation with:

i) different concentrations of etching agents and stains used in our study;
ii) entirely different etching agents and stains; and
iii) different exposure times, would result in more favorable results.

Conversely, perhaps there are no etching, digestion or staining procedures available to enhance or reveal annular growth marks on cross-sectioned wahoo sagittae. This assumes that wahoo annular growth is expressed within sagittal otoliths in the form of alternating opaque and translucent bands.

Preliminary estimates of wahoo age based on evaluation of first dorsal fin spine cross-sections (Franks et al., In press) suggested a relatively high growth rate for wahoo, particularly for their first year of life, which is consistent with Hogarth's (1976) findings based upon assessments of whole sagittal otoliths.

The oldest two fish in our spine assessment study were presumed to be 5 and 6 years of age, Incidentally, those were the same two fish that revealed 5 and 6 bands on their sectioned sagittae in this study. Ultimately, detailed examination of whole otoliths (Luckhurst et al. 1997) may represent the best procedure for estimating the age of wahoo.

Although our research did not advance the knowledge of wahoo age and growth, our findings might be of some use to those concerned with age and growth of wahoo from the WCA. Continued efforts by researchers ultimately will result in accurate estimates of age and an understanding of growth rates, life span, and age structure of the catch. Knowledge of wahoo age and other critical life history aspects, as well as improved documentation of the recreational catch and commercial landings within the WCA, are crucial for wahoo stock assessments and the development of future management strategies throughout the region.

ACKNOWLEDGMENTS

We are indebted to numerous recreational fishers along the northcentral Gulf coast and at Bimini who allowed us to sample their catch of wahoo. We also express our deep appreciation to the directors and officials of several Gulf of Mexico sport fishing tournaments for the opportunity to sample wahoo. We specifically recognize Bobby Carter, Bill McClellan, Danny Pitalo, Laurie Osley, Bill Killduff, Jack and Deona Holmes, Jim Roberson, and Ron Cabassa for their tremendous help and support. We express our sincere gratitude to our friend Raul Miranda, tournament director for the Bimini Big Game Fishing Club, for his advocacy of our research and exceptional hospitality during our research in Bimini. We are very grateful to Walter Grater, USM Institute of Marine Sciences (USM-IMS), for his valued advice and help with the laboratory experiments conducted during this study. For their extremely valuable assistance in collecting biological samples from wahoo, we offer our sincere thanks to T. J. Becker and USM-IMS colleagues Amber Garber, Niki Garber, David Geter, Chuck Blend, Don Barnes, Read Hendon, Jude LeDoux, Jan Welker, and Gerry Griggs. We acknowledge Captain Gary A. Caputi, publisher of *Big Game Fishing Journal*, C. M. "Rip" Cunningham and Deborah Hood of *Salt Water Sportsman* magazine, and Jim Brown of the International Game Fish Association for their encouragement and support of our research. This work was made possible by support from the Mississippi Gulf Coast Billfish Classic (The Bisbee World Billfish Series), USM-IMS, Damon Chouest and the Chouest family of Galliano, LA, particularly the Chouest Fishing Team, and Bob Brown with Red Barron Pizza, Atlanta, GA.

LITERATURE CITED

Collette, B. B. and C. E. Nauen. 1983. FAO species catalogue. Vol. 2. Scombrids of the world. An annotated and illustrated catalogue of tunas, mackerels, bonitos, and related species known to date. *FAO Fish. Synop.*125, **2**:137.

Deree, H. L. 1999. Age and growth, dietary habits, and parasitism of the fourbeard rockling, *Enchelyopus cimbrius*, from the Gulf of Maine. *Fish. Bull.* **97**:39-52.

Franks, J. S., N. J. Brown-Peterson, M.S. Griggs, N.M. Garber, J.R. Warren, and K. M. Larsen.. (In press). Potential of the first dorsal spine for estimating the age of wahoo, *Acanthocybium solandri*, from the northern Gulf of Mexico, with comments on specimens from Bimini, Bahamas. *Proc. Gulf and Carib. Fish. Inst.* Vol 52.

Haake, P. W., C. A. Wilson, and J. M. Dean. 1992. A technique for the examination of otoliths by SEM with application to larval fishes. Pages 12-15 in: C. F. Bryan, J. V. Conner, and F. M. Truesdale (eds), Proceedings of the Fifth Annual Larval Fish Conference. LSU Press, Baton Rouge, LA.

Hogarth, W. T. 1976. *Life history aspects of the wahoo, Acanthocybium solandri (Cuvier and Valenciennes) from the coast of North Carolina.* Ph.D. Dissertation, North Carolina State University, Raleigh.107 pp.

Hunte, W. and R. Mahon. 1985. A preliminary investigation of the migration of oceanic pelagic fish in the Western Central Atlantic. *Proc. of the fourth session of the working party on assessment of marine fishery resources.* Paipa, Columbia, Oct-Nov, 1984. FAO Fish. Rep. No. 327.

Luckhurst, B. E., J. M. Dean, M. Reichert, M. Cameron, S. Manuel, and T. Trott. 1997. Use of microstructure analysis of the sagittal otoliths for age estimation of the wahoo, *Acanthocybium solandri*, from Bermuda. *Proc. Gulf Carib. Fish. Inst.* **49**:64-70.

NMFS. 1996. Fisheries of the United States. Current Fishery Statistics, 1996. Natl. Mar. Fish. Serv. (NMFS), U. S. Dep. Commer., Washington, D.C.

Richter, H. and J. G. McDermott. 1990. The staining of fish otoliths for age determination. *J. Fish Biol.* **36**:773-779.

Rivas, L. R. 1951. A preliminary review of the western North Atlantic fishes of the family scombridae. *Bull. Mar. Sci.* 1(3):209-230.

Robins, C. R. and G. C. Ray. 1986. *A field guide to Atlantic coast fishes: North America.* Houghton Mifflin Company, Boston, MA. 354 pp.

Secor, D. H., J. M. Dean, and E. H. Laban. 1991. Manual for otolith removal and preparation for microstructural examination. Publ. by the Electric Power Research Institute and the Belle W. Baruch Institute for Marine Biology and Coastal Research, p. 85.

Shiao, J. C., C. S. Tzeng, C. L. Leu, and F. C. Chen. 1999. Enhancing the contrast and visibility of daily growth increments in fish otoliths etched by proteinase K buffer. *J. Fish. Biol.* **54**:302-309.

Aspects of the Reproductive Biology of Tripletail, *Lobotes surinamensis*, in the Northern Gulf of Mexico

NANCY J. BROWN-PETERSON AND JAMES S. FRANKS
Department of Coastal Sciences, Gulf Coast Research Laboratory,
Institute of Marine Sciences, The University of Southern Mississippi,
Ocean Springs, MS 39566-7000 USA

ABSTRACT

Tripletail, *Lobotes surinamensis*, is a large pelagic fish that is becoming increasingly popular with recreational anglers throughout the Gulf of Mexico. There is, however, little information on the biology of this species and virtually nothing is known regarding the reproduction of tripletail. We examined tripletail captured from the northern Gulf of Mexico in the spring and summer between July 1994 and September 1999 to determine size at sexual maturity and reproductive patterns. No sexually immature male tripletail were captured during the course of the study. The size at 50% maturity for female tripletail is 485 mm TL, corresponding to an estimated age of one year. Mean monthly female gonadosomatic index (GSI) values were elevated throughout the summer, highest in July and near resting levels by September. Male GSI values were elevated from May through September. Histological analysis of gonadal tissue from 93 male tripletail showed the majority of males captured from May through September were running ripe. However, active spermatogenesis declined in the testis during the course of the reproductive season. Analysis of gonadal tissue from 118 females provides evidence that tripletail are a multiple spawning species. Females were found in the late developing ovarian maturation stage from June through August; regressed and spent females were captured from June through September. A high percentage of females captured during April, May and June were in the immature and early developing ovarian maturation stages. Females with oocytes undergoing final oocyte maturation (FOM) were found June through August, with the highest percentage in July. Thus, tripletail appear to have a three month spawning season in the northern Gulf of Mexico, with July the peak spawning time. However, most reproductively active female tripletail appear to remain well offshore. Estimates of batch fecundity, based on five tripletail undergoing FOM in July suggest fecundity is positively correlated with length; mean relative batch fecundity was 47.6 eggs/g ovary-free body weight. Preliminary estimates of spawning frequency, based on the percentage of females in the late developing ovarian stage undergoing FOM or containing postovulatory follicles during June through August, suggest that female tripletail are capable of spawning once every three days to five days in the northern Gulf of Mexico.

KEY WORDS: Tripletail, reproductive biology, histology

INTRODUCTION

The tripletail, *Lobotes surinamensis*, is a pelagic species that is found world wide in tropical and subtropical seas (Fisher 1978). Adult and juvenile tripletail, while never abundant, commonly occur along the north-central coast of the Gulf of Mexico during late spring and summer (Baughman 1941). Tripletail are often targeted by recreational fishermen along the northern Gulf of Mexico (Benson 1982) and enter the commercial catch along the west coast of Florida (Fischer 1978).

There is little life history information on tripletail, and virtually nothing is known about their reproductive biology in the Gulf of Mexico. Juvenile tripletail are commonly found in floating patches of *Sargassum* or other types of drift algae (Dooley 1972) and appear to be strongly associated with shaded structures (Baughman 1941). Adult tripletail are often found in deeper waters associated with rocky bottoms, channel markers, jetties and wrecks (Baughman 1941). Tripletail appear to be a summer spawning species, as females with ovaries containing large eggs have been reported in Texas from May through August (Baughman 1941, 1944) and in Mississippi in July and August (Baughman 1941). In North Carolina, Merriner and Foster (1974) found running ripe males from June through August and spent females in August and September. Ditty and Shaw (1993) found larval tripletail in the Gulf of Mexico from July through September and concluded that the species is an offshore spawner. The objective of this study was to provide more complete information on the reproductive biology of tripletail in the northern Gulf of Mexico. Aspects of the reproductive biology described include the size at sexual maturity, spawning season, gonadal maturation, oocyte frequency distribution and estimates of batch fecundity and spawning frequency.

MATERIALS AND METHODS

Tripletail samples were obtained from recreational anglers along the north-central Gulf of Mexico off the coast of Mississippi between July 1994 and September 1999. All fish were measured (mm TL) and weighed (0.1 lb, later converted to g) dockside and the sex was determined. Gonads were removed from each fish, placed in a labeled bag and stored on ice no longer than 36 h prior to laboratory processing. In the laboratory, gonads were weighed (0.1 g) and assigned a macroscopic maturity stage. The gonadosomatic index (GSI) was calculated for each fish as GSI = [gonad weight/gonad-free body weight] x 100. A portion of the gonad (2 - 15 g) was removed and preserved in 10% neutral buffered formalin for a minimum of two weeks.

A small (< 2 cm^2) portion of the preserved gonadal tissue was placed in individually labeled cassettes for histological processing. Following an overnight rinse in running tap water, gonadal tissue was dehydrated in a series of graded ethanol solutions and embedded in paraffin following standard histological techniques. Tissues were sectioned at 2-3 μm, placed on slides and stained with hematoxylin and eosin for histological evaluation. Gonads were assigned to developmental stages based on microscopic appearance following Brown-Peterson et al. (1988). Spawning frequency was determined histologically following procedures in Hunter et al. (1986) and Brown-Peterson et al. (1988).

The remaining preserved ovarian tissue was used for oocyte size-frequency distributions and batch fecundity estimates. Approximately 1 g of preserved tissue was weighed to the nearest 0.0001 g and all oocytes > 60 μm were teased from the tissue. The oocytes were suspended in a known volume of water, stirred to a uniform distribution and three to five 1 ml subsamples were removed. All the oocytes in each subsample were counted and measured using a stereo dissecting microscope and a computerized image analysis system. Oocyte size frequency distributions were constructed using all the oocytes measured. Mean batch fecundity was determined for females undergoing final oocyte maturation from the number of oocytes ≥ 550 μm in each replicate subsample.

RESULTS

Size at Sexual Maturity

All male tripletail sampled during this study were sexually mature, exhibiting spermatogenic activity in the testis. The smallest male captured was 290 mm TL, suggesting that 50% maturity for male tripletail is reached at a length < 290 mm TL.

Overall, 26% of female tripletail collected from the northern Gulf of Mexico were sexually immature, with only primary oocytes in the ovary. Figure 1 shows the percentage of sexually mature females by 25 mm TL intervals. The smallest female captured was 300 mm TL and ovarian maturation did not begin until 444 mm TL. All females > 525 mm TL were sexually mature (Figure 1). Female size at 50% sexual maturity is calculated to be 485 mm TL, corresponding to an estimated age of one year.

Spawning Season and Gonadal Maturation

Tripletail were captured sporadically in the northern Gulf of Mexico between late April and late September during the years of this study. Due to the limited number of samples captured during each year, all data from 1994 through 1999 were pooled for the reproductive analyses. GSI values of female tripletail were elevated during June through August, with a peak in July, while male tripletail GSI values were elevated from May through September (Figure 2). Thus, the spawning season for tripletail in the northern Gulf of Mexico extends from June through August. Overall, GSI values of both male and female tripletail were low.

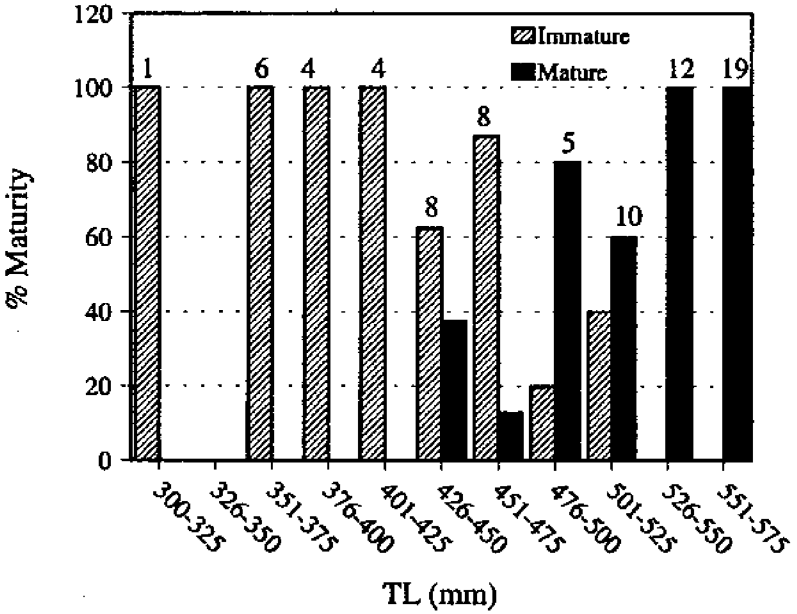


Figure 1. Size at maturity for female tripletail from the northern Gulf of Mexico by 25 mm TL intervals. The number of fish in each interval is above each bar.

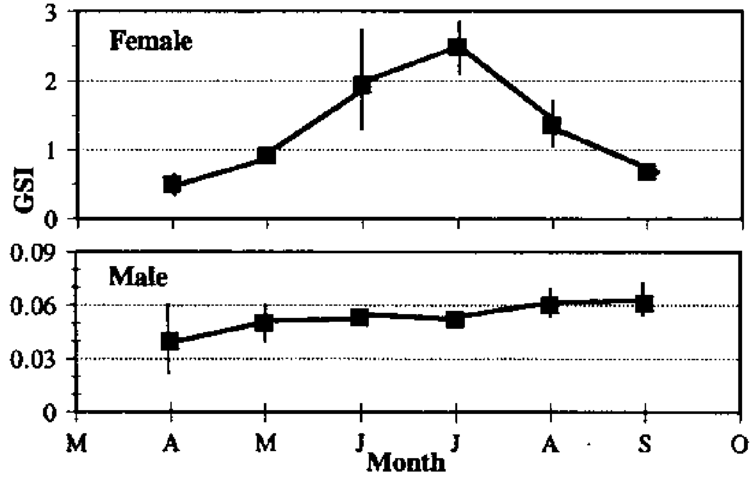


Figure 2. GSI values of female and male tripletail from the northern Gulf of Mexico. Data points represent x̄ ± 1 S.E. Data from 1994 -1999 are combined.

Histological assessment of testicular tissue from 93 male tripletail captured between April and September showed that the majority of the males were in spawning condition (running ripe) from May through September (Table 1). Few spent fish were captured. There was a noticeable progression in spermatogenic stages in the testis during the course of the season. Most males in April were in the mid and late developing testicular stages and undergoing active spermatogenesis (Table 1). Active spermatogenesis occurred throughout the testis in the majority of running ripe males during May (100%) and June (53%; Figure 3A). Active spermatogenesis occurred only in the peripheral portions of the testis in the majority of running ripe males in July (50%), August (46%) and September (50%). At the end of the spawning season, only spermatozoa, with no evidence of active spermatogenesis, was seen in the testis of some running ripe males (Figure 3B).

Histological assessment of ovarian tissue from 118 tripletail revealed that a high percentage of females captured in April, May and June had not yet reached sexual maturity (Table 2). However, sexually mature fish began ovarian recrudescence in May and early stages of ovarian development were observed in females through July. A small percentage of fish were in reproductive condition during June, July and August, with ovaries in either the late developing (Figure 3C) or ripe (oocytes undergoing final oocyte maturation, or FOM; Figure 3D) stages. Regressed fish first appeared in samples by June and the majority of females captured in August and September were regressed (Table 2).

Table 1. Monthly histological testicular developmental stages of tripletail from the northern Gulf of Mexico. Data from 1994 through 1999 are combined. All values are expressed as percentages.

Stage	April	May	June	July	August	Sept
N	5	6	18	25	35	5
Early Developing	20	0	0	0	0	0
Mid Developing	40	17	6	13	0	0
Late Developing	20	17	0	17	3	0
Running Ripe	20	50	94	70	97	80
Spent	0	16	0	0	0	20
Regressed	0	0	0	0	0	0

Table 2. Monthly histological ovarian developmental stages of tripletail from the northern Gulf of Mexico. Data from 1994 through 1999 are combined. All values are expressed as percentages.

Stage	April	May	June	July	August	Sept
N	2	7	23	42	34	10
Immature	100	71	52	19	12	0
Early Developing	0	29	22	17	0	0
Mid Developing	0	0	0	7	0	0
Late Developing	0	0	13	17	15	0
Ripe	0	0	4	14	3	0
Spent	0	0	0	0	12	10
Regressed	0	0	9	26	59	90

Tripletail are a multiple spawning species with asynchronous oocyte development, as shown in oocyte size-frequency diagrams (Figure 4). Females in the early, mid and late developing ovarian stages show continuous recruitment of oocytes in all size ranges with no distinct mode of oocytes (Figure 4). Females in the ripe ovarian stage represent those undergoing FOM. Oocytes $\geq$ 550 μm in diameter will undergo hydration and form a distinct mode of larger oocytes prior to actual spawning. Females in the ripe stage continue to show large numbers of developing oocytes, suggesting multiple batches of oocytes would be spawned during the reproductive season. Strong histological evidence for multiple spawning is the presence of developing oocytes in late developing ovaries (Figure 3C) and in ovaries that are undergoing FOM (Figure 3D). Finally, the presence of postovulatory follicles (POF; Figure 3C), indicative of spawning during the prior 24 hour period, show that tripletail have the ability to spawn more than once during the reproductive season. While POF were not common in tripletail ovaries, they were found during all months of the reproductive season.

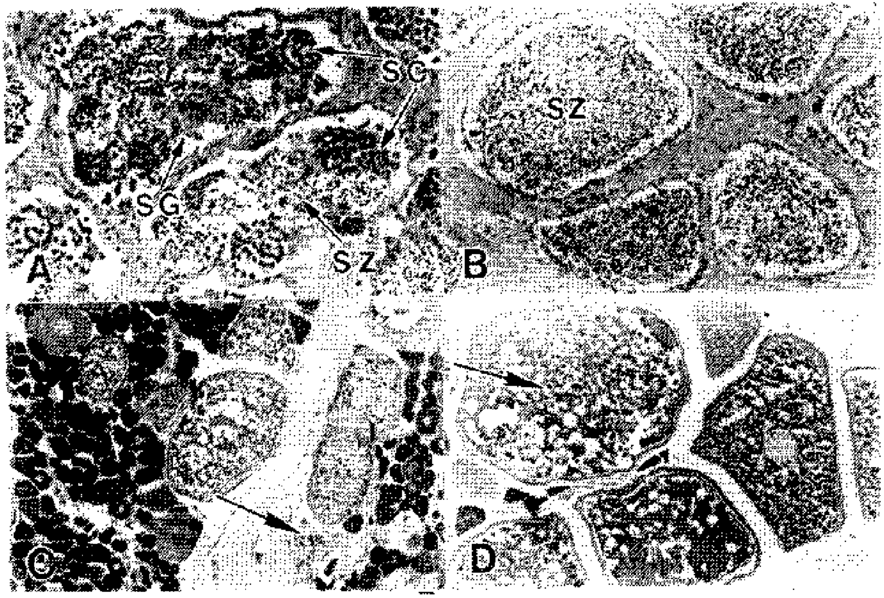


Figure 3. Histological sections of gonadal tissue from tripletail captured in the northern Gulf of Mexico. A. Testis from a tripletail captured in June showing spermatogenic activity throughout the testis. SG, spermatogonia; SC, spermatocytes; SZ, spermatozoa. 138X. B. Testis from a tripletail captured in September with only spermatozoa (SZ) and no spermatogenic activity in the testis. 227X. C. Ovary from a tripletail captured in August in the late developing ovarian stage. Note oocytes in all stages of development and post ovulatory follicle (arrow). 42X. D. Ovary from a tripletail captured in July in the ripe ovarian stage with oocytes undergoing final oocyte maturation (FOM; arrow). Note the presence of other mature oocytes not undergoing FOM in the section. 64X.

Batch Fecundity and Spawning Frequency

Preliminary estimates of batch fecundity were determined from five tripletail with oocytes undergoing FOM that were captured in July 1996 from the northern Gulf of Mexico. Mean batch fecundity values, the length of each fish and the relative batch fecundity, expressed as the number of eggs per g of ovary-free body weight, are presented in Table 3. The overall mean batch fecundity estimate for tripletail is 444,743 ± 201,139 eggs/female. However, batch fecundity appears to be related to fish length in tripletail (Table 3). Relative batch fecundity adjusts for fish size and is a better measurement for comparing fish of different sizes. The mean relative batch fecundity for tripletail is 47.6 ± 18.1 eggs/g ovary-free body weight.

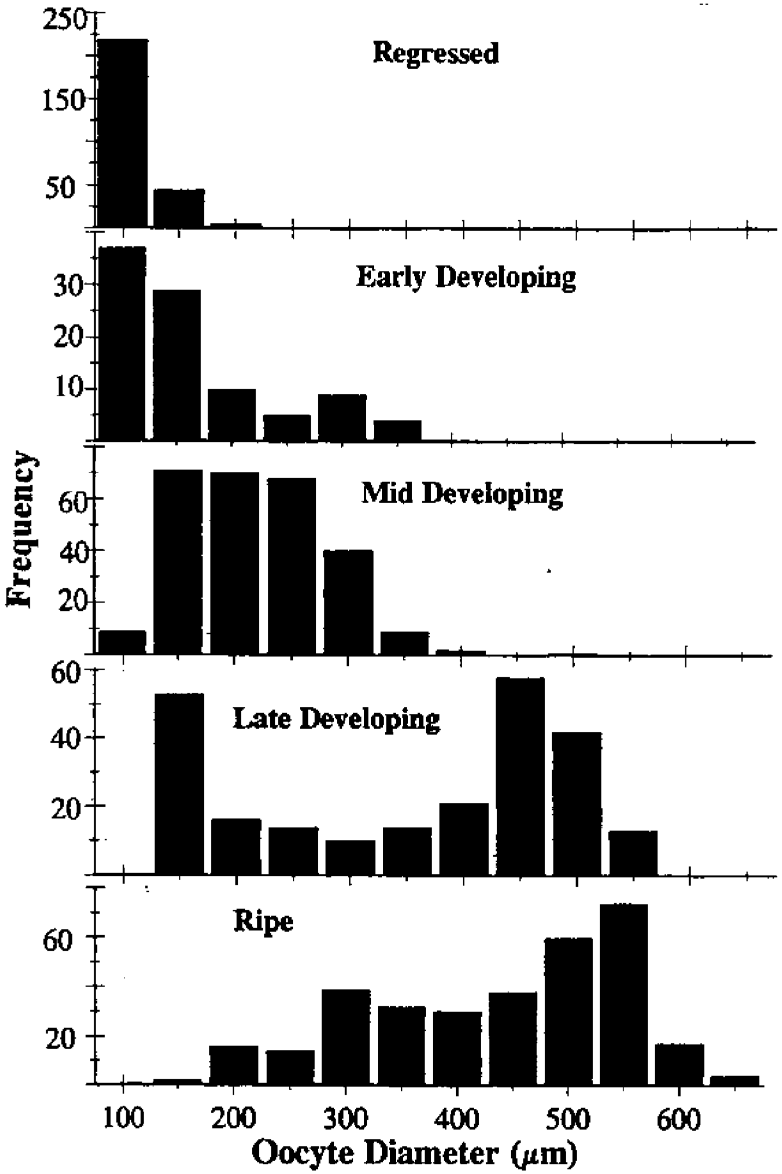


Figure 4. Oocyte size-frequency distribution of female tripletail. Frequency refers to the number of oocytes counted in each 50 μm oocyte diameter interval. Each graph represents data from a single fish at a different reproductive stage.

Table 3. Estimated batch fecundity for five tripletail captured in July 1996 from the northern Gulf of Mexico. Fecundity was estimated from oocytes undergoing final oocyte maturation (FOM).

TL (mm)	Batch Fecundity	Relative Batch Fecundity (eggs/g)
540	66843	20.3
660	335550	49.5
718	162960	22.5
725	455400	N/A[1]
787	1202960	98
Mean	444,743 ± 201,139	47.6 ± 18.1

[1] Fish weight not available for this specimen

Spawning frequency of female tripletail in the late developing or ripe ovarian stages for June, July and August combined (N = 22) was estimated using two methods; the percentage of ovaries containing oocytes undergoing FOM and the percentage of ovaries containing POF. Although the data are limited, spawning frequency estimates range from once every 2.7 days using the FOM method to once every 4.4 days using the POF method. Thus, female tripletail appear to spawn once every three to five days in the northern Gulf of Mexico. Assuming a three month spawning season (June through August), these estimates suggest that an individual female tripletail could spawn between 18 and 31 times during the spawning season, resulting in an estimated total annual fecundity for an average-size adult tripletail of 625 mm TL of 4, 619,866 to 7,956,429 eggs.

DISCUSSION

Data from this preliminary study show that tripletail from the northern Gulf of Mexico are capable of producing multiple batches of oocytes during a spawning season extending from June through August, with peak reproduction in July. Additional collections of tripletail at the beginning and ending of the reproductive season would provide more definitive information on the initiation and cessation of spawning activity. The duration of the spawning season is similar to previous reports of tripletail reproduction in the northern Gulf of Mexico and the southeastern Atlantic Ocean (Baughman 1941 1944, Merriner and Foster 1974). Furthermore, the July peak spawning time coincides with the time of peak larval collections reported by Ditty and Shaw (1993).

The relatively high occurrence of immature female tripletail, as well as the low percentage of reproductively active fish in our collections, is no doubt related to the offshore spawning nature of the species as hypothesized by Ditty and Shaw (1993). The majority of our samples were taken well inshore of the outer shelf, the location where Ditty and Shaw (1993) captured the smallest tripletail larvae. It appears that non-reproductive fish are most common in Mississippi coastal waters during the spawning season. Juvenile and sub-adult female tripletail were captured most frequently during April through June while spent and regressed females made up the majority of the sample July through September. Although Ditty and Shaw (1993) reported the appearance of a small number of larval tripletail in the Gulf of Mexico in May, the histological data collected during this study does not support tripletail spawning in May. Additional collections of offshore adult fish would allow a more accurate description of the gonadal maturation and spawning of the species.

The gonadal maturation of tripletail is similar to most other multiple spawning species and the low mean GSI values of female tripletail are similar to other large pelagic species such as wahoo (*Acanthocybium solandri*; Brown-Peterson et al. 2000) and yellowfin tuna (*Thunnus albacares*; McPherson 1991). However, the exceptionally low GSI values seen for male tripletail (never > 0.1) have not been previously reported for any other species. The reduction in active spermatogenesis noted in the testis of tripletail at the end of the reproductive season has been reported for other multiple spawning, subtropical species such as snook (*Centropomus undecimalis*; Grier and Taylor 1998), cobia (Brown-Peterson et al. in review) and spotted seatrout (*Cynoscion nebulosus*; Brown-Peterson et al. 1988) and is relatively common in species with an extended reproductive season. Male tripletail appear to reach sexual maturity at a smaller size and younger age than female tripletail. Little information is available on size or age at sexual maturity for the species; Merriner and Foster (1974) reported that males in North Carolina were sexually mature at age 1+, corresponding to a size of 445 mm TL. This is much larger than the smallest sexually mature male from this study, a 290 mm TL individual with an estimated age of 0+. However, Merriner and Foster (1974) also reported that females probably reached sexual maturity at age 1+ in North Carolina, which is in agreement with our calculated 50% maturity of 485 mm TL and estimated age 1. Thus, tripletail appear to grow rapidly during their first year of life, but this rapid growth does not seem to interfere with early sexual development.

This study represents the first attempt to estimate batch fecundity and spawning frequency for tripletail. Although the fecundity measurments presented here are based on small sample sizes, they are similar to batch fecundities reported for other large, migratory pelagic species. The relative batch fecundity of 47.6 eggs/g calculated for tripletail is comparable to 57.7 eggs/g for wahoo (Brown-Peterson et al. 1999), 57 eggs/g for bluefin tuna (*Thunnus maccoyii*;

Farley and Davis 1998), 68 eggs/g for yellowfin tuna (Schaefer 1996) and 62.4 eggs/g for cobia (Brown-Peterson et al. in review). Additionally, the preliminary spawning frequency estimates of once every three to five days for tripletail are similar to estimates for other pelagic species that occur in the northern Gulf of Mexico, such as once every two to six days for wahoo (Brown-Peterson et al. 1999) and once every five days for cobia (Brown-Peterson et al. in review). Thus, although tripletail have a shorter reproductive season than other pelagic species in the northern Gulf of Mexico, their gonadal maturation, batch fecundity and spawning frequency appear to be similar to other species with comparative life histories. Future work on tripletail in the northern Gulf of Mexico should include efforts to obtain samples of offshore adults during spring and summer to increase the sample size of spawning fish which would allow more accurate estimates of spawning seasonality, batch fecundity and spawning frequency.

ACKNOWLEDGMENTS

We thank the numerous recreational anglers along the northcentral Gulf of Mexico who allowed us to sample their catches of tripletail, as well as Don Barnes, Casey Nicholson and Read Hendon (Gulf Coast Research Laboratory, GCRL) who made specific trips to provide us with tripletail samples. Tracy Mitchell and A. Meaghin Burke counted numerous oocytes for oocyte distribution and fecundity determinations. Cheryl Crowder, Louisiana State University School of Veterinary Medicine, did the histological preparations and Susan Carranza, GCRL developed and printed the photographs. Mark Peterson reviewed the manuscript. This study was funded by the Mississippi Department of Marine Resources through the Federal Aid in Sport Fish Restoration Project, No. F-120.

LITERATURE CITED

Baughman, J.L. 1941. On the occurrence on the Gulf Coast waters of the United States of the triple tail, ***Lobotes surinamensis***, with notes on its natural history. ***Am. Nat.* 75**:569-579.

Baughman, J.L. 1944. Notes on the Serranidae and Lobotidae of Texas. *Copeia* **1944**:98-90.

Benson, N.G. 1982. Life history requirements of selected finfish and shellfish in Mississippi Sound and adjacent areas. U.S. Fish and Wildl. Serv., Office Biol. Serv., Washington,, D.C., FWS/OBS-81/51, 97 pp.

Brown-Peterson, N.J., J.S. Franks, and A.M. Burke. 2000. Preliminary observations on the reproductive biology of wahoo, *Acanthocybium solandri*, from the northern Gulf of Mexico and Bimini, Bahamas. *Proc. Gulf Carib. Fish. Inst.* **51**:414-427.

Brown-Peterson, N.J., R.M. Overstreet, J.M. Lotz, J.S. Franks, and K.M.

Burns. In review. Reproductive biology of cobia, *Rachycentron canadum*, from coastal waters of the southern United States. *Fish. Bull., U.S.*

Brown-Peterson, N. P. Thomas, and C.R. Arnold. 1988. Reproductive biology of the spotted seatrout, *Cynoscion nebulosus*, in south Texas. *Fish. Bull., U.S.* **86**:373-388.

Ditty, J.G. and R.F. Shaw. 1993. Larval development of tripletail, *Lobotes surinamensis* (Pisces: Lobotidae), and their spatial and temporal distribution in the northern Gulf of Mexico. *Fish. Bull, U.S.* **92**:33-45.

Dooley, J.K. 1972. Fishes associated with the pelagic Sargassum complex, with a discussion of the sargassum community. *Cont. Mar. Sci.* **16**:1-32.

Farley, J.J. and T.L.O. Davis. 1998. Reproductive dynamics of southern bluefin tuna, *Thunnus maccoyii*. *Fish. Bull., U.S.* **96**:223-236.

Fischer, W. 1978. FAO species identification sheets for fishery purposes. Western Central Atlantic (Fishing Area 31), Vol. 3. FAO, Rome.

Grier, H.J. and R.G. Taylor. 1998. Testicular maturation and regression in the common snook. *J. Fish Biol.* **53**:521-542.

Hunter, J.R., B.J. Macewicz, and J.R. Sibert. 1986. The spawning frequency of skipjack tuna, *Katsuwonus pelamis*, from the south Pacific. *Fish. Bull., U.S.* **84**:895-903.

McPherson, G.R. 1991. Reproductive biology of yellowfin tuna in the eastern Australian fishing zone, with special reference to the northwestern Coral Sea. *Aust. J. Mar. Freshwater Res.* **42**:465-477.

Merriner, J.V. and W.A. Foster. 1974. Life history aspects of the tripletail, *Lobotes surinamensis* (Chordata-Pisces-Lobotidae), in North Carolina waters. *J. Elisha Mitchell Sci. Soc.* **90**:121-124.

Schaefer, K.M. 1996. Spawning time, frequency, and batch fecundity of yellowfin tuna, *Thunnus albacares*, near Clipperton Atoll in the eastern Pacific Ocean. *Fish. Bull., U.S.* **94**:98-112.

Spontaneous Spawning of Cobia, *Rachycentron canadum*, Induced by Human Chorionic Gonadotropin (HCG), with Comments on Fertilization, Hatching and Larval Development

JAMES S. FRANKS, JOHN T. OGLE, JEFFERY M. LOTZ,
L. CASEY NICHOLSON, DONALD N. BARNES,
AND KIRSTEN M. LARSEN
The University of Southern Mississippi, Institute of Marine Sciences
Gulf Coast Research Laboratory
P.O. Box 7000
Ocean Springs, Mississippi 39566-7000 USA

ABSTRACT

Two mature female cobia, *Rachycentron canadum*, injected with a single dose of human chorionic gonadotrophin (HCG) at 275 IU/ kg of body weight, and one non-injected ripe male spawned spontaneously in captivity. Oocytes aspirated from each female prior to injection were maturing vitellogenic oocytes nearing the final oocyte maturation (FOM) stage and averaged 625 μ in diameter. Both females spontaneously spawned ~ 42 hours post-injection; spawned oocytes ranged 1.1 - 1.3 μ in diameter. Fertilized eggs hatched ~26 hours later. Estimates for number of eggs spawned (both females combined) and hatched were 3.2 million and 320,000, respectively. Aspects of embryogenesis and larval growth/development were observed. Critical survival period for larvae was day 3 at which time termination of yolk sac absorption occurred and first feeding commenced. Enriched rotifers, wild zooplankton, and artificial food were offered larvae during larval rearing treatments. Larvae contained in a black tank and fed a high density diet of enriched rotifers exhibited highest survival and were reared through day 13, post-hatch. The study documents the spontaneous spawning of wild-caught male and female *R. canadum* from the Gulf of Mexico, and provides comments on fertilization, hatching and larval development. Results of the study strongly suggest that *R. canadum* exhibits potential as an aquaculture species.

KEY WORDS: *Rachycentron canadum*, HCG injection, spontaneous spawning

INTRODUCTION

Cobia, *Rachycentron canadum*, are large, migratory, coastal pelagic fish of the monotypic family Rachycentridae and are distributed worldwide in tropical and subtropical seas, except for the eastern Pacific (Shaffer and Nakamura 1989). In the western Atlantic Ocean, cobia occur from Massachusetts and Bermuda to Argentina, including the Caribbean Sea, but are most common along the U. S. south Atlantic coast and in the northern Gulf of Mexico (Shaffer and Nakamura

1989) where the species supports valuable fisheries and is highly sought for its excellent flesh.

Life history information on cobia from the Gulf of Mexico (Gulf) and U. S. Atlantic coast includes: occurrence, distribution and description of early life stages (Dawson, 1971; Ditty and Shaw, 1992); age and growth (Joseph et al. 1964; Richards 1967 1977, Thompson et al. 1991, Smith 1995, Franks et al. 1999); reproductive biology (Thompson et al. 1991, Biesiot et al. 1994; Lotz et al. 1996, Brown-Peterson et al. in review); feeding habits (Knapp 1951, Franks et al., 1996; Meyer and Franks, 1996); seasonal migratory patterns (Franks et al. 1991, Biesiot et al. 1994); and population genetics (Hrincevich 1993). Life history information on cobia from the Caribbean Sea is scant.

Lotz et al. (1996) and Brown-Peterson et al. (in review) observed that females caught in spring and summer in the northern Gulf had ovaries in the final oocyte maturation (FOM) stage, as well as oocytes in less developed stages, indicating that cobia are batch (indeterminate) spawners. Cobia spawning season in the northern Gulf extends from April through September, and highest gonosomatic index (GSI) values for males and females occur during April and May (Lotz et al. 1996, Brown-Peterson et al. in review).

Studies on aquaculture-related aspects of cobia are few. Ryder (1889) collected fertilized cobia eggs from U.S. east coast waters and described the development of eggs and larvae. Hassler and Rainville (1975) collected fertilized cobia eggs from the Gulf Stream off North Carolina, successfully hatched most of them and reared larvae through juvenile stage. Biesiot et al. (1994) successfully stripped wild-caught female Gulf cobia which had been injected with human chorionic gonadotropin (HCG) and also induced spontaneous spawning in a female cobia using HCG injections but were unsuccessful in attempts to fertilize eggs with cryopreserved sperm. Liao et al. (1995) reported that *R. canadum* is a promising aquaculture species in Taiwan.

Human chorionic gonadotrophin has been used successfully to accelerate oocyte maturation and induce ovulation and spawning in many marine fish species, as reviewed by Lam (1982) and Donaldson and Hunter (1983). To the best of our knowledge, the present study represents the first published account of the spontaneous spawning (not stripped) of wild-caught, captive male and female *R. canadum* from the Gulf of Mexico using HCG to induce ovulation and release of eggs, with observations on subsequent fertilization, hatching and larval development.

MATERIALS AND METHODS

Two female cobia (9.0 and 16.0 kg TW) were opportunistically caught by hook and line gear in the northern Gulf of Mexico from Mississippi coastal waters on 3 June 1996 and transported to the University of Southern Mississippi, Institute of Marine Sciences' Aquaculture Facility located at the Gulf Coast Research Laboratory, Ocean Springs, Mississippi, USA. Since cobia do not exhibit sexual dimorphism, laboratory examination was required to determine their sex and gonadal condition. Intra-gonadal samples gently aspirated from each fish using a small polyethylene cannula (3 mm diameter) revealed large, maturing vitellogenic oocytes nearing the FOM stage. Fish were held in a large recirculating seawater tank, with continuous aeration.

The following morning (4 June 1996) at 1030 h, approx. 20 hours after capture, each female was injected intramuscularly with HCG dissolved in distilled water at a dosage of 275 IU/kg body weight (BW). The amount of HCG used was based on the recommended dosage for induction of ovulation in striped bass (Harrell et al. 1990) and cobia (Biesiot et al. 1994). Injections were administered posterior to the base of the left ventral fin using a disposable syringe and an18G hypodermic needle. Both fish were minimally resistant to handling during the injection procedure. Fish were transferred to a circular fiberglass tank (3.7 m diameter x 1.2 m height, 11,350 L capacity) which was a component of an indoor, recirculating seawater system described by Ogle (1992). A male cobia (9.0 kg TW) previously captured on 2 May 1996 was available for the spawning attempt and was placed in the tank with the females. Prior examination of milt from the male revealed active sperm. The male cobia was not injected.

Water in the spawning tank was continuously aerated and filtered, and temperature and salinity were maintained at 23.0°C and 28.0 ‰., respectively. Fish were held under a 12 hour light-dark photo period and were not fed.

Spontaneous spawning occurred two days later (6 June 1996). Floating eggs were collected by an egg collector (180 µm mesh) described by Ogle (1992). Total numbers of spawned eggs were estimated volumetrically. Fertilized eggs were placed in 120 L hatching tanks and maintained at 23.0°C and 28.0‰. The number of larvae in hatching tanks was estimated volumetrically. The ratio of the number of larvae to total number of eggs incubated multiplied by 100 was taken as an estimate of hatching rate.

Surviving yolk sac larvae were stocked into circular, fiberglass tanks: two black tanks (0.6 m diameter x 0.4 m height, 120 L) and three light blue tanks (1.13 m diameter x 0.4 m height, 410 L) for larval rearing experimentation. The water was gently aerated and periodically cleaned of dead larvae and debris. Water temperature and salinity were maintained at 23.0°C and 28.0‰, respectively, in all tanks. No significant differences in general water quality were observed between the experimental groups. At day three, larvae were offered several types of food at varying densities. Experimental diets included rotifers (*Brachionus plicatilis*, 130 - 340 µm) enriched with protein Selco (Inve, Animal Health, Gent, Belgium), wild zooplankton, and a commercially available "artificial plankton" (AP-R, Ocean Star International). The rotifers were cultured on-site using the diatom *Chaetocerous gracilis*, and wild zooplankton was collected from nearby coastal waters by plankton net (335 μm mesh). Larvae were fed two times a day (morning and afternoon). Aeration kept the larvae and food in suspension during larval rearing treatments. The generalized details of the larval rearing treatments (five), including specific diet and pertinent observations, are presented in the results section of this paper.

Ten larvae from one of the tanks (Treatment 1) were arbitrarily chosen every day over the duration of the study and preserved in 95% ethanol. Total length (TL, mm) measurements of these larvae were made using a dissecting microscope with an ocular micrometer. Unless otherwise noted, reference to "day" throughout this paper refers to the number of days post-hatch (including hatch date).

All research activities conducted during this study were conducted in clarified, natural estuarine water (16.0‰) adjusted to a salinity of 28.0‰ with artificial sea salt (Marine Environment).

RESULTS

Ovarian samples aspirated from females prior to injection with HCG contained large, maturing vitellogenic oocytes that were spherical in shape. Twenty of the large oocytes from each fish averaged 625 μ in diameter. On the morning following the day that all fish were placed in the spawning tank, the male displayed what was interpreted as courtship behavior, included constant circling the females and swimming with its snout at their vent. We observed slight redness around the vent of the females, as previously reported by Biesiot et al. (1996) for females in a near-spawning state.

Ovulation and Spawning

A single administration of HCG 275 IU/kg BW was effective in producing hydration and the spontaneous shedding (not stripped) of eggs. Spawning commenced approximately 42 hours (0430 h, 6 June 1996) after injection with HCG. We presume that both females released eggs simultaneously. Based on the numbers of eggs collected, we estimated 3.2 million eggs were spawned (eggs from females combined). We were unable to determine the number of eggs spawned per female and the percentage of eggs that were viable, hydrated oocytes. The mean diameter of spawned ooctyes was 1.2 mm. The male spontaneously released sperm, simultaneous with the release of eggs by the females.

Incubation and Hatching

Fertilization rate was not determined but was presumed to be moderately high. At 12 hours post-spawn, eggs ranged 1.3 - 1.5 in diameter (mean = 1.4 mm), the embryo was developed on a three-quarter circumference, and the embryo, as well as a single yellow oil droplet, were bright yellow. Eggs basically appeared as described by Hassler and Rainville (1975) and Ditty and Shaw (1992). Live, fertilized eggs were buoyant and were also distinguishable from unfertilized, "dead" eggs by the lumpy, opaque appearance of the yolk in dead eggs. Dead eggs were periodically removed from hatching tanks.

Eggs were incubated in hatching tanks at 23.0°C. Hatching commenced ~24 h post-spawn, and most eggs hatched within four hours. An estimated 320,000 eggs hatched (presumed hatching success of 10%). At hatching, mean length of larvae was 2.5 mm, the yolk sac was large, and larvae appeared similar to that illustrated by Ditty and Shaw (1992).

Larval Rearing Treatments

An estimated 140,000 day one, yolk sac larvae were used in our small-scale larval rearing treatments. All treatments were conducted simultaneously and were unreplicated; the results of one treatment (Treatment 1) were superior to all others. Larval survival at day three appeared high. By day three, yolk sac absorption was nearing completion for most larvae, and food was offered all larvae. Mean size of larvae on day three was 4.3 mm TL. Summary results of treatments are presented as follows:

Treatment 1: 120 L black fiberglass tanks; larvae offered high density diet of enriched rotifers — The tank was stocked with ~10,000 larvae (84,000/m^3). Larvae were fed a high density of "standard-size" enriched rotifers 2 X per day in an attempt to maintain a feeding density of 200/ml which represents a density of prey considerably greater (~5 - 10X) than often recommended for the culture of some fish larvae (Ludwig 1994, Puvanendran and Brown 1999). First feeding was observed to commence on day three for some larvae, concurrent with the near-final absorption of remaining yolk. Feeding response was positive, and by day 5 many larvae were observed swimming actively and "striking" at rotifers in an S-shaped strike maneuver described by Hassler and Rainville (1975).

Survivability of the larvae appeared relatively high until day 13 when an aeration system malfunction resulted in the death of all remaining larvae. Although some larval mortalities (numbers unknown) occurred during the treatment period prior to day 13, possibly linked to the lack of successful first feeding, this treatment represents the most successful larval rearing experiment conducted during the study.

Treatment 2: 120L black fiberglass tank; larvae offered standard density diet of enriched rotifers — The tank was stocked with ~10,000 larvae (84,000/m^3). Larvae were fed "standard size"enriched rotifers 2 times per day in an attempt to maintain a feeding density of 20/ml. A few larvae were observed feeding on day 3, however, the overall feeding response was not positive. The food density probably resulted in insufficient prey surrounding the larvae. Most larvae never initiated first feeding, while others, although possibly successful at initiating first feeding, probably encountered prey too infrequently to survive. Total mortalities occurred by day seven.

Treatment 3: 410 L light blue plastic tank; larvae offered standard density diet of enriched rotifers — The tank was stocked with ~40,000 larvae (96,000/m^3). Larvae were fed "standard size"enriched rotifers 2 times per day in an attempt to maintain a feeding density of 20/ml. A few larvae were observed to commence feeding on day three, however most did not feed, and total mortalities occurred by day five. A high larvae:low prey density ratio presumably was the primary cause of the mortalities.

Treatment 4: 410 L light blue fiberglass tank; larvae were offered wild zooplankton — The tank was stocked with ~40,000 larvae (96,000/m^3). Composition of the zooplankton was not analyzed, but cursory observation revealed prodigious numbers of adult copepods. Zooplankton densities were not determined but were considered very low. Close scrutiny revealed that none of the larvae commenced first feeding, and total mortalities occurred by day five.

Treatment 5: 410 L light blue plastic tanks; larvae offered commercial food — The tank was stocked with ~40,000 larvae (96,000/m^3). Large quantities of a viable feed ("artificial plankton", 100 μm) were offered larvae 2 times per day (specific amount undetermined). First feeding was never initiated, and total mortalities occurred by day five.

General observations on growth and survival — Although not calculated, daily mortalities of yolk sac larvae, prior to yolk sac absorption on day three, appeared low in all tanks. On day five, observations indicated that a large percentage of the larvae in Treatment 1 had food in their gut, close to the time of 100% mortality in most other treatments.

The following information was based on examinations and observations of larvae from Treatment 1 only. Reported lengths include specimens with and without food in their gut, and lengths were recorded for specimens preserved in 95% ethanol for a period of four months. The percentage shrinkage of larvae in the preservative was unknown. Twenty-four hour-old yolk sac larvae exhibited a large yolk sac, averaged 3.5 mm TL, and were colorless. The yolk sac was absorbed in most larvae by the end of day three, at which time we observed relatively well developed mouth parts and a near functional digestive tract; mean size, 4.3 mm TL. First feeding was actually initiated by some three day-old individuals. By day five, larvae averaged 4.5 mm TL, and the development of the body, particularly the head and mouth, permitted continuous swimming, accompanied by rapid, darting motions, and an active feeding behavior. At day 10 (mean TL, 6.3 mm) the head was large and the eyes were well developed, and larvae were a yellowish-golden color. Mean length of 13 day-old larvae was 8.0 mm

Daily survival rates in Treatment 1 were not calculated, and assumptions about high survivorship in that treatment were based on the existence of a large number of well developed larvae on day 13.

DISCUSSION

Because northern Gulf cobia have a relatively well-defined spawning season, induced spawning of mature females caught during peak reproductive potential appears to represent a viable approach for obtaining large numbers of eggs in late maturation stages. Males reportedly are not as abundant as females (Franks et al. 1999), but are available. Our research was conducted opportunistically which necessarily limited the sample size of adults.

Our small-scale study demonstrated the effectiveness of gonadotropin (HCG) to induce ovulation and spawning in mature, wild-caught female *R. canadum* with well-developed oocytes prior to administration of HCG. The study also demonstrated that captive, ripe male cobia do not require administration of gonadotropin to released viable sperm in the presence of spawning females.

Administration of a single dose of gonadotropin to successfully induce ovulation and spawning in fishes is not unique, and the single dosage of HCG (275 IU/kg BW) applied in our experiment is similar to that often used with other marine fish (Harrell et al. 1990). The success of induced ovulation and spawning in fishes is dependent, in great part, on females exhibiting advanced stage, vitellogenic oocytes, which effectuates a greater response to the injection of hormones (Lam 1982). Furthermore, studies by De Montalembert et al. (1978) and Roland (1985) showed if degeneration of mature oocytes has begun in mature ovaries, success of hormone-induced ovulation and spawning is greatly reduced. In our study, there was no apparent significant variation in the microscopic appearance of advanced-stage oocytes aspirated from either female cobia prior to injection with HCG. Interestingly, Rowland (1988) observed that the lowest dosage of HCG required to induce ovulation or spawning generally resulted in the highest hatch rates.

Our spawn estimate of 3.2 million eggs (both females combined) is 2.5X greater than the estimated mean batch fecundity (53 eggs/g ovary-free body weight) reported for cobia by Brown-Peterson et al. (in review). Post-mortem examination of ovaries conducted shortly after the spawning event (females succumbed to poor water quality conditions in holding tanks a few days after spawning) suggested recent spawning. It is presumed that not all spawned oocytes were viable, hydrated ooctyes, i.e., the affects of HCG also may have caused the release of less developed oocytes.

We considered our estimated hatch rate of 10% moderately successful, however we have no data on the actual number of viable eggs spawned. Ayson (1989) commented that the choice of hormone for induced spawning should be based not only on its capacity to stimulate egg release but also its effect on egg quality and hatch potential. Our observations suggest that the dosage of HCG administered in our study had no apparent adverse effect on the general quality of cobia eggs, as suggested by a presumed high rate of fertilization, a successful hatch, and the number of yolk sac larvae produced. Most early larvae examined in our study appeared "normal" with morphological features and pigmentation basically as that described by Hassler and Rainville (1975).

A critical time in the life of larval fish is the onset of exogenous feeding, and failure to initiate feeding before the yolk is totally exhausted generally results in mortality (Blaxter 1986). The combination of prey concentrations and larval density significantly influence growth and survival of larvae. The larvae stocking density of 84,000/m^3 (83/L) in Treatments 1 and 2 was comparable to that reported for seabream (Fukusho 1990) but high when compared with that for rockfish (Ko and Lee, 1991) and some other species. Larval stocking density of 97/L in Treatments 3, 4, and 5 were high. We believe growth and survivability of larvae in Treatment 1 were directly related to the offering of suitable live food at the proper time and at high density, probably augmented by some visual advantage in locating prey against a black background.

Interestingly, Olsen et al. (1999) reported that the size distribution of prey organisms may be more important than the total number of prey per unit volume for the growth and survival of some marine fish species. Naas et al. (1996) illuminated "first feeding tanks" of various colors and found that black was best suited to reproduce natural illumination conditions, suggesting that black background may be important to creating an optimum first-feeding environment for larval cobia. The mortalities associated with the unsuccessful treatments suggest larvae were unable to initiate or maintain sufficient feeding. Although small particles of prey were observed in stomachs of some larvae from Treatment 2 (also a black tank), results strongly suggest that encounters with prey were not sufficient to prevent mass starvation mortality.

Low densities of wild zooplankton were presumed to be the cause of mortalities in Treatment 4. Field sampling efforts produced inadequate amounts of zooplankton, and larvae did not initiate first feeding. Adequate densities of zooplankton of appropriate composition and size may have potential as food for cobia larvae. Wild zooplankton, particularly copepods, may produce the best growth and survival rates for some larval fishes (Holmefjord et al. 1993), however, a zooplankton diet can be a limiting factor due to the uncertainty of supply, the occasional dominance of unfavorable species in the plankton fauna, and the variability in its nutrient content (Holmefjord et al. 1993). Although large quantities of a proven commercial food were offered larvae in Treatment 5, none of the larvae initiated feeding, suggesting that cobia require live food at first feeding.

Experimental rearing of pelagic fishes from fertilized eggs yields important knowledge on the biology of the species. Our successes with induced ovulation and spontaneous spawning of cobia, coupled with moderate success at larval rearing, are encouraging and enlarge the catalogue of possible areas for cobia cultivation. In fact, we consider cobia a promising aquaculture candidate. Adult brood stock, juveniles and sub-adults adjust well to captivity, lack aggressive behavior, and tolerate crowding.

Artificial inducement of final gonad maturation and spawning in cobia can provide extended access to viable gametes and may prove useful in future mass culture of cobia for commercial markets or in future efforts to rebuild stocks. Under exemplary conditions, captive adult cobia might be conditioned to spawn voluntarily.

ACKNOWLEDGMENTS

We thank Kathy Wilson, Dyan Wilson, Melanie Griggs, Nikola Garber, Amber Garber, Nate Jordan, and Diane Scott for their valued assistance during the study. The study was supported in part by the University of Southern Mississippi/Institute of Marine Sciences, the Mississippi Department of Marine Resources, and the Sport Fish Restoration Program of the U. S. Fish and Wildlife Service, Atlanta Georgia.

LITERATURE CITED

Ayson, F.G. 1989. The effect of stress on spawning of brood fish and survival of larvae of the rabbitfish, *Siganus guttatus* (Bloch). *Aquaculture* **80**: 241-246.

Biesiot, P.M., R.M. Caylor, and J.S. Franks. 1994. Biochemical and histological changes during ovarian development of cobia, *Rachycentron canadum*, from the northern Gulf of Mexico. *Fish. Bull., U. S.* **92**:686-696.

Blaxter, J.H.S. 1986. Development of sense organs and behaviour of teleost larvae with special reference to feeding and predator avoidance. *Trans. Am. Fish. Soc.* **115**:98-114.

Brown-Peterson, N J., R.M. Overstreet, J.M. Lotz, J.S. Franks, and K.M. Burns. (In review). Reproductive biology of cobia, *Rachycentron canadum*, from coastal waters of the southern United States.

Dawson, C.E. 1971. Occurrence and description of prejuvenile and early juvenile Gulf of Mexico cobia, *Rachycentron canadum. Copeia* **1960(3)**:171-180.

De Montalembert, G., G. Jalabert, and C. Bry. 1978. Precocious induction of maturation and ovulation in northern pike (*Esox lucius*). *Ann. Biol. Anim., Biochim., Biophys.* **18(4)**:969-975.

Ditty, J.G. and R.F. Shaw. 1992. Larval development, distribution, and ecology of cobia, *Rachycentron canadum* (Family:Rachycentridae), in the northern Gulf of Mexico. *Fish. Bull., U. S.* **90**:668-677.

Donaldson, E.M. and G.A. Hunter. 1983. Induced final maturation, ovulation and spermiation in cultured fish. Pages 351-403 in: W. S. Hoar, D. J. Randall, and E. M.. Donaldson (eds.), Fish Physiology, IXB. Acad. Press, London.

Franks, J.S., M.H. Zuber, and T.D. McIlwain. 1991. Trends in seasonal movements of cobia, *Rachycentron canadum*, tagged and released in the northern Gulf of Mexico. *J. Miss. Acad. Sci.* **36**(1):55.

Franks, J. S., N. M. Garber, and J. R. Warren. 1996. Stomach contents of juvenile cobia, *Rachycentron canadum*, from the northern Gulf of Mexico. *Fish. Bull., U. S.* **94**:374-380.

Franks, J.S., J.R. Warren, and M.V. Buchanan. 1999. Age and growth of cobia, *Rachycentron canadum*, from the northeastern Gulf of Mexico. *Fish. Bull.*, **97**(3):451-479.

Fukusho, K. 1990. Outline of aquaculture:red seabream culture. Kanagawa International Fisheries Training Center, Japan Int. Coop. Agency. 8 pp.

Harrell, R.M., J.H. Kerby, and R.V. Minton (eds.), 1990. Culture and propagation of striped bass and its hybrids. Striped Bass Committee, Southern Division, Am. Fish. Soc., Bethesda, MD. 323 pp.

Hassler, W. W., and R. P. Rainville. 1975. Techniques for hatching and rearing cobia, *Rachycentron canadum*, through larval and juvenile stages. Univ. N.C. Sea Grant Coll. Prog., UNC-SG-75-30, Raleigh, NC. 26 pp.

Holmefjord I., J. Guldbransen, I. Lein, T. Refstie, P. Léger, T. Harboe, I. Huse, P. Sorgeloos, S. Bolla, Y. Olsen, K.I. Reitan, O. Vadstein, G. Oie, and A. Danielsberg. 1993. An intensive approach to Atlantic halibut fry prduction. *J. World Aquacult. Soc.* **24**:275-284.

Hrincevich, A. W. 1993. Analysis of cobia *Rachycentron canadum* population structure in the northern Gulf of Mexico using mitochondrial DNA. M.S. thesis, Univ. Southern Miss., Hattiesburg, MS, 91 pp.

Joseph, E.B., J.J. Norcross, and W.H. Massmann. 1964. Spawning of the cobia, *Rachycentron canadum*, in the Chesapeake Bay area, with observations of juvenile specimens. *Chesapeake Sci.* **5**:67-71.

Knapp, F.T. 1951. Food habits of the sergeantfish, *Rachycentron canadus*. *Copeia* **1951**:101-102.

Ko, T.S. and C.C. Lee. 1991. Mass production experiment of rockfish seed. *Tech. Rep. Natl. Fish. Res. Develop. Agency, Korea.* **90**:45-50.

Lam, T.J. 1982. Applications of endrocrinology to fish culture.Can. J. Fish. *Aquat. Sci.* **39**:111-137.

Liao, I.C., S. Mao-Sen, and C. Su-Lean. A review of the nursery and growout techniques of high-value marine finfishes in Taiwan, Pages 121-137 in: M. L. Kevan and C. Rosenfeld (eds.), *Proceedings: Culture of high-value marine fishes in Asia and the United States.* The Oceanic Institute, Honolulu, HI, ISBN 1-886608-01-6.

Lotz, J.M., R.M. Overstreet, and J.S. Franks. 1996. Gonadal maturation in the cobia, *Rachycentron canadum*, from the northcentral Gulf of Mexico. *Gulf Res. Rep.* **9**(3):147-159.

Ludwig, G.M. 1994. Tank culture of sunshine bass *Morone chrysops* X *M. saxatilis* fry with freshwater rotifers *Brachionus calyciflorus* and salmon starter meal as first food sources. *Aquaculture* **25**(2):337-341.

Meyer, G.H., and J.S. Franks. 1996. Food of cobia, *Rachycentron canadum*, from the northcentral Gulf of Mexico. *Gulf Res. Rep.* **9**(3):161-167.

Naas, K., I. Huse, and J. Iglesias. 1996. Illumination in first feeding tanks for marine fish larvae. *Aqua. Engin.* **15**(4):291-300

Olsen, A.I., Y. Attramadal, A. Jensen, and Y. Olsen. 1999. Influence of size and nutritional value of *Artemia franciscana* on growth and quality of halibut larvae (*Hippoglossus hippoglossus*) during the live feed period. *Aquaculture* **179**:475-487.

Ogle, J.T. 1992. Design and operation of the Gulf Coast Research Laboratory penaeid shrimp maturation facility I., *Penaeus vannamei.* Tech. Rpt. 4. Gulf Coast Research Laboratory, Ocean Springs, MS 39564, 41 pp.

Puvanendran, V and J.A. Brown. 1999. Foraging, growth and survival of Atlantic cod larvae reared in different prey concentrations. *Aquaculture* **175**:77-92.

Richards, C. E. 1967. Age, growth and fecundity of the cobia, *Rachycentron canadum*, from the Chesapeake Bay and adjacent Mid-Atlantic waters. *Trans. Am. Fish. Soc.* **96**:343-350.

Richards, C.E. 1977. Cobia (*Rachycentron canadum*) tagging within Chesapeake Bay and updating of growth equations. Chesapeake Sci.

18:310-311.

Rowland, S J. 1985.Aspects of the biology and artificial breeding of the Murray cod, *Maccullochella peeli* and the Eastern freshwater cod, *M. ikei* n sp. Ph.D. Thesis, Macquarie Univ., N.S.W., 253 pp.

Ryder, J.A. 1887. On the development of osseus fishes, including marine and fresh water forms. *Rept. U. S. Fish. Comm.* 1885, **13**:488-604.

Shaffer, R. V., and E.L. Nakamura. 1989. Synopsis of biological data on the cobia *Rachycentron canadum* (Pisces:Rachycentridae). FAO Fisheries Synop. 153 (NMFS/S 153). U. S. Dep. Comm., NOAA Tech. Rep. NMFS 82. 21 pp.

Smith, J.W. 1995. Life history of cobia, *Rachycentron canadum* (Osteichthyes: Rachycentridae), in North Carolina waters. *Brimleyana* **23**:1-23.

Thompson, B.A., C.A. Wilson, J.H. Render, and M. Beasley. 1991. Age, growth and reproductive biology of greater amberjack and cobia from Louisiana waters. Year 1. Rep. to U. S. Dep. Commer., NOAA, NMFS, Coop. Agreement NA90AA-H-MF089, Marine Fisheries Initiative (MARFIN) Prog., Coastal Fish. Inst., Louisiana St. Univ., Baton Rouge. 55 pp.

Characterization of the Mitochondrial DNA Control Region of the Wahoo, *Acanthocybium solandri*, from the Northcentral Gulf of Mexico and Bimini, Bahamas

AMBER F. GARBER, KENNETH C. STUCK, JAMES S. FRANKS,
NIKOLA M. GARBER and DAVID R. GETER
The University of Southern Mississippi, Institute of Marine Sciences
Gulf Coast Research Laboratory
P.O. Box 7000
Ocean Springs, Mississippi 39566-7000 USA

ABSTRACT

Wahoo, *Acanthocybium solandri*, rank among the most coveted of fish sought by offshore recreational anglers in many parts of the world and are the basis of important commercial fisheries. Little information is available on the biology and life history of the wahoo and no information exists on the molecular genetics of this pelagic, circumglobally distributed species. Our preliminary research consisted of a molecular characterization of the mtDNA control region, including structure and sequence of the flanking tRNA genes. Total DNA was isolated from ten wahoo tissue samples: five from the northcentral Gulf of Mexico and five from Bimini, Bahamas. Universal primers produced a fragment of approximately 1830 base pairs (bp) containing the tRNAs for proline, threonine, and phenylalanine plus the entire control region of one fish from each site. The fragments were gel purified for direct sequencing and species specific primers were designed in the tRNAs immediately flanking the control region and used in subsequent amplifications. The control region of the ten wahoo ranged in size from 886 bp to 892 bp. One termination activation sequence (TAS I), three conserved sequence blocks (putative CSB I, CSB II, CSB III), five repeats, and two inverted repeats were identified. The 5' end of the control region contained the greatest sequence variability with approximately 100 variable bases and 3 indels within the first 330 bp. Multiple restriction sites for HinfI, RsaI, and SspI were identified that could be used to distinguish each fish as a unique haplotype by RFLP analysis. Results provide a characterization of the wahoo mtDNA control region for possible use in future investigations.

KEY WORDS: *Acanthocybium solandri*, genetics, mitochondrial DNA

INTRODUCTION

The wahoo, *Acanthocybium solandri*, is a pelagic species found worldwide in tropical and warm temperate oceanic waters (Collette and Nauen 1983). Wahoo typically occur a considerable distance offshore, particularly around islands, pinnacles, ancient coral heads, and offshore banks. Attracted to current edges and temperature breaks, wahoo also occasionally occur nearshore along reef edges and walls. Common names for *A. solandri* depend on geographic location and include wahoo, ocean barracuda, tigerfish, kingfish, pride of Bermuda, peto, ono, oahu, and robalo.

Wahoo commercial landings from the western central Atlantic, including the Gulf of Mexico, increased from an average of 2.9 metric tons per year between 1974 and 1978 to 164 metric tons in 1992, and steadily decreased to 102 metric tons in 1996 (Mahon 1996). National Marine Fisheries Service (1999) data indicated that the wahoo landings in the Gulf of Mexico and Atlantic (off US states only), also peaked in 1992 and fluctuated between 1993 and 1998, with an overall decrease. Increased recreational fishing in U.S. and Caribbean waters has put additional fishing pressure on this species. There is very little information available on the biology and life history of the wahoo, and there is no published information on molecular genetics. With the exception of tag and release data, little information exists concerning the spatial limits of the pelagic habitat, or how this relates to the amount of genetic exchange actually occurring among populations of large, oceanic fishes (Rosel and Block 1996).

In this study, we used mitochondrial DNA (mtDNA), because it accumulates mutations at a rate up to ten times faster than the nuclear genome (Meyer, 1993). Within the mitochondrial genome, there are 13 genes that code for proteins, two that code for ribosomal RNAs (rRNA), and 22 that code for transfer RNAs (tRNA). Also, there is a noncoding region, termed the control region, that accumulates mutations two to five times faster than the rest of the mitochondrial genome (Meyer 1993). Since this is the most rapidly changing region of DNA in the cell, it can be useful in assessing population structure. This region, also referred to as the "D-loop" or "displacement loop region," contains conserved motifs and the sites of initiation for both heavy-strand replication and heavy- and light-strand transcription (Chang and Clayton 1987, Clayton, 1991a, 1991b; Digby et al. 1992).

The purpose of our study was to obtain the entire sequence of the tRNAs flanking the control region, proline (tRNA-pro) and phenylalanine (tRNA-phe), and use this information to design species specific primers to direct sequence and characterize the control region of wahoo.

MATERIALS AND METHODS

Wahoo were collected from the northcentral Gulf of Mexico, USA and off the coast of Bimini, Bahamas. Tissue was excised and fixed in SED buffer (250 mM EDTA, pH 7.5, 20% DMSO, 3.42 M NaCl). Total genomic DNA was extracted from each sample using a procedure modified from Taggart *et al.* (1992), quantified using fluorescence spectrophotometry (Gallagher 1994) and adjusted to concentrations of 100 ng/µL using 1 mM Tris, pH 8.5.

The universal primers CB3R (5' CACATTCAACCAGAATGATATTT 3';

Palumbi, 1996) and 12SA-H1067, referred to in the present work as 12SAR, (5' ATAATAGGGTATCTAATCCTAGTT 3'; Martin et al. 1992) were used to amplify portions of the cytochrome b gene and 12S rRNA from four fish (two from each site), the tRNAs flanking the control region, and the entire control region by PCR (Figure 1). Appropriate PCR-products were gel-purified (QIAquick Gel Extraction Kit, Qiagen, Inc.), quantified, and sequenced. Species specific primers in the tRNA-pro (WahPro, 5' CCTACCCCTAACTCCCAAAGCTAG 3') and tRNA-phe (WahPhe, 5' GCTTTCTAGGGCCCATCTTAACAT 3') were designed from sequence information and used to amplify the entire control region from additional fish by PCR (Fig. 1). PCR amplification was conducted for replicate 50 μL reactions containing 200 ng template DNA, 1.5 mM $MgCl_2$, 200 μM each deoxynucleotide triphosphates (Promega, Inc.), 0.4 μM of each primer, and 3.5 units of *Taq* DNA polymerase with 10x PCR buffer (Amersham Life Science). Cycling parameters utilizing wahoo primers were a 3 min denaturation at 94°C, followed by 35 cycles consisting of 0.5 minutes at 94°C, 0.5 min at 58°C, and 1.5 minutes at 72°C, and a final elongation of 7 minutes at 72°C. Appropriate PCR-products were gel-purified, quantified, and sequenced. All sequencing was conducted at the University of Maine DNA Sequencing Facility on an ABI model 373A automated sequencer.

In an effort to verify that the PCR product obtained using the species specific primers resulted from amplification of mtDNA and not a nuclear pseudogene, a nested PCR using control region primers from a previously published study (Lee et al. 1995) was conducted. In this procedure, gel purified products obtained from the amplification with CB3R and 12SAR were used as template for a second reaction with the afore mentioned internal primers designated as "A" 5' TTCCACCTCTAACTCCCAAAGCTAG 3' and "G" 5' CGTCGGATCCCATCTTCAGTGTTATGCTT 3' (Figure 1).

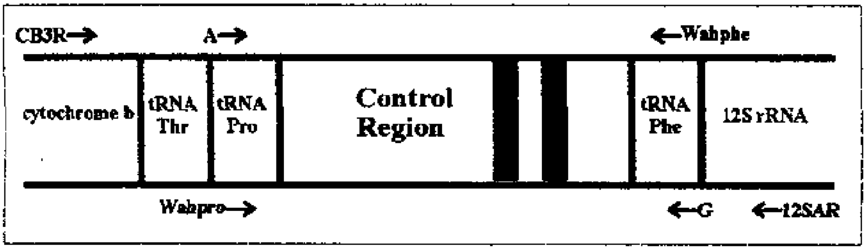


Figure 1. A view of a portion of the piscine mitochondrial genome. Arrows indicate the relative position of primers employed to amplify the control region of the wahoo, *Acanthocybium solandri*.

Secondary structures of the tRNAs threonine (thr), pro, and phe were elucidated using Sean Eddy's Lab tRNAscan-SE Search Server (Lowe and Eddy 1997). DNA sequences were imported into OMIGA, ver. 1.1 (Oxford Molecular Ltd., Oxford, England), a multiple sequence editor, and aligned with CLUSTALW (Higgins and Sharp 1988) using default settings. OMIGA was also employed to search for restriction endonuclease digestion patterns and assign each sequence to a distinct haplotype.

RESULTS

The universal primers CB3R and 12SAR produced a DNA-fragment of approximately 1800 base pairs (bp) in the four initial samples (Figure 2a). The complete control region sequence was obtained from two fish, one from each sampling site and deposited in GenBank (accession numbers AF197149 and AF197152). These sequences were compared against sequences deposited at the National Center for Biotechnology Information (NCBI) using the NCBI's BLAST WWW Server (Basic Local Alignment Search Tool; Altschul *et al.*, 1990), and the closest matches were obtained with the mtDNA control region of several species of fish including *Etheostoma zonale*, *Gymnocephalus cernuus* and *Gymnocephalus schraetser*. The sequence and structure of the flanking tRNAs were also identified in all four samples (Figure 3). This sequence information was utilized to design species specific primers located in the tRNA-pro (WahPro) and the tRNA-phe (WahPhe). These primers produced a DNA-fragment of approximately 950 bp from eight fish (Figure 2b). BLAST searches indicated these sequences were the mtDNA control region. Nested PCR of the gel-purified CB3R/12SAR product with the primers A and G produced a DNA-fragment of approximately 950 bp (Figure 2c), as anticipated.

The control region of ten fish ranged from 886 to 892 bp (GenBank accession numbers AF197149-58), with a consensus sequence of 897 bp (Figure 4). The consensus sequence contained 128 transitions, 22 transversions, and 20 insertions/deletions (indels). One termination associated sequence (TAS) was identified: TAS-I, located in the tRNA-pro (Figure 4). Two conserved sequence blocks (CSB), CSB-II and CSB-III, were identified at the 3' end of the control region, and a putative CSB-1 was identified at the 5' end (Figure 4). A hypervariable region, containing 100 variable nucleotides and three indels in 320 bp, was identified at the 5' end of the control region (Figure 4). Five direct repeats were identified, as well as, two indirect repeats (Table 1 and Table 2). Eight of the 30 restriction endonucleases used in computer-simulated digestions revealed polymorphism (Table 3a). Three of these eight (Hinf I, Rsa I, and Ssp I) revealed several morphs that were unique to each sampling site (Table 3b).

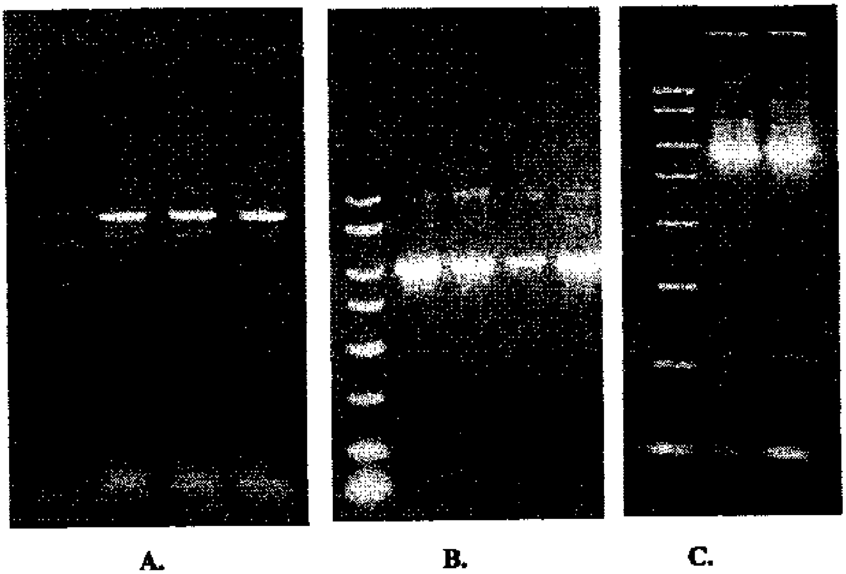


Figure 2. Agarose gels of the mitochondrial DNA control region of the wahoo, *Acanthocybium solandri*. The control region was PCR amplified using the following primer sets: (A) CB3R/12SAR, (B) WahPro/WahPhe, and (C) A/G (see Materials and Methods for details). Agarose Gel Ladder sizes: 2000, 1500, 1000, 750, 500, 300, 150, and 50 base pairs (Amresco PCR marker, Amresco, Inc.).

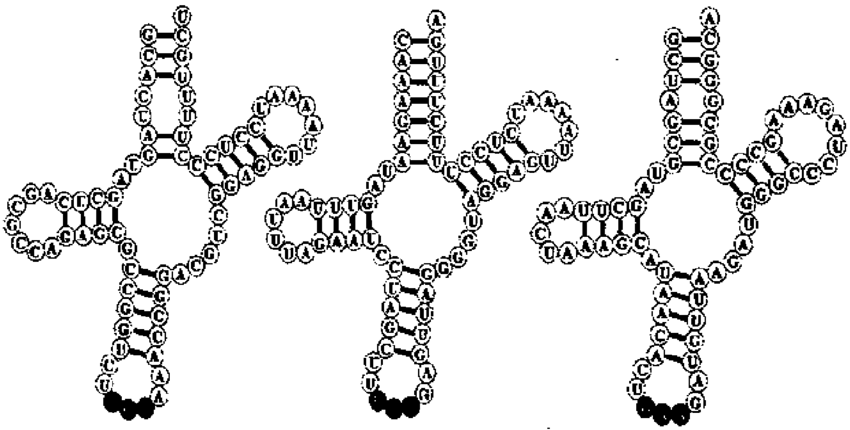


Figure 3. Sequence and structure of the mitochondrial tRNAs (from left to right), threonine, proline, and phenylalanine from the wahoo, *Acanthocybium solandri*. Black circles represent the anticodons.

```
                                            TAS I
                                CTAAATTAAA CTATTCTTTG PRO
    GCGACGCTCA CGCAATATAy AATAATATGT CTGTGAAGTA  40
    CATrTATGTA TTTACACCAT AyATTTATAT TAAyCATAyw  80
    AwTrCATATm TdGTACTCAw GGACATryry ATGyTdAAyr 120
    TACATryCTA TTCTyGrrTA yAATryTyrT ATTGCyyGAy 160
                             CSB I
   yyyAyTAGrA ryTAyATryG TACATAAACC AyAATrTGTA 200
    yTArryyyrk TayAyrhyrC TGATyTCwrG GayAGAyGAr 240
    ATTTAAGAyC TAGCACGAAA ACGTCCATrG rCAAATAyAT 280
    ACCAAGyAyT CAryAyCArr ATGrryyyny dyAAATyTTA 320
    ATGyAGTAAG ArCCTACCAT CAGTTGATTC CTTAATGCAT 360
    ACGGTTmTTG AAGGTGAGGG rCAATAryhG TGrGGGTyrC 400
    ACyyAGTGAA ywwTyCyyGG GCATTkGGGT TCCTAATTTC 440
    AGGTCCATyT rAyTTGAywy AyTCCyCATT CTTTCCTTGA 480
    CGCTGGCATA AGTTAATGGT GGAGTACATT rGACTCGTTA 520
    CCCACCATGC CGrGCryTCT TTCCATCGGG CATrGGkTTT 560
    yyyyTTTTkG GTTyCyTywC AyTTGrCATC CCAGAGTGCA 600
    CACrAAGAAT AGyTrAyAAG GTTGAACATT TCCTTGCTTG 640
    CAGrGTAAAT rTCATGAAGG GTGGAAAGAC TTTACrykAA 680
    rrArCCACAy ryTyGGATAT CAAGAGCATA ArTGAAATAT 720
    yACTCGAArA ATATyTAAGG TGCCCCCTCT CGGCTTTTGC 760
                        CSB II
  GCGTTAAACC CCCCTACCCC CCTAAAyTCG TGAAATATyT 800
                        CSB III
  AACACTCCTG AAAACCCCCC GTAAACAGGA AAACTCTCGA 840
    GTGGGGTATT TTATAGCCCA AAACGTATCT ATTTACATTA 880
    TTGTAAATAT TGCGTAT                          920
```

Figure 4. Consensus sequence of the mitochondrial DNA control region of the wahoo, *Acanthocybium solandri*. Termination associated sequences (TAS) and conserved sequence blocks (CSB) are in bold text. The 320 base pair hypervariable region is underlined. The top line is the 3' end of the tRNA proline.

Table 1. Direct repeats located in the mitochondrial DNA control region of the wahoo, *Acanthocybium solandri*. Repeats were identified from the consensus sequence of ten individuals; five from the northcentral Gulf of Mexico and five from Bimini, Bahamas. The minimum repeat length was seven base pairs (bp).

Number	Fragment from base	Repeated from base	Size of repeat (bp)	Repeat sequence 5' 3'
1	37	503	7	AGTACAT
2	49	869	8	TATTTACA
3	472	629	8	TTTCCTTG
4	766	812	9	AAACCCCCC
5	768	776	8	ACCCCCCT

Table 2. Indirect repeats located in the mitochondrial DNA control region of the wahoo, *Acanthocybium solandri*. Repeats were identified from the consensus sequence of ten individuals; five from the northcentral Gulf of Mexico and five from Bimini, Bahamas. The minimum repeat length was eight base pairs (bp).

Number	Fragment from base	Repeated from base	Size of repeat (bp)	Repeat sequence 5' 3'
1	10	887	9	ACGCAATAT
2	537	662	8	TCTTTCCA

Table 3a. Restriction endonucleases employed in a computer-generated restriction digestion of the mitochondrial DNA control region of the wahoo, *Acanthocybium solandri*. The procedure was conducted on each sequence for ten individuals; five from the northcentral Gulf of Mexico and five from Bimini, Bahamas.

Alu I**
BamHI
Bgl I
Bgl II**
Dra I
EcoRI
EcoRV
Hae III
Hind III
Hinf I**
Hpa I
Kpn I
Mlu I
Msp I**
Nco I**
Not I
Nru I
Pst I
Pvu II
Rsa I**
Sac I
Sal I
Sca I**
Sfi I
Sma I
Sph I
Ssp I**
Taq I*
Xba I
Xho I*

* denotes the presence of cuts without polymorphism between the ten sequences

** denotes polymorphism between the ten sequences

Table 3b. Unique haplotypes observed with respect to geographic location in a computer-generated restriction digestion of the mitochondrial DNA control region of the wahoo, *Acanthocybium solandri*. These haplotypes were generated from analysis of each sequence for ten individuals; five from the northcentral Gulf of Mexico and five from Bimini, Bahamas.

	Hinf I					Rsa I								Ssp I				
Location	A	B	C	D	E	A	B	C	D	E	F	G	H	A	B	C	D	E
Gulf	3	0	0	1	1	1	2	0	0	0	0	1	1	0	3	0	1	1
Bimini	2	2	1	0	0	1	0	1	1	1	1	0	0	4	0	1	0	0

DISCUSSION

The universal primers, CB3R and 12SAR, when used to amplify wahoo template, produced an approximately 1800 bp piece of DNA having significant similarity to the mtDNA control region and flanking tRNAs of numerous fish when compared with a BLAST search. After species specific primers were developed in the flanking tRNAs, a fragment of the expected length (approximately 950 bp) was produced showing high similarity to both the corresponding sequence of the original fragment and the mtDNA control region of other fish when compared using a BLAST search. The fact that the PCR product obtained with the species specific primers was the predicted length, alone, was a direct indication the designed primers were amplifying the desired portion of the mtDNA (Palumbi 1996). An additional precaution was taken to exclude the possibility that the PCR products obtained with the species specific primers were in fact mtDNA and not nuclear pseudogenes similar to those observed in other species of marine organisms such as sea urchins, crabs, and corals (Jacobs et al. 1983, Palumbi 1996). Following the concept of "long PCR" (Cheng et al. 1994a, 1994b), the 1800 bp piece of DNA produced using the universal primers CB3R and 12SAR was used as template for a second internal PCR with previously published fish mtDNA primers located in the tRNAs for proline and phenylalanine. The internal PCR product produced a band of the expected size, confirming the piece of mtDNA amplified was the desired control region sequence.

The length of the control region in wahoo was 886-892 bp which is comparable to the control region of other fish species. Lee et al. (1995) found the control region of 27 species of fish ranged from 856-1500 bp. In that study, cichlids and gadids contained the shorter control regions, and the control regions of pleuronectids were longest. Control region structure, a central conserved region flanked by more variable segments, was also similar to that reported for other fish species (Buroker et al. 1990, Johansen et al. 1990, Digby et al. 1992, Lee et al. 1995). We identified a TAS-I, as well as, a putative CSB-I, a CSB-II, and a CSB-III. The nucleotide sequences of these regions are similar to those of numerous fish (Table 4). CSB-I was conserved in the ten wahoo sequenced, but its location was atypical from other species of fish (Digby et al. 1992, Wilson et al. 1997) and, thus, was reported as putative.

Table 4. Comparative alignment of termination associated sequence (TAS) and conserved sequence blocks (CSB) identified in the wahoo, *Acanthocybium solandri*, and other fish species.

TAS-I		
	Wahoo	AAATTAAACTATT
	Trout[1]	AAATTAAACTACC
	Sturgeon[2]	ACATTAAACTATT
Putative CSB-I		
	Wahoo	ACATA
	Trout	ACATA
	Snook[3]	ACATA
CSB-II		
	Wahoo	TAAACCCCCCTACCCCC
	Trout	TAAACCCCCCTACCCCC
	Sturgeon	-AAACCCCC-TACCCCC
CSB-III		
	Wahoo	TG-AAAACCCCCCGTAAAC-A
	Trout	TGTTAAA-CCCC—-TAAACCA
	Swordfish[4]	TG-AAAA-CCCCC

[1]Digby et al. 1992; [2]Buroker et al. 1990; [3]Wilson et al. 1997; [4]Rosel and Black 1996

We also identified a 320 bp hypervariable segment at the 5' end of the control region that contained 100 variable nucleotides and three indels. Similar hypervariable segments have been observed near the 5' end of the control region of other fishes (Lee et al. 1995, Rosel and Block 1996, Garber 1999). Rosel and Block (1996) found a high level of polymorphism in the control region of swordfish collected from three different ocean basins. They concluded a DNA region less variable than the control region might be necessary to study the population structure of fish from the same ocean basin. Although, to arrive at such a conclusion with wahoo, would require the sequencing of more fish and an outgroup.

Eight of 30 restriction endonucleases produced polymorphic fragments and three of these enzymes; Hinf I, Rsa I, and Ssp I separated the ten wahoo sequences into unique haplotypes. A rapidly evolving region such as the mtDNA control region has the potential for use as a stock identification tool and/or a stock separation tool. If a large number of unique mtDNA haplotypes are observed, restriction endonucleases can be a suitable molecular tool to identify a stock, as well as, to evaluate the impact of a stock enhancement effort on reproduction and recruitment in subsequent generations (Gold and Richardson 1991; Gold et al. 1997, Wilson et al. 1997). Again, further wahoo samples need to be sequenced to arrive at such a conclusion.

Increased pressure has been placed on the wahoo due to its rising importance in commercial and sport fisheries. Therefore, obtaining information on its biology, life history, and genetics is essential to prevent a decrease in stocks beyond a sustainable level and the associated loss of genetic diversity. This study provides initial molecular genetic data that will prove useful in future genetic research.

ACKNOWLEDGEMENTS

We would like to thank Raul Miranda of the Bimini Big Game Fishing Club, Bobby Carter and the Directors of the Mississippi Gulf Coast Billfish Classic, and Jim Roberson of the Destin/Fort Walton Beach Sportfish Tournament for giving us the opportunity to collect wahoo. We would also like to thank Walter Grater for making editorial comments, and Dr. Gary and Tooie Garber for aiding with sample collection.

LITERATURE CITED

Altschul, S.F., W. Gish, W. Miller, E.W. Myers and D.J. Lipman. 1990. Basic local alignment search tool. *J. Mol. Biol.* **215**:403-410.

Buroker, N.E., J.R. Brown, T.A. Gilbert, P.J. O'Hara, A.T. Beckenbach, W.K. Thomas and M.J. Smith. 1990. Length heteroplasmy of sturgeon mitochondrial DNA: an illegitimate elongation model. *Genetics* **124**:157-163.

Chang, D.D. and D.A. Clayton. 1987. A novel endonuclease cleaves at a priming site of mouse mitochondrial DNA replication. *EMBO L.* **6**:409-417.

Cheng, S., C. Fockler, W.M. Barnes and R. Higuchi. 1994a. Effective amplification of long targets from clone inserts and human genomic DNA. *Proc. Natl. Acad. Sci. USA* **91**:5695-5699.

Cheng, S., R. Higuchi and M. Stoneking. 1994b. Complete mitochondrial genome amplification. *Nat. Genet.* **7**:350-351.

Clayton, D.A. 1991a. Nuclear gadgets in mitochondrial DNA replication and transcription. *Trends Biochem. Sci.* **16**:107-14.

Clayton, D.A. 1991b. Replication and transcription of vertebrate mitochondrial DNA. *Annu. Rev. Cell Biol.* **7**:453-478.

Collette, B.B. and C.E. Nauen. 1983. FAO species catalogue. Vol. 2. Scombrids of the world. An annotated and illustrated catalogue of tunas, mackerels, bonitos, and related species known to date. *FAO Fish. Synop.* 125, **2**:137p.

Digby, T.J., M.W. Gray and C.B. Lazier. 1992. Rainbow trout mitochondrial DNA: sequence and structural characteristics of the non-coding control region and flanking tRNA genes. *Gene* **113**:197-204.

Gallagher, S.R. 1994. Commonly used biochemical techniques, Appendix 3. Pages A.3.10-A.3.11 in: K. Janssen, (series ed.) Current Protocols in Molecular Biology. John Wiley and Sons, Inc., USA.

Garber, N.M. 1999. Application of the mitochondrial DNA control region in population structure studies of *Mugil cephalus* (striped mullet) in North America. M.S. Thesis. The University of Southern Mississippi, Hattiesburg, MS. 86 pp.

Gold, J.R. and L.R. Richardson. 1991. Genetic studies in marine fishes. IV. An analysis of population structure in the red drum (*Sciaenops ocellatus*) using mitochondrial DNA. *Fish. Res.* **12**:213-241.

Gold, J.R., F. Sun and L.R. Richardson. 1997. Population structure of red snapper from the Gulf of Mexico as inferred from analysis of mitochondrial DNA. *Trans. Amer. Fish. Soc.* **126**:386-396.

Higgins, D.G. and P. Sharp. 1988. Clustal: a package for performing multiple sequence alignment on a microcomputer. *Gene* **73**:237-244.

Jacobs, H.T., J.W. Posakony, J.W. Grula, J.W. Roberts, J.H. Xin, R.J. Britten and E.H. Davidson. 1983. Mitochondrial DNA sequences in the nuclear genome of *Strongylocentrotus purpuratus*. *J. Mol. Biol.* **165**:609-632.

Johansen, S., P.H. Guddal and T. Johansen. 1990. Organization of the mitochondrial genome of Atlantic cod, *Gadus morhua*. *Nucleic Acids Res.* **18**:411-419.

Lee, W., J. Conroy, W.H. Howell and T.D. Kocher. 1995. Structure and evolution of teleost mitochondrial control regions. *J. Mol. Evol.* **41**:58-66.

Lowe, T. and S.R. Eddy. 1997. tRNAscan-SE: a program for improved detection of transfer RNA genes in genomic sequences. *Nucl. Acids Res.* **25**:955-964.

Mahon, R. 1996. Fisheries and research for tunas and tuna-like species in the western central Atlantic. FAO Fish. Tech. Pap. No. 357. 61 pp.

Martin, A.P., R. Humphreys and S.R. Palumbi. 1992. Population genetic structure of the armorhead, *Pseudopentaceros wheeleri*, in the North

Pacific ocean: application of the polymerase chain reaction to fisheries problems. *Can. J. Fish. Aquat. Sci.* **49**:2386-2391.

Meyer, A. 1993. Evolution of mitochondrial DNA in fishes. Pages 1-38 in: P.W. Hochachka and T.P. Mommsen, (eds.) Biochemistry and molecular biology of fishes, Molecular Biology Frontiers, Vol. 2. Elsevier Science Publishers, Amsterdam, The Netherlands.

National Marine Fisheries Service. 1999. Commercial fisheries landings statistics, 1974-1998; Atlantic and Gulf. NOAA, Sustainable Fisheries Division, Miami, FL.

Palumbi, S.R. 1996. Nucleic acids II: the polymerase chain reaction. Pages 205-247 in: D.M. Hillis, C. Moritz and B.K. Mable, (eds.) Molecular Systematics, 2nd ed. Sinauer Associates, Inc., Sunderland, Massachusetts, USA.

Rosel, P.E. and B.A. Block. 1996. Mitochondrial control region variability and global population structure in the swordfish, *Xiphias gladius*. *Mar. Biol.* **125**:11-22.

Taggart, J.B., R.A. Hynes, P.A. Prodohl and A. Ferguson. 1992. A simplified protocol for routine total DNA isolation from salmonid fishes. *J. Fish. Biol.* **40**:963-965.

Wilson, R.R., Jr., K.A. Donaldson, M.E. Frischer and T.B. Young. 1997. Mitochondrial DNA control region of common snook and its prospect for use as a genetic tag. *Trans. Amer. Fish. Soc.* **126**:594-606.

Groundings and Spills: Addressing Anthropogenic Insults Through Natural Resource Damage Assessment and Restoration

JOHN W. ILIFF[1], LISA DIPINTO[2] and MARGARET MILLER[3]
[1]*National Marine Fisheries Service*
Restoration Center, SE Region,
9721 Executive Center Drive, N.
St. Petersburg, FL 33702 USA
[2]*National Ocean Service, Damage Assessment Center*
1305 East West highway
Silver Spring, MD, 20910 USA
[3]*National Marine Fisheries Service*
Fisheries Science Center
75 Virginia Beach Drive
Miami, FL 33149 USA

ABSTRACT

Hazardous material spills (including oil) and vessel groundings, including the ensuing salvage efforts, are examined in the context of natural resource damage assessment as acute anthropogenic activities resulting in both long and short term impacts to marine and freshwater fisheries and their habitats in particular. Two case studies and their restoration approaches are presented. The grounding of the M/V FORTUNA REEFER on the reefs of Mona Island, Puerto Rico resulted in approximately 2.75 hectares of coral injury. The grounding resulted in a discreet impact zone, but the salvage effort increased the injury to many times the size of the grounding area. Emergency restoration efforts at the site entailed the use of stainless steel wire and nails to reattach detached and broken colonies of Acropora palmata. A restoration status report is provided. A phosphate industry spill of approximately 189 - 211 million liters (50-56 million gallons) of process water containing phosphoric acid (pH 2) resulted in the instantaneous kill of over 1.3 million freshwater and marine fish in the Alafia River and Tampa Bay, Florida. The fish injury, including both the direct kill and the lost future somatic growth, was estimated at 64,892 kilograms of biomass lost. In addition, the spill injured approximately 377 acres of freshwater wetlands and contributed to nutrient loading in Tampa Bay. Restoration or creation of emergent estuarine habitat and/or reef creation are being considered to compensate for the fish injuries.

KEYWORDS: Assessment, restoration, spills

INTRODUCTION

Under several Federal environmental statues, NOAA is the principal federal Trustee for marine and estuarine natural resources in U.S. waters of the Gulf of Mexico and the Caribbean. As a Trustee, NOAA's Damage Assessment and Restoration Program (DARP) conducts natural resource damage assessments (NRDA) and restoration of coastal and marine resources injured as a result of oil spills, releases of hazardous materials and ship groundings where there is a threat of a spill, herein referred to as incidents.

NOAA's trust resources include living marine and estuarine resources and the physical habitats that support them. Commercially and recreationally important species as well as species that often seem inconsequential to the public are covered by NOAA's stewardship mandate. Additionally, this trust responsibility encompasses intangible services that flow from one natural resource to another or from natural resources to humans. Such services may include habitat services in the case of resource to resource services, or recreational services in the case of human use services.

Stewardship for these marine and estuarine resources and services is shared by other government and tribal entities in the Gulf and Caribbean. In the coastal states of the Gulf of Mexico and Puerto Rico and the Virgin Island's territorial waters, the Governors are the natural resource Trustees. However, this Trustee responsibility is typically delegated down to the head of the natural resource agencies within each government. It is at the natural resource agency level that NOAA works with each government to resolve liability from spills and groundings with potentially responsible parties.

In the past 25 years, Congress and the President have enacted a suite of environmental laws to address the degradation of the Nation's natural resources. Explicit statutory authority to restore injured natural resources began with the Clean Water Act amendments of 1977 and continued with the later enactment of the Comprehensive Environmental Response, Compensation and Liability Act (CERLCA or Superfund), the National Marine Sanctuaries Act (NMSA), and the Oil Pollution Act of 1990 (OPA) and other related laws. As the primary Federal natural resource Trustee for coastal resources, NOAA has responsibility for ensuring the restoration of coastal resources injured by releases of hazardous materials and of national marine sanctuary resources injured by physical impacts. The Clean Water Act, CERCLA and OPA mandate that parties that release hazardous materials and oil into the environment are responsible not only for the cost of cleaning up the release, but also for restoring any injury to natural resources that resulted. The National Marine Sanctuaries Act mandates that parties who destroy, cause the loss of, or injure sanctuary resources are responsible for their restoration.

The damage assessment and restoration process provides the framework for determining:

i) what resources have been injured and what is the loss to the public?; and

ii) how can the resources be restored and what type and amount of restoration is appropriate?

The process involves three overall steps: preassessment, restoration planning (including injury assessment and selection of restoration activities) and restoration implementation.

Preassessment

NOAA and other Trustees evaluate whether injury was sustained by examining the resources at risk, the nature of their exposure, potential and direct observations of injury. Trustees begin by coordinating with response agencies to determine whether response actions are sufficient to eliminate the threat of ongoing injury. If injuries are expected to continue, and feasible restoration alternatives exist to address such injuries, Trustees proceed to conduct an assessment.

Restoration Planning

Efforts during this phase are directed at evaluating potential injuries to determine the need for, and scale of, restoration actions. Two closely coordinated activities take place during this phase: injury assessment, to determine the nature and extent of injuries to natural resources and services; and restoration selection, to select restoration action(s) which will compensate the public for the loss of natural resources and services resulting from the incident. Trustees evaluate the alternatives available for returning the injured resources and services to their baseline condition (the condition of the resource had injury not occurred) and compensating for the loss from the onset of injury until recovery. The responsible party is liable for paying the cost of restoration plus reasonable assessment costs.

Restoration Implementation

NOAA works with co-Trustees and, in some circumstances, responsible parties, to design and implement restoration actions. Restoration plans are developed and presented for public comment before implementation unless an emergency situation exists. All restorations include monitoring provisions to allow for corrections, to measure progress and determine the restoration effort's overall success. In many cases, the responsible party assumes responsibility for implementing the restoration with Trustee oversight.

The ultimate goal of a damage assessment is to restore injured coastal and marine resources. The process outlined above ensures an objective and cost-effective assessment of damages to the benefit of both the party responsible for injury and the public interest.

The case studies below illustrate how NOAA is addressing its stewardship responsibilities for restoring injured resources in a coral reef grounding incident and in an incident resulting in the spill of hazardous material.

METHODS

M/V Fortuna Reefer Grounding

In July 1997, the 99 meter (325-foot) container ship Fortuna Reefer ran aground on the fringing coral reef surrounding Mona Island, Puerto Rico. This natural resource damage action was pursued under the Oil Pollution Act of 1990 because natural resource injuries resulted from response actions taken to abate a substantial threat of discharge of oil from the grounded vessel to sensitive habitats. The Commonwealth of Puerto Rico is a Trustee along with NOAA.

Preassessment activities focused on determining the extent and degree of coral injury but were otherwise limited. Biologists from the Trustee agencies and consulting firms representing the vessel owners and their insurance company jointly examined the injury area using SCUBA and aerial photography. The injury site was dominated by a well-established community of Elkhorn coral, *Acropora palmata*, at a depth of 3-9 meters. Visual observations by divers determined that injury was caused by two factors:

i) The pressure exerted by the weight of the vessel; and,

ii) The use of steel tow cables by tugboats during salvage operations.

The pressure of the vessel crushed and fractured the reef framework leaving very few living corals within the impact footprint. Those few colonies that survived complete destruction were broken into many fragments. The steel tow cables attached to the stern of the freighter which were used to pull the freighter free were dragged across the bottom. The cable severed and broke the standing coral colonies over a 2.75 hectare (6.8 acres) area as it dragged across the bottom.

The need to address ongoing loss of live tissue from broken coral fragments from wave and surge action was identified as a priority concern. Further injury assessment studies were not immediately undertaken. After evaluation of the preliminary injury assessment data, a cash settlement for $1.25 million dollars was quickly reached between the responsible party and the Trustees to fund restoration. The quick settlement eliminated the need for more lengthy and costly injury determinations and focused attention on emergency restoration planning and implementation, and compensatory restoration.

The primary objective of the emergency restoration was to salvage and stabilize live coral fragments and minimize ongoing coral mortality. Without stabilization of the fractured coral pieces, coral tissue quickly abrades away in the high energy environment of Mona Island's fringing reef. The long-term goal of the restoration is to provide the best possible conditions for the coral to naturally re-establish its structural complexity to baseline conditions.

The coral stabilization consisted of immobilizing loose and broken branches of elkhorn coral to the reef buttress and to existent relic reef framework with stainless steel wire and stainless steel nails. Due to the density and hardness of the reef structure, the procedure involved drilling holes into the reef, driving nails into the holes and wiring the broken and/or loose corals to the hard substrate. Use of cement was first identified as the preferred stabilization methodology but rejected due to the strong underwater surge created by ocean swells. Plastic tie wraps were also used to secure smaller pieces of coral; however, the wave surge at the site loosened the tie wraps. As a result, corals that were secured with tie wraps were further stabilized with wire.

To track the success of the restoration and determine the need for mid-course corrections, monitoring stations were established. Monitoring was designed to measure mortality rate of transplants, survival rate on different substrate types used as transplant sites, success of different transplant sizes, coral re-attachment to the substrate and percent of remaining live tissue cover. Three monitoring attempts were thwarted by poor weather and only limited data was collected. A fourth monitoring effort was conducted in 1999 and data are currently being analyzed.

RESULTS

Within a 2.5 month period after the grounding, all restoration work was complete. At the conclusion of the restoration effort, 1,857 coral fragments had been stabilized over the 2.75 hectare area (Spadoni 1997). This total includes 653 small (25 cm - 50 cm), 869 medium (50 cm- 1.0 m), and 335 large (>1.0 m) fragments.

Assessment of stabilized fragment status in summer 1999 showed that over half of the fragments still contained live tissue (Table 1). An analysis of variance indicated that large fragments (>1m maximum length) had significantly more remaining live tissue than small fragments (25-50 cm) as shown in Figure 1. Additional observations showed both positive and negative outcomes. *A. palmata* was observed overgrowing the stainless wire with no adverse impacts. Other positive signs of success were active tissue growth, new proto-branches (suggesting net upward growth), and for some corals, active coral re-attachment to the substrate. Some wire failure was observed around the site and is the most significant threat of long term project failure. Analysis of wire collected from the site after two years in service indicates severe pitting that leads to accelerated corrosion and ultimately wire failure. The rate of wire failure is much faster than the coral's observed ability to re-attach to the substrate. Only 14% of corals examined during monitoring have reattached to the substrate and the bonds that have formed will not provide long-term stability for years to come. Because wire failure will outpace coral re-attachment, a mid-course correction is being considered over the entire site.

Table 1. Summary of mortality monitoring data fro August 1999 (Bruckner and Bruckner, unpubl.)

Total # Fragments Monitored	Living Fragments	Missing Fragments	Dead Fragments
705	405 (57%)	118 (16.7%)	182 (25.8%)

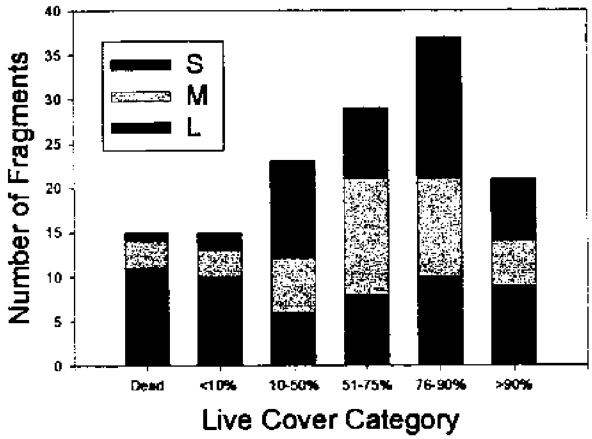


Figure 1. Percent live cover (tissue) vs size class. Bars represent large fragments (>1 m in length), medium fragments (>50 cm - 1 m), and small fragments (25 - 50 cm). Large fragments fared significantly better than small fragments with medium fragments doing intermediately well (1-way ANOVA; p = 0.026, n = 42-54

DISCUSSION

This emergency restoration action utilized a novel attachment technique to meet the requirements of a rigorous environment. Based on monitoring data, over half of the restored fragments persist and have live tissue almost two years after the incident. The site has also survived hurricane Georges, a class three storm. If no emergency restoration had been undertaken, it is likely that none of the fragments would have survived (Bruckner and Bruckner unpublished data). Wire failure did occur and further analysis of the wire issue has raised the possibility of a mid-course correction. Assessment has also indicated that large fragments are better candidates for restoration than small ones.

METHODS

Alafia River Acid Spill

On December 7, 1997, the wall of a phosphogypsum stack breached at the Mulberry Phosphates Inc. (MPI) phosphoric acid/fertilizer production facility in Mulberry, Polk County, Florida. Approximately 189 - 211 million liters (50 - 56 million gallons) of acidic process water flowed through the breach, overflowed return and collection systems and flowed into the Alafia River. Over the course of the next week to 10 days, the volume of released process water traversed approximately 58 km (36 miles) of the river to Tampa Bay. The process water contained about 1.5% phosphoric acid, a hazardous substance under CERCLA, and exhibited a pH of approximately two standard units.

One of the primary categories of natural resource injury was a significant fish kill in the Alafia River. Due to space limitations, injuries to other natural resources will not be presented here.

The fish kill injury determination consists of the sum of two components: the estimate of instantaneous mortality resulting from the spill (i.e., the direct kill) and the future somatic production or growth normally to be expected of the killed organisms over the remainder of their life span (i.e., lifetime production). That is, in addition to the direct mortality, if the spill had not occurred, the killed organisms would have continued to grow until they died naturally or to fishing. This lost future (somatic) production is estimated and added to the direct kill injury. The total is the total production lost, and is determined on a biomass basis (kg).

Preassessment data gathering focused on estimating the magnitude of the instantaneous fish kill (including blue crab and pink shrimp) which resulted from exposure to the acidic process water. Biologists representing both the Trustee agencies and MPI conducted sampling in the lower, tidally-influenced portion of the Alafia River. Due to the location of the dead fish at the time of response, all sampling efforts were conducted within the tidally-influenced portion of the river, from the mouth of the river to river km ~16. Three types of data were collected:

i) smaller animal seine and trawl data,
ii) larger animal visual survey data, and
iii) larger animal "clean-up" data, and are described below.

For the seine and trawl and larger animal visual survey data collections, basic area sampling principles were applied. Dead fish observed in randomly selected areas are counted and measured; these counts are then expanded over the entire affected area to provide an estimate of the total number of large dead fish present in the study area.

Seine and Trawl Sampling

Smaller animal data was collected by Florida DEP's Florida Marine Research Institute, Fisheries-Independent Monitoring Program (DEP/FIM) using methods consistent with an existing seine and trawl sampling program. Following DEP/FIM protocols, small-mesh seines and trawl data were used to assess juvenile populations of larger species and juvenile-to-adult populations of smaller species (< 8 cm total length).

A stratified random sampling design was used for sample site selection. The seine stratum included shoreline areas with water depths less than 1.8 m, assumed to be representative of the shoreline community. The trawl stratum included non-shoreline areas with water depths greater than 1.0 meter, assumed to be representative of the river channel community. All fish were identified to the lowest practical taxonomic level and counted, and representative length frequencies were recorded. A total of 14 seine and 5 trawl samples were collected during the sampling effort.

Visual Surveys

Data on larger animal (>8 cm total length) mortalities were collected by visual surveys. Floating and beached fish specimens in the tidally-influenced segments of the river were sampled following the American Fisheries Society (AFS) visual survey protocols (AFS 1992) for the estimation of fish kills. In this assessment, the lower Alafia was divided into six segments, and each segment was divided into countable units, or transects. A total of 40 transects were counted in the lower portion of the river. Expansion factors were derived from the area covered by the surveyors in a given river segment, relative to the total area in that segment.

Larger Animal Clean-up Data

Dead fish were removed from the river by fish clean-up contractors, and collected in roll-off boxes for disposal. Data from this larger animal clean-up effort was provided by Florida Game and Freshwater Fish Commission (FGFC) based on their examination of the dead fish removed from the river. Data on species composition, numbers, length frequencies and average weight was recorded by FGFC for subsamples of the dead fish, as well as the total weight of all dead fish removed from the river.

RESULTS

The data from these three preassessment activities were compiled and used by FMRI to estimate mortalities for both smaller and larger animals.

Smaller animal mortality, the estimate of smaller fish, blue crab and pink shrimp killed, was derived from consideration of the seine and trawl data using an "observed mortality method". This method estimates the population of dead animals in the lower portion of the river sampled, based on data gathered from seine and trawl data on December 12, and is calculated as the number of each species collected per area sampled (e.g., catch per unit effort reported as number/m^2). The mean population estimate for dead animals (following stratified random sampling) was then calculated following Snedecor and Cochran (1967). Lower and upper mortality estimates for the observed mortality method were calculated by either subtracting (for lower estimate) or adding (for upper estimate) the standard error to the mean dead-animal population estimates. Lower, mean and upper dead animal population estimates were multiplied by the total area of the segments used in the analysis to estimate the total number of small dead animals in the lower portion of the river. The data and the methods used by FMRI to calculate these estimates are presented in detail in the DEP/FMRI report dated December 10, 1998. Smaller fish and shellfish killed estimated by this method is 1,244,800 (mean).

The estimate of larger fish killed is the sum of two estimates:

i) the number of dead fish present in the surveyed portion of the river, as calculated using the visual survey data following AFS methods for estimating fish kills, and
ii) the number of additional dead fish removed from the river by clean-up contractors, as calculated using the larger fish clean-up data provided by FGFC.

These estimates were 57,900 and 15,000, respectively for a total of 72,900 large fish killed.

While the impact on each species was locally significant, loss of future production and recruitment associated with the estimates of the direct kill are unlikely to be large enough to significantly alter future populations in the river. The Trustee agencies believe that production from unaffected organisms and recruitment from unimpacted tributaries, upstream areas, and Tampa Bay will provide sufficient egg and young production to sustain populations of fish injured by the spill. Under these circumstances, further studies to assess an impairment of reproductive capacity are not required.

The loss of future productivity associated with the estimates of direct kill was calculated based on information contained in the biological database in the CERCLA type A model, Natural Resource Damage Assessment Model for Coastal and Marine Environments (NRDAM/CME, Version 2.5), other information augmenting the database for species killed by this spill, and the population model component in the NRDAM/CME model to predict the duration of such losses.

Under this approach, based upon species composition, the direct kill is quantified by age class using standard population models. The net (somatic) growth normally to be expected of the killed organisms is computed and summed over the remainder of their life spans (i.e., lifetime production), and future interim losses are calculated in present day values using discounting at a 3% annual rate. The survival rates per year and length-weight by age relationships are used to construct a life table of numbers and kg for each annual age class. Lifetime production is estimated as the sum of the net somatic growth normally to be expected of the killed individual over the remainder of its life span. The age-class specific weight gain per year times the percent expected to be left alive by the end of that year is summed over all years to calculate total lifetime production. The total injuries by species for direct kill and production foregone is summarized in Table 2 .

DISCUSSION

This assessment approach facilitates restoration planning in that it allows the Trustees to select one or more restoration projects that will produce an equivalent biomass of fish lost, with scaling based on secondary productivity estimates (i.e. kg production/acre/year). In addition to determining the extent of injury, DEP/FIM's regular sampling of the estuarine portion of the river under its historic sampling program continues and is an ongoing source of information for use in monitoring the recovery of small species populations and juvenile populations of larger species post-spill.

The restoration goal for the all injury categories is to restore, replace or acquire natural resources or services like those injured as a result of the spill as a basis for compensating for the interim losses of natural resources and resource services which occurred. The restoration objective for fish injuries is to replace the biomass of fish, crabs and shrimp lost due to the spill through creation or enhancement of habitat(s) capable of generating an equivalent biomass over time. Restoration planning therefore, is several habitat types capable of providing the lost fishery biomass. Emergent marsh (*Spartina alterniflora*), oyster reefs and artificial reefs are among the habitats likely to play a part in the fish restoration.

Table 2. Summary of total injuries by species, number killed and estimates of direct biomass loss and production foregone. (French 1999)

Species	kill (#)	kill (kg)	Production Foregone (kg)	Total Injury (kg)
bay anchovy	1,107,745	1,329	160	1,489
gizzard shad	112	13	8	20
Menidia (silverside)	19,465	14	6	20
gulf killifish	3,013	3	1	5
rainwater killifish	4,954	5	2	7
sheepshead minnow	2,107	2	1	3
snook	2,389	2,055	2,892	4,947
spotted seatrout	602	56	8	64
sand seatrout	17,930	136	28	164
kingfish	956	83	13	96
ladyfish	1,925	321	490	810
red drum	628	226	1,639	1,865
hogchoker	48,292	72	6,785	6,857
grunt	19	2	1	3
mullet	1,219	188	76	264
sheepshead	10,253	1,743	3,956	5,699
mojarras	53,280	6,926	17,375	24,302
blue crab	6,828	816	135	951
pink shrimp	2,941	204	0	204
common carp	19	52	52	104
bullhead catfish	8,340	835	1,920	2,755
channel catfish	168	39	42	81
other catfish	4,516	1,032	1,117	2,150
gar	7,641	5,977	3,645	9,622
largemouth bass	709	220	82	302
sunfish	5,988	240	591	831
butterfly ray	19	76	0	76
gafftopsail catfish	37	50	0	50
Gobiosoma (goby)	4,113	21	0	21
spadefish	38	17	0	17
gulf toadfish	39	2	0	2
sailfin molly	1,170	12	0	12
tilapia	1,805	1,101	0	1,101
Total	1,319,260	23,868	41,024	64,892

ACKNOWLEDGEMENTS

Thanks to Harold Hudson and Bill Godwin of NOAA Florida Keys National Marine Sanctuary and James Timber of the Puerto Rico Department of Natural and Environmental Resources for their long term commitment to the Fortuna Reefer restoration. Also, thanks to Robin and Andy Bruckner (NMFS) who, along with Earthwatch volunteers, collected valuable monitoring data on the Fortuna Reefer restoration project.

LITERATURE CITED

American Fisheries Society. 1992. Investigation and valuation of fish kills. American Fisheries Society Special Publication 24. 96 pp.

Florida Marine Research Institute. 1998. Assessment of Fish, Blue Crab, and Pink Shrimp Mortality in the Tidal Portion of the Alafia River Following the December 1997 Process Water Spill. December 10, 1998.

French, D.P. 1999. Estimating Injuries Resulting from the December 1997 Process Water Spill into the Alafia River, June 25, 1999. Unpubl. MS.Snedecor, G.W., and W.G. Cochran. 1967. Statistical. H. [1997] Report on the Reef Repair Project at Mona Island, Puerto Rico. Coastal Planning and Engineering, Boca Raton, FL. 9 pp. Unpubl. MS.

Selecting Biodegradable Fasteners and Testing the Effects of Escape Panels on Catch Rates of Fish Traps

NEETHA SELLIAH[1], HAZEL OXENFORD[1]
and CHRISTOPHER PARKER[2]
[1]*Marine Resource and Environmental Management Programme*
University of the West Indies
Cave Hill, Barbados
[2]*Fisheries Division, Ministry of Agriculture and Rural Development*
Princess Alice Highway
Bridgetown, Barbados

ABSTRACT

The 1998 Fisheries (Management) Regulations of Barbados require that all fish traps be fitted with an escape panel and carry an identification marker. The first part of the study tests various biodegradable escape panel fasteners to find one that will open in no less than three weeks and no more than five weeks. Fasteners were made of materials which were low cost and readily available, and included: cane lily, coconut bark, caulking, no. 6 cotton, cane trash, paper ribbon, and hemp twine. The second part of this study compares the catch rates of conventional fish traps with identical traps fitted with escape panels and identification markers (regulation traps). This was done using three commercial trap fishers over a 5-month period (January - May 1999), each fishing a different reef habitat using an equal number of conventional traps and regulation traps fished side-by-side. All trap hauls were recorded by number, size and species of fish taken. Hemp twine was found to be the most suitable fastener for the escape panel. Catch rates and taxonomic composition of catches were generally unaffected by the regulation specifications, suggesting that they will be readily accepted by fishers as a tool to reduce ghost fishing by lost traps.

KEY WORDS: Catch rates, fish traps, management measures.

INTRODUCTION

The trap fishery in Barbados, which targets nearshore coral reef fish species, using mainly wire mesh z-shaped and arrowhead Antillean traps fished from small open boats (moses) with outboard engines, is of significant cultural and economic importance, despite the relatively low landings (annual catch represents an average (1994 - 1998) of 0.4% of total fish landings).

The fishery has a long tradition in Barbados, with fishers reporting at least three generations of trap fishers. The trap fishery is particularly important in providing employment and food fish during the summer (July - October) which represents the pelagic fishery off-season. There are approximately 50 trap fisher captains and at least 50 additional crew members who engage in trap fishing during the summer and 15 - 20 of these trap fish year-round (Selliah in press).

Trap fish landings during the pelagic off-season average (1994 - 1998) 6.5 mt which represents 3% of total landings during this period. However, at some of the island's 31 landing sites (e.g. Pile Bay) trap fish landings represent as much as 36.6% of summer fish landings and 12.6% of total annual fish landings. The fishery also has important links with the tourism industry, providing an exotic array of food fish in the markets and for the restaurant trade.

The trap fishery resource in Barbados, like many other countries in the Caribbean, is believed to be over-exploited, particularly along the south and west coasts where trap fishers are concentrated (Mahon and Drayton 1990, Fisheries Division 1997, Selliah in press). The recent Fisheries Act 1993 - 1996 with accompanying Fisheries (Management) Regulations 1998, and the first ever 1997 Fisheries Management Plan for Barbados have attempted to address this problem by the introduction of a number of new management measures to control fishing mortality in the trap fishery. These include *inter alia* the introduction of a minimum legal mesh size regulation of 3.18 cm (1.25") and a plan to increase that to 3.80 cm (1.5") within the next two years, and perhaps incrementally to 5.1 cm (2.0") at some future time if deemed necessary to further reduce fish mortality, particularly on immature fish (Fisheries Division 1997). A second regulation states that every fish trap shall be fitted with an escape panel of a size and design approved by the Chief Fisheries Officer. This is to reduce the incidence of continued fishing mortality when traps are lost (ghost fishing). A third regulation states that all traps should be marked for identification in a manner approved by the Chief Fisheries Officer. This is to assist in the enforcement of other regulations related to the trap fishery.

Several studies have examined the effects of increasing mesh size in various Caribbean trap fisheries including Jamaica (Munro 1983, Nicholson and Hartsuijker 1983, Sary et al. 1997), Southern Florida (Bohnsack et al. 1989), Saba Bank (Wolf and Chislett 1974), St. Thomas (Olsen et al. 1978), Puerto Rico (Stevenson and Stuart-Sharkey 1980, Rosario and Sadovy 1991), Guadeloupe (Beliaeff et al. 1992), and Barbados (Robichaud et al. 1999). However, there have been few, if any, studies on the potential effects of using escape panels and or identification marks on the catch rates of fish traps, and none in the Barbados trap fishery. Furthermore, biodegradable panel designs have not been tested for approval by the Chief Fisheries Officer of Barbados.

This study attempts to test various biodegradable fasteners on a sprung mesh door (the Modified Dowridge Design proposed by the Fisheries Division) for suitability in providing fish traps with an effective escape panel. This study also investigates the effects of the chosen fasteners and escape panel design and the approved identification marks on the catch rates of trap fishers in Barbados.

METHODS

Escape Panel Design

The design of the escape panel (the Modified Dowridge Design) was developed by the Fisheries Division in Barbados and consisted of a sprung mesh door held closed by biodegradable fasteners, and designed to spring open into the trap (thereby avoiding any damage to adjacent coral) when the fasteners degrade (Figure 1). The door consisted of a square wooden frame covered by 1.25" mesh wire. It was set against a wooden frame measuring approximately 14 x 14", but specifically 2" larger all around than the diameter of the entry funnel, built into the side panel of the trap. The door was hinged at the bottom with strapping wire, weighted at the top with a small lead strip, and fastened at the top with two strips of biodegradable material (Figure 1). The springing mechanism comprised a 2 cm wide strip of tyre inner tubing tied to the top of the door and to the bottom of the trap at a 45 degree angle and pulled taut when closed (Figure1).

Selecting and Testing Biodegradable Fasteners

The criteria for selecting the biodegradable fasteners followed those of Kumpf (1980) in that the material should be relatively inexpensive, readily available locally, simple to replace, and have some predictable durability. In this case, we were looking for a durability of no less than three weeks and no more than five weeks. Seven different fastening materials were selected (dried cane lily, coconut bark, cane trash, caulking, no. 6 cotton, hemp twine, and paper ribbon) for *in situ* durability testing.

In situ tests were conducted over a four month period (June-October 1998), using two z-shaped fish traps which were modified to carry six escape panels each. Up to four different biodegradable fasteners were tested simultaneously and these were randomly allocated to the 12 escape panels so that each fastener was tested in triplicate. The two traps were set side-by-side in a shallow (6.5 - 7.5 m depth), relatively calm area, approximately 0.5 km from shore in Carlisle Bay on the west coast of Barbados (Figure 2). They were hauled and checked every five to six days. Fasteners that had not biodegraded after five weeks were deemed unsuitable and replaced with new materials for testing.

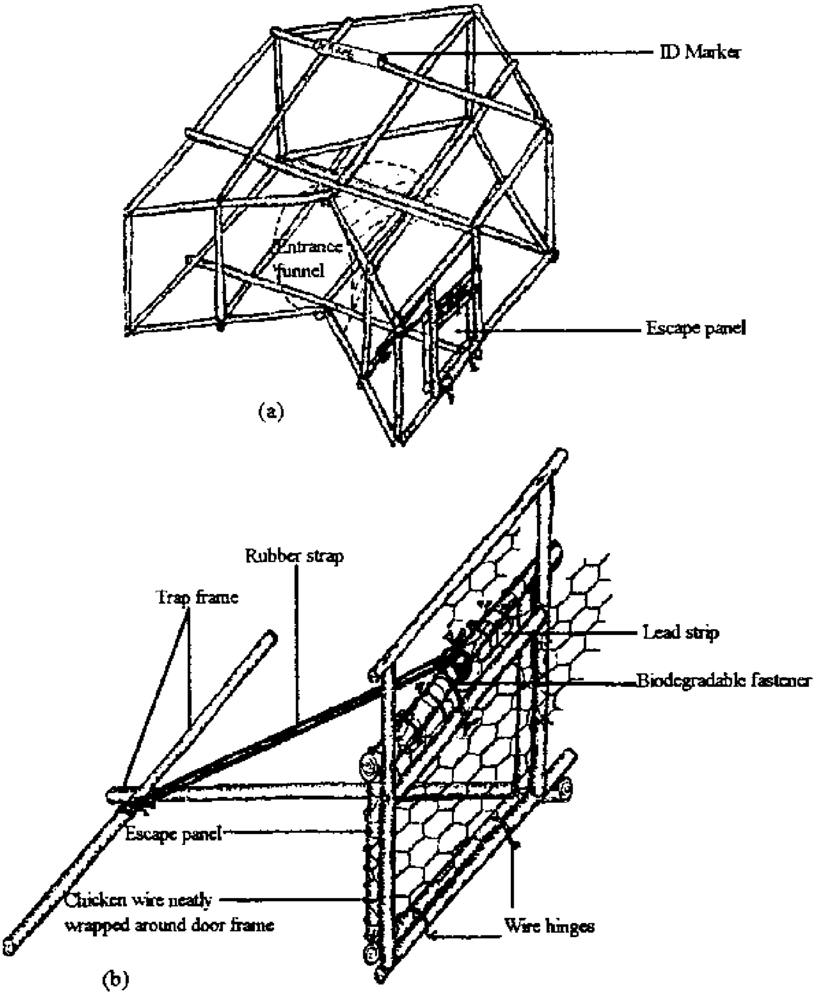


Figure 1. Diagram of a regulation fish trap showing: (a) positions of the identification marker and escape panel; and (b) details of the Modified Dowridge Design escape panel.

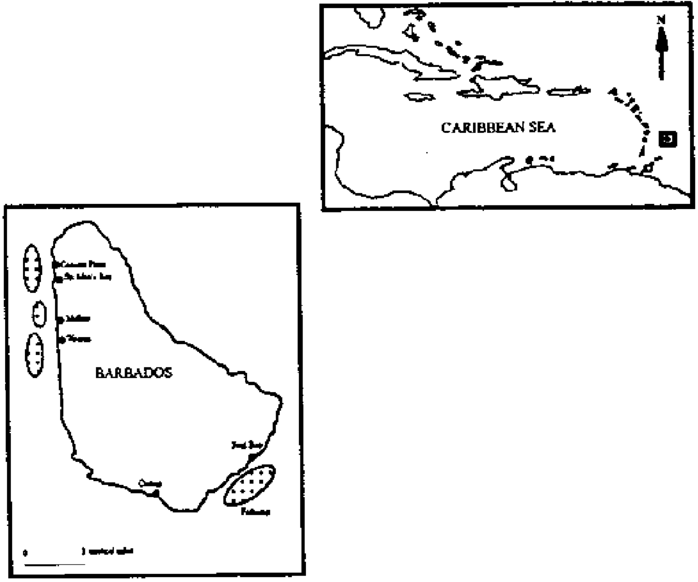


Figure 2. Locations of the fishing grounds and reefs used in the study to compare the catch rates between regulation and conventional traps.

Figure 2. Map of Barbados showing the location of study sites and fishing grounds.

Comparison of Catches between Experimental Regulation Traps and Conventional Traps

Site selection — Three sites (Six Men's, Weston, Oistins; Figure 2) were chosen along the west and south coasts of Barbados where trap fishing activity is concentrated. These three sites provide a good representation of the different sea conditions and types of coral reef habitat used by trap fishers in Barbados.

Six Men's, at the north end of the west coast, is characterised by calm seas (leeward shore). Nearshore to the south (off Mullins) there are fringing reefs extending up to 200 m from shore and interspersed by sandy areas, and to the north (off the Arawak Cement Plant) there is a flat rock reef platform dominated by a mixture of soft and hard corals. Offshore (at about 1 km) is a continuous coral-rich bank reef at 18 - 50 m depth.

Weston, in the centre of the west coast, is also calm and characterised to the north and south by shallow fringing reefs extending up to 200m from shore and interspersed by sandy bays, offshore deep (15 - 40 m) patch reefs dominated by rubble, hard corals and sponges, and a deep (18 - 50 m) coral-rich bank reef approximately 1 km from shore.

The fishing grounds off Oistins extend along the southeast (windward) coast and are characterised by choppy seas and a coral-rich relatively shallow bank-barrier reef system located 0.5 - 1 km from shore.

Fishing gear and methods — One full-time (year-round) trap fisher from each site was invited to assist in the study, and asked to build a replicate set of traps (identical to their own conventional traps) to which the regulation escape panel with biodegradable (hemp twine) fasteners and the approved identification marker were then added (subsequently referred to as 'regulation traps').

Since the primary aim was to investigate possible effects of the new trap regulations on the catch rates, the fishers were instructed to continue fishing exactly as they normally would (but with the assistance of a researcher). This meant that trap design, soak times, depths and habitats fished were not standardised within or between fishers, but traps were fished side-by-side in pairs comprising one regulation and one conventional trap of the same design. This allowed paired-comparisons of catches by the two trap types, to be made. Fishing continued to the fishers' own schedules for a period of five months (January - May, 1999).

Specifically, six pairs of traps (arrowhead and z-shaped traps all measuring 2' deep x 4' wide x 8' long) were fished from Six Men's. The traps were set and hauled once or twice a week, from January to February and from April to May, 1999. Two distinct fishing grounds were used, one off the Arawak Cement Plant between 0.5 - 1 km from shore at a depth of approximately 25 m, the other was on the bank reef off Mullins Beach about 1 km from shore and at a depth of approximately 40 - 50 m (Figure 2).

At Weston, five pairs of arrowhead traps (measuring 2' deep x 5' wide x 4' long) were set and hauled once or twice a week, from January to April, 1999. The traps were set adjacent to fringing reefs less than 0.3 km from shore at a depth of approximately 6-8 m.

Eight pairs of z-shaped traps (measuring 2' deep x 5' wide x 10' long) were fished from Oistins. These traps were set and hauled twice a week, from January to April, 1999. Two distinct fishing grounds on the bank-barrier reef system were used, one off South Point lighthouse known as the Fathom located about 0.5 - 1 km from shore and at a depth of 19 - 20 m, the other was off Foul Bay, 0.5 km from shore and at a depth of 26 m (Figure 2).

Data Collection and Analyses

For every fishing day, the commercially valuable catch from each trap pair was collected as the traps were hauled, and was stored in separate marked buckets.

Fish of no commercial value were thrown back by the fisher. On landing, all finfish were identified to species and measured for wet weight to the nearest 1 g and for fork length (FL) to the nearest 0.5 cm. Crustaceans were noted, but not measured and not included in any analyses. Moray eels (which were generally used only for bait), scorpionfish (which were generally left untouched in the traps), and nurse sharks (which were thrown back by the fisher) were also noted but not measured and were not included in any analyses.

All catch data were stored in spreadsheets. The few missing weights for individuals were either filled on the basis of their length (by examination of other individuals in the data base of the same species and size) or on the basis of the mean size of that species landed at the given site. Non-parametric statistical analyses were performed throughout since not all of the data sets were normally distributed. Catch rates (kg/trap haul and no. fish/trap haul) were compared between regulation and conventional traps using Wilcoxon paired-sample tests, and in one instance, where some traps were not strictly paired, a Mann-Whitney test was used. Species composition of catches by weight were compared between regulation and conventional traps using Chi-square contingency tests to examine the top ten families taken. Mean individual fish weights were compared between regulation and conventional traps using Mann-Whitney tests.

RESULTS

Durability of Fasteners

The material deemed the most suitable for use as fasteners on the escape panel door was hemp twine, which degraded sufficiently to allow the door to open in all three replicates in 22 - 26 days. Other materials which showed some potential included two stranded caulking (fasteners on two out of the three doors degraded in 22 - 26 days; one door remained closed beyond five weeks), strips of cane trash (fasteners on one door degraded in 16 days, a second door opened after 22 - 26 days, the third door remained closed beyond five weeks), and paper ribbon (fasteners on one door degraded in 22 - 26 days, while two doors remained closed beyond five weeks). Materials deemed unsuitable as fasteners were dried cane lily, coconut bark, five stranded caulking and no. 6 cotton since all of these remained intact for five weeks or more.

Hemp twine was therefore chosen for use in the experimental regulation traps that were used to investigate the effects of the new regulation trap specifications on catch rates. The effectiveness of hemp twine as a biodegradable fastener was subsequently reconfirmed during the investigation when regulation traps were left in the water (on one occasion at Oistins) for a little over three weeks, and all doors were found to have opened when they were hauled.

Description of Catches

Altogether 28 fishing trips were taken over a period of five months (January to May, 1999). A total of 182 traps were hauled and 3,747 commercially valuable finfish weighing 663.13 kg were taken. The catch composition is summarised by species, by trap type and by site in Table 1. An additional 38 crustaceans including four species of crab, two species of lobster and a mollusc were also taken in the traps but were not considered further in this study (Table 1).

The commercially valuable finfish catch comprised 73 species from 26 families. The most common species (by weight) in the catches overall were princess parrotfish (*Scarus taeniopterus*), redband parrotfish (*Sparisoma aurofrenatum*), and whitespotted filefish which represented 11%, 10% and 9% of the total catch respectively (Table 1). The most common species by number of fish were princess and redband parrotfish and ocean surgeon (*Acanthurus bahianus*) which each accounted for 12% of the total number of finfish caught.

The catch composition is further summarised by family, trap type and site in Table 2. The most common family in the catches overall was Scaridae which accounted for 24% by weight and 26% by number of the total catch (Table 2).

The key characteristics and catch performance of the two trap types at each of the three sites is summarised in Table 3. Overall, the mean catch per trap haul was 3.31 kg (19 fish) for regulation traps and 4.00 kg (22 fish) for conventional traps. The mean individual size of fish taken was 173 g for regulation traps and 181 kg for conventional traps and did not differ between them (Mann-Whitney Test: U = 1.011, n = 1800+1947, P = 0.312; Table 3).

Comparison of Catch Rates between Regulation and Conventional Traps

As a result of obvious differences among sites in gear and habitats, analyses were initially done separately for each site.

Six Men's — Since two different trap designs were used by the Six Men's fisher, an initial comparison of their catch performance was undertaken using regulation traps. There was no significant difference between arrowhead and z-shaped traps in either the weight or number of fish per trap haul (Mann-Whitney test; for weight: U = 0.995, n = 15+15, P = 0.320; for numbers: U = 0.613, n= 15+15, P = 0.540). Therefore, all data including mixed design pairs were pooled for comparative analyses of regulation versus conventional traps.

Table 1. Catch composition of regulation and conventional traps sampled at three sites in Barbados between January and May 1999, shown by total weight (kg) and number of fish for each species.

GROUP		SIX MEN'S				WESTON				OISTINS				ALL SITES	
		Reg. traps		Con. traps		Reg. traps		Con. traps		Reg. traps		Con. traps		All traps	
Family	Species	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish
FINFISH															
Acanthuridae	*Acanthurus coeruleus*	0.51	6	0.22	4	1.84	15	0.21	1	6.26	59	22.74	146	31.78	231
	Acanthurus chirurgus	0	0	0	0	0.36	2	0	0	0	0	0	0	0.36	2
	Acanthurus bahianus	2.16	29	1.79	21	1.45	18	2.30	27	15.93	155	21.45	197	45.08	447
Aloustomidae	*Aulostomus maculatus*	0.48	2	0	0	0	0	0	0	0	0	0	0	0.48	2
Balistidae	*Melichthys niger*	0.28	3	0.14	2	0	0	0	0	0	0	0	0	0.43	5
	Balistes vetula	2.39	2	7.23	7	0	0	0.73	1	2.41	3	3.04	4	15.80	17
	Xanthichthys ringens	12.68	101	10.94	93	0	0	0	0	0	0	0	0	23.63	194
Bothidae	*Bothus lunatus*	0	0	0	0	0	0	0	0	0.41	1	3.07	2	3.48	3
Carangidae	*Caranx ruber*	0.6	5	1.03	9	0	0	0	0	0.23	3	4.13	27	5.99	44
	Caranx crysos	0	0	0	0	0	0	0	0	0	0	4.08	1	4.08	1
Chaetodontidae	*Chaetodon striatus*	0.33	6	0.48	8	0	0	0	0	0.81	17	1.34	28	2.96	59
	Chaetodon capistratus	0.15	3	0.13	5	0	0	0	0	0.3	6	0	0	0.58	14
	Chaetodon sedentarius	0.04	1	0	0	0	0	0	0	0	0	0	0	0.04	1
	Chaetodon ocellatus	0.03	1	0.05	2	0	0	0	0	0	0	0.09	1	0.17	4
Diodontidae	*Diodon holocanthus*	0	0	0	0	0	0	0	0	2.16	2	0.56	2	2.72	4
	Diodon hystrix	1.13	1	0	0	0	0	0	0	0	0	0	0	1.13	1
	Chilomycterus schoepfi	0	0	0	0	0	0	0	0	0	0	0.31	2	0.31	2
	Chilomycterus antillarum	0	0	0	0	0	0	0	0	0.16	1	0	0	0.16	1
Echeneididae	*Echeneis naucrates*	0	0	0	0	0	0	0	0	0.41	1	0	0	0.41	1
Grammistidae	*Rypticus saponaceus*	0.69	5	0.74	4	0	0	0	0	0.64	3	0.50	3	2.57	15
Haemulidae	*Haemulon carbonarium*	0	0	0.33	3	0.27	2	0.64	4	5.17	44	3.38	23	9.78	76
	Haemulon flavolineatum	1.95	18	2.20	19	3.27	37	5.33	61	7.22	69	2.48	21	22.45	225

Table 1. continued	*Haemulon chrysargyreum*	0.86	10	0.54	5	0.15	2	0.09	1	0.46	6	3.43	32	5.54	56
	Haemulon aurolineatum	0.22	2	0	0	0	0	0	0	0	0	0	0	0.22	2
	Haemulon plumieri	2.05	3	0	0	0	0	0	0	0	0	0	0	2.05	3
Holocentridae	*Myripristis jacobus*	1.58	19	3.02	32	1.91	20	1.77	20	6.15	58	2.73	30	17.16	179
	Holocentrus marianus	0.13	2	0.30	1	0	0	0	0	0	0	0	0	0.43	3
	Holocentrus rufus	9.57	93	10.44	89	3.82	36	2.16	21	6.87	52	4.60	35	37.46	328
	Holocentrus adscensionis	0.47	2	5.54	21	0	0	0	0	3.56	17	3.27	14	12.84	54
Kyphosidae	*Kyphosus sectatrix*	0	0	0	0	0	0	0	0	0.68	2	0.55	1	1.24	3
Labridae	*Clepticus parrai*	0	0	0	0	0	0	0	0	0.22	1	0.47	3	0.68	4
	Bodianus rufus	0	0	0	0	0.28	1	0	0	3.43	8	0.81	2	4.52	11
Lutjanidae	*Lutjanus buccanella*	0.36	3	1.12	8	0	0	0	0	0	0	0	0	1.48	11
	Lutjanus joco	0	0	1.1	2	0	0	0	0	0	0	0	0	1.10	2
	Lutjanus griseus	0	0	5.25	3	0	0	0	0	0	0	0	0	5.25	3
	Lutjanus mahogoni	2.09	6	1.64	7	1.14	4	0	0	1.58	11	1.38	7	7.83	36
	Ocyurus chrysurus	0.24	2	0.8	1	0	0	0	0	0	0	1.45	1	2.49	4
Malacanthidae	*Melancanthus plumieri*	0.62	2	1.09	4	0	0	0	0	0	0	0	0	1.71	6
Monacanthidae	*Cantherhines pullus*	0	0	0.57	6	0	0	0	0	0.54	6	0.42	5	1.53	17
	Aluterus scriptus	1.85	5	1.37	2	3.11	5	5.58	8	2	5	3.58	2	17.49	27
	Cantherhines macroceros	8.72	17	8.94	17	1.55	2	0.95	7	27.15	69	13.98	34	61.29	146
Mullidae	*Pseudupeneus maculatus*	0.73	4	0.24	4	0	0	0	0	1.42	10	2.08	14	4.47	32
	Mulloidichthys martinicus	2.82	19	2.04	13	0.88	5	0.11	1	11.10	65	11.99	72	28.95	175
Ostraciidae	*Lactophrys polygonia*	2.77	7	3.61	10	0	0	0	0	1.79	5	3.75	12	11.93	34
	Lactophrys quadricornis	0.75	1	0.59	1	0	0	0	0	0	0	0	0	1.34	2
	Lactophrys triqueter	0.30	1	0.14	2	0	0	0	0	0.58	8	1.24	7	2.25	18
	Lactophrys bicaudalis	0	0	0.81	4	0	0	0	0	0.64	1	0	0	1.44	5
	Lactophrys trigonus	0.15	3	0.13	2	0	0	0	0	0	0	0.29	2	0.58	7
Pomacanthidae	*Pomacanthus paru*	0.39	1	0	0	0	0	0	0	1.00	1	0.06	1	1.45	3

Table 1. continued	*Holacanthus ciliaris*	0	0	0	0	0	0	0	0	0.46	1	1.65	2	2.11	3
	Holacanthus tricolor	0.05	1	0.54	4	0	0	0	0	2.60	13	10.87	9	14.05	27
Pomacentridae	*Stegastes variabilis*	0	0	0	0	0	0	0	0	0	0	0.12	2	0.12	2
	Abudefduf saxatilis	0	0	0.12	1	0	0	0	0	0	0	0.10	1	0.22	2
	Microspathodon chrysurus	0	0	0	0	0	0	0	0	0	0	0.24	3	0.24	3
Priacanthidae	*Priacanthus cruentatus*	0.45	5	0.15	2	0	0	0	0	1.05	2	0.48	1	2.14	10
Scaridae	*Scarus taeniopterus*	23.26	154	36.59	232	0.53	4	0.59	5	4.30	19	10.13	46	75.40	460
	Sparisoma aurofrenatum	4.75	38	7.60	62	0.57	6	0.56	6	26.23	169	27.61	176	67.32	457
	Sparisoma rubripinne	0	0	0	0	0	0	0	0	0.15	1	0.32	1	0.48	2
	Sparisoma chrysopterum	0.16	2	0.14	1	1.08	6	0.42	3	4.68	12	2.01	4	8.47	28
	Scarus iserti	0.66	3	0.14	1	0	0	0.53	2	0	0	0	0	1.32	6
	Sparisoma viride	0	0	0.61	4	0.36	2	1.56	9	3.98	12	0.40	2	6.90	29
Sciaenidae	*Equetis punctatus*	0	0	0	0	0	0	0	0	0.36	1	0	0	0.36	1
Serranidae	*Cephalopholis fulvus*	3.66	14	1.12	5	1.31	5	0.78	2	10.77	35	9.12	24	26.76	85
	Paranthias furcifer	0	0	0.31	4	0	0	0	0	0	0	0	0	0.31	4
	Cephalopholis cruentata	3.07	16	5.38	3	8.97	39	4.20	21	0	0	0.49	2	22.10	81
	Epinephelus guttatus	0	0	0.76	3	0	0	0	0	1.25	1	3.53	4	5.54	8
	Epinephelus adscensionis	2.27	1	0	0	0	0	0	0	12.37	13	3.21	3	17.84	17
	Mycteroperca sp.	0.19	1	0	0	0	0	0	0	0	0	0	0	0.19	1
Sparidae	*Calamus bajonado*	0.09	1	0	0	0	0	0	0	0	0	0.47	1	0.56	2
	Calamus calamus	0	0	0.25	1	0	0	0	0	0	0	0	0	0.25	1
Sphyraenidae	*Sphyraena barracuda*	0	0	1.36	1	0	0	0	0	0	0	0	0	1.36	1
INVERTEBRATES															
Crustacea	*Carpilius corallinus*	-	2	-	1	-	2	0	0	-	1	-	2	-	8
	Panulirus argus	-	10	-	4	-	4	-	3	-	2	-	1	-	24
	conch	0	0	0	0	0	0	-	1	0	0	0	0	-	1
		0	0	0	0	0	0	0	0	-	1	0	0	-	1

Table 2.. Finfish catch composition of regulation and conventional traps at three sites in Barbados between January and May 1999, shown by total weight (kg) and number of fish, summarised by family.

	SIX MEN'S				WESTON				OISTINS				ALL SITES	
	Reg. traps		Con. traps		Reg. traps		Con. traps		Reg. traps		Con. traps		All traps	
FAMILY	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish	Wt (kg)	# Fish
Scaridae	28.82	197	45.08	300	2.54	18	3.65	25	39.34	213	40.46	229	**159.89**	**982**
Monacanthidae	10.57	22	10.88	25	4.66	7	6.53	15	29.70	80	17.98	41	**80.12**	**190**
Acanthuridae	2.68	35	2.01	25	3.65	35	2.51	28	22.2	214	44.19	343	**77.22**	**680**
Serranidae	8.19	32	7.57	15	10.28	44	4.98	23	24.39	49	16.34	33	**72.74**	**196**
Holocentridae	11.75	116	19.30	143	5.73	56	3.83	41	16.59	127	10.60	79	**67.89**	**562**
Haemulidae	5.08	33	3.07	27	3.69	41	6.06	66	12.86	119	9.29	76	**40.04**	**362**
Balistidae	15.36	106	18.31	102	0	0	0.73	1	2.41	3	3.04	4	**39.85**	**216**
Mullidae	3.56	23	2.28	17	0.88	5	0.11	1	12.52	75	14.07	86	**33.41**	**207**
Lutjanidae	2.68	11	9.81	21	1.14	4	0	0	1.58	11	2.83	8	**18.14**	**55**
Pomacanthidae	0.44	2	0.54	4	0	0	0	0	4.06	15	12.58	12	**17.62**	**33**
Ostraciidae	3.97	12	5.28	19	0	0	0	0	3.01	14	5.28	21	**17.53**	**66**
Carangidae	0.6	5	1.03	9	0	0	0	0	0.23	3	8.22	28	**10.07**	**45**
Labridae	0	0	0	0	0.28	1	0	0	3.64	9	1.28	5	**5.20**	**15**
Diodontidae	1.13	1	0	0	0	0	0	0	2.32	3	0.87	4	**4.32**	**8**
Chaetodontidae	0.55	11	0.66	15	0	0	0	0	1.11	23	1.43	29	**3.75**	**78**
Bothidae	0	0	0	0	0	0	0	0	0.41	1	3.07	2	**3.48**	**3**
Grammistidae	0.69	5	0.74	4	0	0	0	0	0.64	3	0.50	3	**2.57**	**15**
Priacanthidae	0.45	5	0.15	2	0	0	0	0	1.05	2	0.48	1	**2.14**	**10**
Malacanthidae	0.62	2	1.09	4	0	0	0	0	0	0	0	0	**12.90**	**6**
Sphyraenidae	0	0	1.36	1	0	0	0	0	0	0	0	0	**1.36**	**1**
Kyphosidae	0	0	0	0	0	0	0	0	0.68	2	0.55	1	**1.24**	**3**
Sparidae	0.09	1	0.25	1	0	0	0	0	0	0	0.47	1	**0.82**	**3**
Pomacentridae	0	0	0.12	1	0	0	0	0	0	0	0.46	6	**0.58**	**7**
Aulostomidae	0.48	2	0	0	0	0	0	0	0	0	0	0	**0.48**	**2**
Echeneididae	0	0	0	0	0	0	0	0	0.41	1	0	0	**0.41**	**1**
Sciaenidae	0	0	0	0	0	0	0	0	0.36	1	0	0	**0.36**	**1**
ALL FINFISH	98.70	621	129.61	735	32.82	211	28.51	200	179.50	968	193.98	1012	**663.13**	**3747**

Table 3. A summary of the key characteristics and finfish catch performance of regulation and conventional traps at three sites in Barbados between January and May 1999.

CHARACTERISTIC	SIX MEN'S		WESTON		OISTINS		ALL SITES	
	Regulation traps	Conventional traps	Regulation traps	Conventional traps	Regulation traps	Conventional traps	Regulation traps	Conventional traps
N (no. hauls examined)	33	27	30	30	31	31	94	88
Mean weight/trap/haul (kg)	3.01	4.82	1.08	1.05	5.75	5.62	3.31	4
Mean number/trap/haul	20	28	7	7	31	33	19	22
Mean size of fish (g)	159	176	156	143	184	192	173	181
Dominant family by weight	Scaridae	Scaridae	Serranidae	Haemulidae	Scaridae	Acanthuridae	Scaridae	Scaridae
Weight as % of total catch	23.6	29.2	31.3	21.3	21.9	22.8	22.7	28.7
Dominant family by number	Scaridae	Scaridae	Holocentridae	Haemulidae	Scaridae	Acanthuridae	Scaridae	Scaridae
Number as % of total catch	24.8	40.6	26.5	33	22	33.9	23.8	28.4
Dominant species by weight	princess parrotfish	princess parrotfish	graysby	scrawled filefish	whitespotted filefish	redband parrotfish	redband parrotfish	princess parrotfish
Weight as % of total catch	29.2	28.2	27.3	19.6	15.1	14.2	10.1	13.4
Dominant species by number	princess parrotfish	princess parrotfish	graysby	french grunt	redband parrotfish	ocean surgeon	redband parrotfish	princess parrotfish
Number as % of total catch	31.7	31.6	18.5	30.5	17.5	19.5	11.8	14.5
Normal soak time (days)	3-4	3-4	7	7	3-4	3-4	3-7	3-7
Trap size (m^3)	1.81	1.81	1.13	1.13	2.83	2.83	1.31-2.83	1.31-2.83
Depth range (m)	25, 40-50	25, 40-50	6-8	6-8	19-20, 26	19-20, 26	6-50	6-50

The Six Men's fisher did not always follow instructions to fish regulation and conventional traps side-by-side in pairs. Therefore two analyses were conducted, the first examined all traps regardless of whether or not they were strictly paired, and the second used only strict pairs. The mean weight of fish per trap haul differed significantly between regulation traps (3.01 kg/ trap haul) and conventional traps (4.82 kg/trap haul) when all traps were considered (Mann-Whitney test: U = 2.474, n = 33+27, P = 0.013), with slightly higher catches for conventional traps. However, the mean number of fish per trap haul (20 for regulation traps; 28 for conventional) was not significantly different between the two trap types (Mann-Whitney test: U = 1.502, n = 33+27, P = 0.133). Interestingly, when only strict pairs of traps were used, both weight and number of fish per trap haul varied significantly between traps (Wilcoxon paired-sample test; for weight: T = 2.676, n = 21, P = 0.0077; for numbers: T = 2.572, n = 21, P = 0.010) with both weight and number per trap haul being greater for conventional traps.

Weston — At the Weston site, the mean weight of fish per trap haul was not significantly different between regulation traps (1.08 kg/trap haul) and conventional traps (1.05 kg/trap haul) (Wilcoxon paired-sample test: T = 0.165, n = 30, P = 0.869). The mean number of fish per trap haul (7 for regulation traps; 7 for conventional traps) was also not significantly different between the two trap types (Wilcoxon paired-sample test: T = 0.114, n = 30, P = 0.909).

Oistins — At the Oistins site, the mean weight of fish per trap haul was also not significantly different between regulation traps (5.75 kg/trap haul) and conventional traps (5.82 kg/ trap haul) (Wilcoxon paired-sample test: T = 0.088, n = 31, P = 0.930). The mean number of fish per trap haul (31 for regulation traps; 33 for conventional) was also not significantly different between the two trap types (Wilcoxon paired-sample test: T = 0.108, n = 31, P = 0.914).

Overall sites — When the data from all the three sites (Six Men's, Weston, and Oistins) were pooled the mean weight of fish per trap haul was not significantly different between regulation traps (3.30 kg/ trap haul) and conventional traps (3.88 kg/trap haul) (Wilcoxon paired-sample test: T = 1.012, n = 82, P = 0.311). The mean number of fish per trap haul (20 for regulation traps; 22 for conventional) was also not significantly different between the two trap types (Wilcoxon paired-sample test: T = 1.379, n = 82, P = 0.168).

Comparison of Catch Composition between Regulation and Conventional Traps

Again as a result of differences in gear and habitat among sites, initial analyses were site specific.

Six Men's — A total of 1,356 finfish belonging to 56 species from 21 families were recorded at the Six Men's site (Table 1). Not surprisingly, the most abundant species in the catches of regulation traps was the princess parrotfish which represented 24 % of total weight and 25 % of total number (Table 1). The most abundant family was Scaridae which represented 29 % of total weight and 32 % of total number (Table 2). In conventional traps, the most abundant species was also princess parrotfish which represented 28% of total weight and 32% of total number, and the most abundant family was also Scaridae which represented 29 % of total weight and 41% of total number (Tables 1,2).

A statistical comparison of the catch composition using the top ten families (by weight) showed no significant difference between regulation and conventional traps (2 x 10 Chi-squared contingency test: $X^2 = 7.014$, $P = 0.636$).

Weston — A total of 411 finfish belonging to 21 species from 12 families were recorded at the Weston site (Table 1). The most abundant species (by weight) in the catches of regulation traps was the graysby (*Cephalopholis cruentata)* which accounted for 27% of the total catch weight. Three species (graysby; french grunt, *Haemulon flavolineatum;* and longspine squirrelfish, *Holocentrus rufus*) were the most abundant by number, accounting for 18%, 18% and 17 % of total numbers respectively (Table 1). The most abundant families in regulation traps were Serranidae (which accounted for 31% of total weight) and Holocentridae (which accounted for 27% of the total number). In conventional traps, the most abundant species by weight were scrawled filefish (*Aluterus scriptus*) and french grunt which represented 20% and 19% of total weight respectively (Table 1). French grunt was the most abundant species by number, accounting for 31% of the total catch of conventional traps. The most abundant family was Haemulidae which represented 21% of total weight and 33% of total number taken by conventional traps (Table 2).

Again the top ten families were used to compare the catch composition (by weight) of regulation and conventional traps. No significant difference was found between them (2 x 10 Chi-squared contingency test: $X^2 = 5.933$, $P = 0.747$).

Oistins — A total of 1,980 finfish belonging to 54 species from 24 families were recorded at the Oistins site. The most abundant species (by weight) in the catches of regulation traps were whitespotted filefish (*Cantherhines macroceros*) and redband parrotfish, each representing 15% of total weight (Table 1). Redband parrotfish was the most abundant species by number, accounting for 17% of the total catch. The most abundant family in regulation traps were Scaridae which represented 2 % of total weight, and Acanthuridae and Scaridae which each accounted for 22% of total number taken. In conventional traps, the most abundant species (by weight) was redband parrotfish which represented 14% of total weight, while ocean surgeon was the most abundant by number, accounting for 19% of the total. The most abundant family in conventional traps was Acanthuridae which represented 23% of total weight and 34% of total number.

Again the top ten families (excluding Carangidae (bar jacks) on the basis that they are essentially mobile pelagic species) were used to compare the catch composition (by weight) of regulation and conventional traps. As with Six Men's and Weston, there was no significant difference found between them (2 x 10 Chi-squared contingency test: $X^2 = 14.470$, $P = 0.107$).

Overall sites — Although there were no differences in the taxonomic composition of catches between regulation and conventional traps, there was an interesting and significant difference in the catch composition among sites (using pooled data from both trap types; 3 x 10 Chi-squared contingency test: $X^2 = 139.289$, $P < 0.001$) and between all pairs of sites ($P < 0.001$ in all cases) (Table 4).

DISCUSSION

Acceptance of fishery regulations by fishers (which is essential for effective enforcement) is much more likely if they are conceptually simple, and are proven to be effective in achieving the stated fishery management objective without causing unnecessary hardship to fishers (e.g. Adams 1996).

In this case, the primary objective of the new fish trap regulations (which require a minimum mesh size of 3.18 cm, an escape panel and an identification marker) is to reduce fishing mortality in the trap fishery by minimising the level of ghost fishing. Since the regulated minimum mesh size corresponds to the size currently used by the vast majority of trap fishers in Barbados (Selliah, in press) this specification is clearly not intended to have any impact on current levels of fishing mortality or current catch rates. It should however, sensitise fishers to the need for control on mesh size (and thereby prepare them for possible increases in minimum mesh size in the future). It should also prevent fishers from moving to smaller mesh sizes in the future if further decreases in reef fish abundance occur. This may be particularly important given the very recent large-scale reef fish kills affecting Barbados (Fisheries Division 1999).

The identification marker is also not intended to have any impact on fishing mortality or catch rates within the trap fishery.

Table 4. Comparison of finfish catch composition by weight (using pooled data from regulation and conventional traps) among the three sites in Barbados between January and May 1999, shown by the top 10 families at each site.

	SIX MEN'S		WESTON		OISTINS	
FAMILY	Wt (kg)	Rank	Wt (kg)	Rank	Wt (kg)	Rank
Scaridae	73.90	1	6.19	5	79.80	1
Balistidae	33.68	2	0.73	9	5.45	10
Holocentridae	31.04	3	9.66	4	27.18	5
Monacanthidae	21.45	4	11.19	2	47.67	3
Serranidae	16.76	5	15.25	1	40.73	4
Lutjanidae	12.59	6	1.14	7	4.42	-
Ostraciidae	9.25	7	0	-	8.42	8
Haemulidae	8.14	8	9.75	3	22.14	7
Mullidae	5.84	9	0.99	8	26.59	6
Acanthuridae	4.69	10	6.16	6	66.38	2
Pomacanthidae	1.65	-	0	-	7.64	9
Labridae	0	-	0.28	10	4.92	-

Ghost fishing is recognised by trap fishers and managers alike to be a significant problem in the trap fishery, with the rate of trap loss estimated to be as high as 25 % of all traps per year (Wilson 1983). The use of an escape panel in each trap to minimise ghost fishing by lost traps is a conceptually simple and "common sense" measure. However, it is important that the escape panel functions effectively (i.e. that it becomes operational shortly after the trap is lost, but not during the normal coarse of fishing), so that ghost fishing is significantly reduced. It is also important that no unintentional hardships are incurred by fishers through loss of catches resulting from early opening of escape doors. Furthermore, it is important that the visual impact of the escape panel and identification marker does not cause unintended reduction in ingress rates and therefore result in reduced catch rates. Visual image effects on ingress rates and thus catch rates have been reported by several authors (e.g. Luckhurst and Ward 1987, Bohnsack et al. 1989).

In this study, we have tested a number of biodegradable materials for suitability as escape panel fasteners, and investigated the effects of the presence of the required escape panel and identification marker on the catch rates of trap fishers.

Hemp twine was found to be a very effective fastener for the Modified Dowridge Design escape panel, and functioned equally well with or without the tyre inner tube spring.

Catch rates (by number of fish per trap haul) were not affected by the regulation specifications for traps at any of the sites (except at Six Men's when only strictly paired traps were analysed). There was also no difference in the catch rates (by weight of fish per trap haul) between regulation and conventional traps at Weston and at Oistins. However, there was a difference detected at Six Men's with regulation traps catching less on average than conventional traps. The lower catch rate at this site could not be attributed to early opening of the escape panel doors, which suggests that it could have been caused by lower ingress rates for regulation traps. This may be so, but we feel that it is more likely a reflection of the more casual attitude of the Six Men's fisher to hauling his traps. Unlike the other fishers, the Six Men's fisher would frequently abandon a fishing trip before all traps (especially conventional traps) had been hauled. This meant that even within strictly paired traps (regulation and conventional traps fished side-by-side) soak times varied, usually with greater soak times for the conventional traps. This could account for the larger catches (see Munro et al. 1971, Luckhurst and Ward 1987). Interestingly when paired data from all three sites were pooled there was no significant difference in the weight of fish per trap haul, and there was no difference in the size (weight) of individual fish taken by the two trap types.

The taxonomic composition of catches was also unaffected by the regulation specifications. Despite large between-site differences in the species of fish caught (which most likely reflects the different habitats being fished), there were no within-site differences in the species taken by regulation and conventional traps.

In conclusion, the newly regulated trap specifications do not appear to have unintentional negative impacts on the fishing power of the trap, and are therefore likely to be well accepted by trap fishers. This should assist in achieving the management objective of reducing ghost fishing in the trap fishery.

ACKNOWLEDGEMENTS

This study was funded by a MAREMP student research grant to NS, MacArthur Foundation/CCA and the Barbados Fisheries Division. We are extremely grateful for the cooperation and assistance of trap fishers W. Straker, O. Boyce and C. McClean; research assistants S. Carter, E. Henry, M. Pena, G. Selliah, T. Sinckler, A. Singh; and Fisheries Division personnel G. Franklin, A. Marshall, T. Moore H. Sargeant and C. Taylor.

LITERATURE CITED

Adams, T.J.H. 1996. Modern institutional framework for reef fisheries management. Pages 337 - 360 in: N.V.C. Polunin and C.R. Roberts (eds.) *Reef Fisheries*. Chapman and Hall, London.

Beliaff, B., M. Louis, and J. Souprayen. 1992. Preliminary results on the efficiency and selectivity of the chevron-shaped trap, a traditional fishing device in Guadeloupe. *Proc. Gulf Carib. Fish. Inst.* **41**:150-165.

Bohnsack, J.A., D.L. Sutherland, D.E. Harper, D.B. McClellan, M.W. Hulsbeck, and C.M. Holt. 1989. The effects of fish trap mesh size on reef fish catch off southern Florida. *Mar. Fish. Rev.* **51**(2):36-46.

Fisheries Division 1997. Barbados fisheries management plan: schemes for the management and development of fisheries in the waters of Barbados. Fisheries Division, Ministry of Agriculture and Rural Development, Barbados. 1/1 – 7/5 p.

Fisheries Division 1999. Report of the regional technical workshop on fish kill events. Preliminary draft report to CARICOM secretariat, Guyana.

Hunte, W. and Mahon, R. 1994. Size selection of fish by the mesh of fish traps. CARICOM Fishery Management Unit, Belize City, Belize. 28 pp.

Kumpf, H.E. 1980. Practical considerations and testing of escape panel material in fish traps. *Proc. Gulf Carib. Fish. Inst.* **32**:211-214.

Luckhurst, B. and J. Ward. 1987. Behavioral dynamics of coral reef fishes in Antillean fish traps at Bermuda. *Proc. Gulf Carib. Fish. Inst.* **38**:528-546.

Mahon, R. and N. Drayton. 1990. Trap fishery management in Barbados: the fisherman's perspective. *Carib. Mar. Stud.* **1**:94-100

Munro, J.L. 1983. *Caribbean Coral Reef Fisheries Resources.* ICLARM studies and reviews 7, International Center for Living Aquatic Resources Management, Manila, Phillippines.

Munro, J.L., P.H. Reeson, and V.C.Gaut. 1971. Dynamic factors affecting the performance of the Antillean fish trap. *Proc. Gulf Carib. Fish. Inst.* **23**:184-194.

Nicholson, W. and L. Hartsuijker. 1983. The state of the fisheries resources of the Pedro Bank and south Jamaica shelf. *FAO Fish Rept.* **228** (Suppl): 215 - 254.

Robichaud, R., W. Hunte, and H. Oxenford. 1999. Effects of increased mesh size on catch and fishing power of coral reef fish traps. *Fisheries Research* **39**:275-294.

Olsen, D.A., A.E. Dammann, and J.A. LaPlace. 1978. Mesh selectivity of West Indian fish traps. *Mar. Fish. Rev.* **40**(7):15-16.

Rosario, A. and Y. Sadovy. 1991. Trap mesh selectivity off the west coast of Puerto Rico (January 1990 - December 1990). Fisheries Research Laboratory, Puerto Rico Department of Natural Resources. 56 pp.

Sary, Z., H.A. Oxenford, and J.D. Woodley. 1997. Effects of an increase in trap mesh size on an overexploited coral reef fishery at Discovery Bay, Jamaica. *Mar. Ecol. Prog. Ser.* **154**:107-120.

Selliah, N.M. In press. The Barbados trap fishery: an investigation of the current status and effects of an escape panel regulation on catch rates. M.Sc. thesis, University of The West Indies, Barbados.

Stevenson, D.K. and P. Stuart-Sharkey. 1980. Performance of wire fish traps on the western coast of Puerto Rico. *Proc. Gulf Carib. Fish. Inst.* **32**:173-193.

Wilson, S.B. 1983. A report on the trap fishing industry of Barbados. Division of Fisheries. Ministry of Agriculture, Food and Consumer Affairs, Barbados. 32 plus xxxiii pp.

Wolfe, R.S. and G.R. Chislett. 1974. Trap fishing explorations for snapper and related species in the Caribbean and adjacent waters. *Mar. Fish. Rev.* **36**(9):49-61.

Integrating Fish Fauna and Habitat Assessments: A Fundamental Step in Developing Fishery Reserve Design Criteria

CONRAD W. RECKSIEK[1], BRIAN R. MURPHY[1],
RICHARD S. APPELDOORN[2] and KENYON C. LINDEMAN[3]
[1]Department of Fisheries, Animal and Veterinary Science
University of Rhode Island
Kingston, RI 02881 USA
[2]Department of Marine Sciences
University of Puerto Rico
Mayaguez, Puerto Rico 00681-9013 USA
[3]Division of Marine Biology and Fisheries
RSMAS, University of Miami
Miami, FL 33149 USA

ABSTRACT

We report on a preliminary sampling and analysis protocol for the insular shelf of southwest Puerto Rico that identifies coastal habitats of particular significance to early life stages of representative species. This can facilitate fishery reserve designs that include habitats essential for long-term fishery production and ecosystem conservation. The spatial framework employed uses a cross-shelf habitat matrix with a vertical axis that includes a hierarchy of structural habitat types and a horizontal axis that includes an estuarine to shelf-edge gradient of geomorphological zones. This framework aids the quantification of areal habitat cover and size distributions of species within diverse structural habitat categories across multiple cross-shelf gradients. Structural habitats or geomorphic zones can be collapsed or expanded according to data availability. Such attributes aided the design of a sampling program used to map substrates of key cross-shelf habitats in the La Parguera area of Puerto Rico. Representative fish faunas, i.e., those of trophic and fishery significance, were quantified by visual assessment after mapping transect substrates. This allowed the estimation of relative areas of differing habitat types and size-specific fish densities from transect-based visual assessments. Superimposing density data for differing life stages using available GIS datalayers of cross-shelf habitat distributions for the La Parguera area may yield a practical set of habitat selection criteria for the design of viable marine fishery reserves.

KEY WORDS: Haemulidae, mangrove habitats, marine reserves

INTRODUCTION

Establishing quantitative associations between fish populations and their habitats is of critical interest in determining the proposed boundaries of no-take fishery reserve areas. In establishing marine fishery reserves (MFR), managers should be able to justify their area selections in terms of which species, life history stages, and habitats are being protected (Roberts et al. 1995). If the structural and functional values of habitats are well understood, habitats can be ranked according to which habitat attributes and life stages are most essential to preserving the integrity of the system. However, the availability of both:

i) detailed and tractable habitat classifications, and
ii) fish abundance data collected in concert with existing habitat classification systems, is often limited.

Multiple abiotic and biotic variables can both describe and influence the distributions of organisms across spatially heterogeneous shelf systems. Various approaches for characterizing diverse arrays of habitats exist and many have focused on either structural attributes of bottom-types or water column characteristics. Approaches which combine these attributes in geographically logical manners may have considerable utility. For example, cross-shelf spatial frameworks that stratify complex spatial systems using fine-scale structural habitats and larger-scale geomorphic zones can foster detailed examinations of multiple habitat types across continental and insular shelf systems (Lindeman et al. 1998, Recksiek and Appeldoorn 1998, Appeldoorn et al. in press).

We report on preliminary results from the use of cross-shelf habitat frameworks in southwest Puerto Rico to map distributions of multiple life stages across diverse habitats and geomorphic zones. We focus on preliminary examples of ontogenetic variations in cross-shelf habitat use among early life stages of common coastal marine fishes.

METHODS

Cross-shelf spatial frameworks employ matrices with vertical axes representing structural habitat types and horizontal axes representing geomorphic strata across the shelf area under study. The habitat types include structures on a scale of 1 m^2, while the geomorphic strata are physiographic proxies for wind exposure, distance from shore, and depth across the shelf. By superimposing the structural habitats on the vertical axis and geomorphic strata on the horizontal axis, a cross-shelf habitat matrix of primary habitat combinations across the shelf is produced (Lindeman et al. 1998). Intersections of individual habitat types and geomorphic strata form cells in the matrix termed cross-shelf habitats. For example, one cross-shelf habitat is mangrove roots within the mainland inshore geomorphic stratum. A variety of species and life stage abundance measures can then be stratified within the matrix to examine ontogenetic patterns of habitat use within and among species, as well as combinations of habitat descriptors at differing spatial scales.

A preliminary cross-shelf habitat (CSH) matrix for the La Parguera area of the southwest shelf of Puerto Rico was assembled at a workshop in the spring of 1998 involving approximately ten fish biologists familiar with the area. This was the basis for the revised spatial matrix for La Parguera used in the present study (Figure 1). The matrix contains 20 structural habitat types and 36 geomorphic zones within three broad categories of the insular shelf (inner through outer shelf). We here focus on a subset of cross-shelf habitats within the La Parguera CSH framework based on a larger investigation in progress (Appeldoorn et al. in press). We emphasized transects in three geomorphic strata: Windward Inshore (WI), Leeward Shallow (LS), and Windward Shallow (WS). Most strata examined occurred within a depth range of 0 -3 m. Preliminary analyses focused on two structural habitat types: Mangrove Roots (MR) and Dead Coral-High Relief (DH) (Figure 1).

Transects were oriented along isobaths or parallel to local features such as mangrove cays or shallow reefs. The transects measured 24 m by 4 m ($96m^2$). A measuring tape was used as the transect center line. Structural habitat types were mapped *in situ* by a diver. A 1 m^2 square of plastic pipe was used by the diver to map habitats along the tape (illustrated for two transects within the windward inshore geomorphic zone in Figure 2). When the structural habitat within a meter square was largely homogeneous, the entire square was considered one habitat type; when heterogeneous, the square was subdivided into multiple habitat types (Figure 2).

The transect's fish fauna was assessed after mapping by a diver proceeding slowly above or alongside the transect area. Generally two passes through the transect were required, the first to assess roaming species, the second to assess site-attached fishes, e.g., damselfishes. We recorded 40 representative species. By representative, we mean species that are of likely trophic or fishery significance. This preliminary report emphasizes two common species of grunts (Haemulidae), ecologically and economically important components of the La Parguera ichthyofauna. For example, one of these species, the French grunt (*Haemulon flavolineatum*), was the most abundant species censused in La Parguera mangrove habitats by Rooker and Dennis (1991).

To estimate density of 1 cm total length classes over each structural habitat type, fishes were assessed according to three modes: points, polygons, or lines. When one or more fishes were sighted at a single location, a coded point corresponding to the individual fish or group of fish was recorded on a data sheet with an overlaid transect grid. In the second case, a figure was drawn on the data sheet's grid drawing to represent the area occupied by a school of a site-attached species, regardless of whether or not the polygon crossed habitat types. In the third case, designed for roaming species, a line was sketched on the drawing.

Fishes noted at each of those modes were recorded on the data sheet using a simple coding system. At each point, polygon, or line, the numbers of individuals together with the apparent range of total lengths (cm) were recorded. Lengths were judged after training using fish models (Rooker and Recksiek 1992); in situ lengths were estimated after that experience using memorized hand/finger configurations (Beets pers. comm.). Species were recorded to the lowest taxon possible; most fishes were identified to species. When a group contained more than one species, as is often the case for juvenile grunt schools, the numbers of constituent taxa were estimated (densities of each were estimated as described below).

Sketches on the grid drawings of our field data sheets were scanned to create maps of transect areas using ArcView and ArcInfo GIS software. This software enabled calculation of the areal coverage of each habitat type within each transect. For fish assessments, we also entered the points, polygons, or lines, together with component taxa, to the database so we could overlay them onto habitat maps of each transect (Figure 3). When the polygons crossed habitat types, respective areas were determined using GIS procedures; the areas were needed to calculate densities. In practice, line counts have occurred rarely; to date, we have considered the location of first sightings as a "point."

The fish data were entered into a spreadsheet to enable density calculations by species, total length interval, and cross-shelf habitat cell. Species density in a given habitat type was calculated on the basis of estimated numbers of fishes in the transect occurring over respective habitat areas (obtained from the GIS). When more than one individual was noted, the calculation of species abundance across size classes required three inputs: number of individuals, minimum total length, and maximum total length. By dividing the total number of individuals by the number of 1-cm size classes, a uniform frequency distribution was generated. For example, if 12 individuals were recorded from 6 - 10 cm TL, the mean number of individuals per size class is 2.4. This transformation was automatically performed by the spreadsheet upon entry of fish counts; one record therefore included computed length frequencies from the siting. When the record was derived from a polygon that crossed habitat category boundaries, additional records were created such that numbers of fishes were allocated in proportion to the subdivided areas of the polygon. The total surface area of the habitat types was included and used to calculate density of the species' total length classes for each habitat type in the transect. This generates a length frequency distribution of densities for each habitat type for each transect.

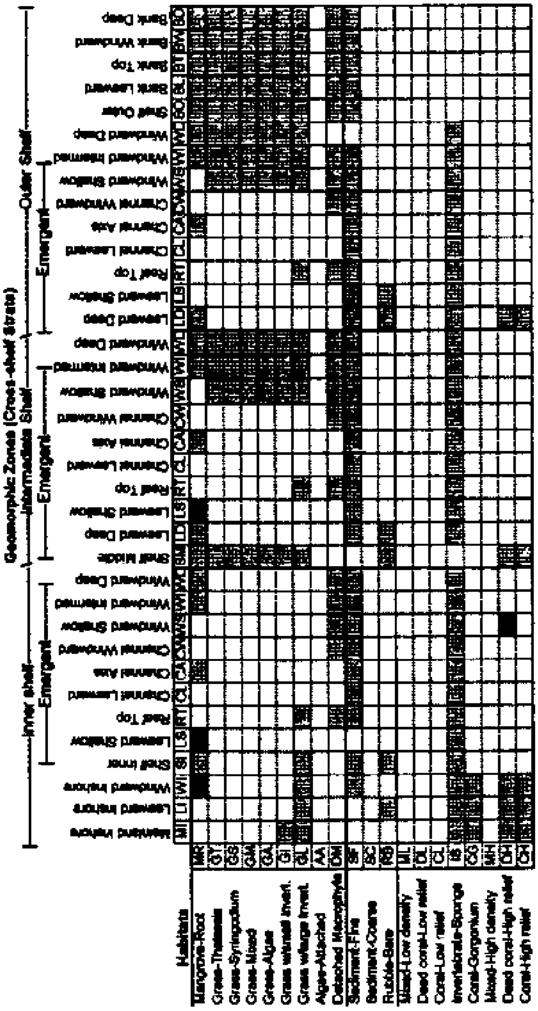


Figure 1. Preliminary cross-shelf habitat matrix for La Parguera, southwest Puerto Rico. Mnemonic acronyms denote habitat types (rows) and geomorphic zones (columns) as modified from Lindeman et al. (1998). Crosshatched cells indicate structural habitat types (Mangrove Root, Dead Coral-High Relief) and geomorphic zones (Windward Inshore–Inner Shelf, Leeward Shallow–Inner Shelf, Windward Shallow–Inner Shelf, and Leeward Shallow Intermediate Shelf)emphasized in the present study. Shaded cells indicate combinations of habitats and geomorphic zones that are absent from the system.

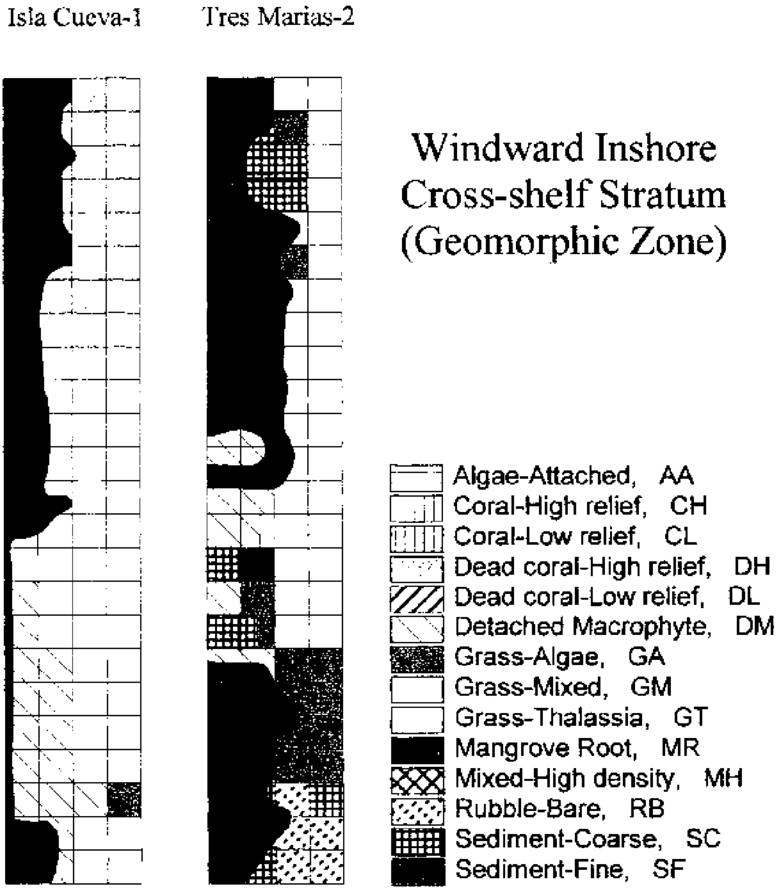
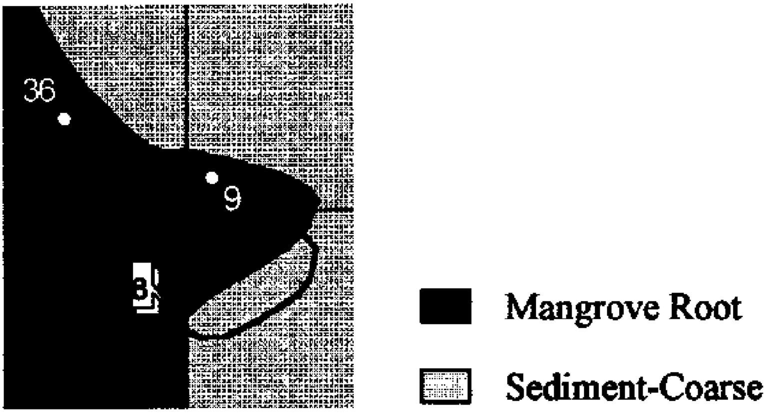


Figure 2. Habitat maps for two transects within the windward inshore cross-shelf stratum, La Parguera. Grid cells are 1m². Legend key includes habitat types shown in this figure and in Figure 4.

Leeward Shallow Cross-shelf Stratum

Las Pelotas - 6



Location Code	Species	N	Total Length (cm)
8	Hfla	5	8 - 12
8	Hfla	2	8 - 12
9	Sbar	1	25
36	Asax	1	3
36	Asax	1	1

Figure 3. Examples of how fish positions on mapped transects were delimited using either discrete points or polygon areas. The figure illustrates the interface between a red mangrove drop root canopy and sediment halo within a leeward shallow geomorphic zone of the inner shelf at Las Pelotas, La Parguera, July 1998.

RESULTS

To date, we have conducted 67 transects at the following sites in the La Parguera area: Isla Cueva - 10, N of Tres Marias – 10, Collado - 10, Las Pelotas – 9, Enrique – 6, La Gata – 6, Laurel – 4, San Cristobal – 4, Margarita – 4, W of El Palo – 4 (several transect examples are shown in Figure 4). These transects occurred within a total of seven geomorphic zones and included 18 structural habitat types. These data are under analysis and the following summaries employ only a small subset.

Size classes of the grunt species examined here were not uniformly distributed among either structural habitats or geomorphic zones. Ontogenetic differences in the density of French grunts among differing habitats were suggested by the preliminary results. For example, a bimodal size class distribution was observed when densities were compared between mangrove roots of windward inshore transects and dead coral of high relief within a windward shallow geomorphic zone (Figure 5). The latter cross-shelf habitat, used by older life stages, was positioned further seaward on the shelf. A bimodal pattern was seen among size classes of the bluestriped grunt but densities for the smaller fish within the windward inshore transects were clearly lower, in contrast to the French grunt case (Figure 5).

A preliminary examination of differential habitat use of the same structural habitat (mangrove roots) within differing geomorphic zones across the shelf is given in Figure 6. In the French grunt, these data suggest early life stages occurred closer to the mainland than older stages. The rarity of censused life stages under 9 cm TL made comparisons less distinct in bluestriped grunts. In all habitats and cross-shelf strata examined here, French grunts were 2-10 times more abundant than bluestriped grunts (Figures 5 and 6).

DISCUSSION

Recent initiatives to identify essential fish habitats and to design fishery reserve boundaries are complimentary because optimal reserve design logically implies that boundaries include habitats essential to stock production and sustainability. Therefore, habitats used during juvenile migrations (Appeldoorn et al. 1997) and at settlement and spawning (Lindeman et al. in press) should be protected from fishing gear impacts and coastal construction activities. Since the spawning areas of many species are discrete from settlement areas, spatial frameworks that can stratify comparisons of fish abundance among diverse structural habitat types and geomorphic zones may have utility in the cross-shelf positioning of marine reserve boundaries.

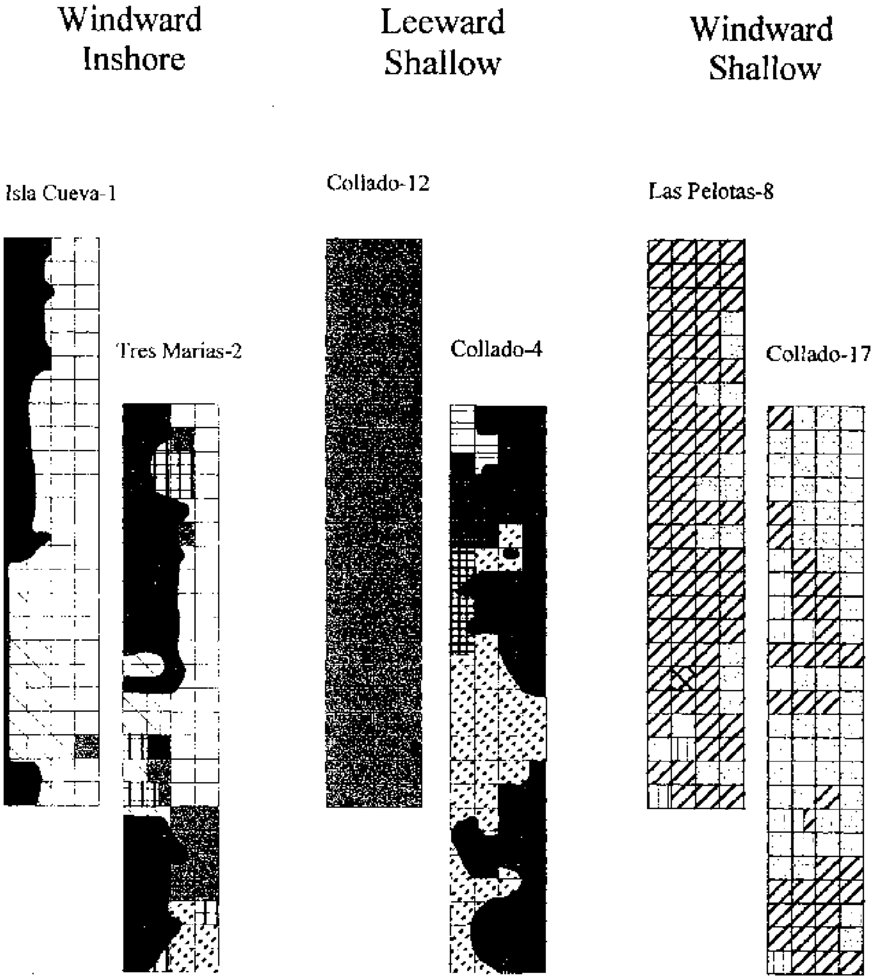


Figure 4. Habitat maps for six transects within three cross-shelf strata of the inner shelf of La Parguera. The key for habitat types is given in Figure 2.

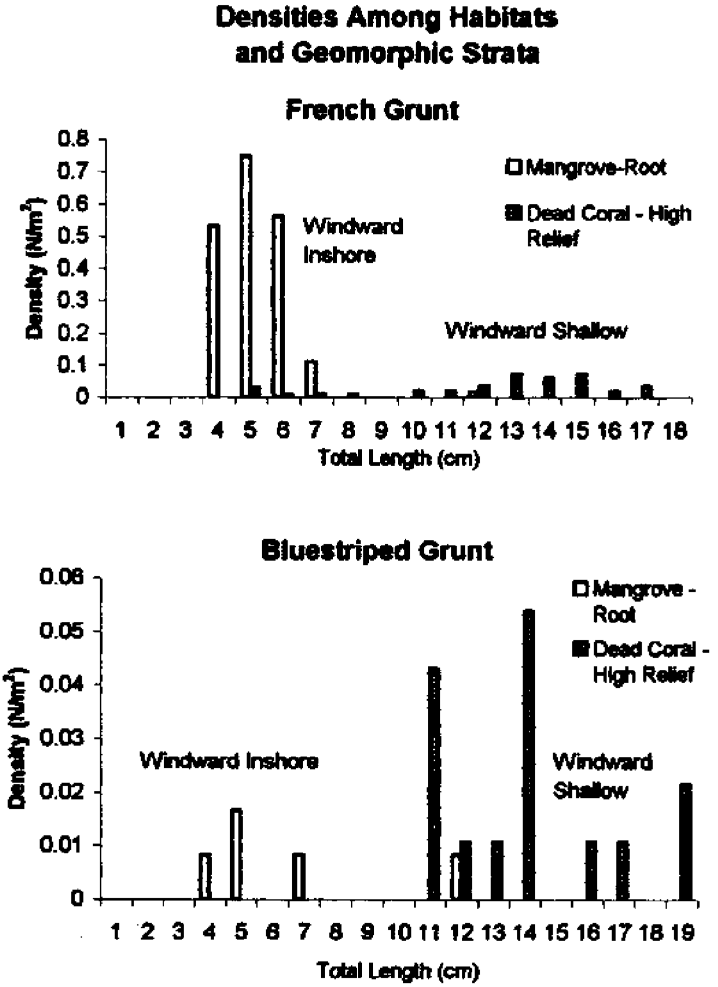


Figure 5. Densities of French grunt and bluestriped grunt size classes in mangrove root and dead coral-high relief habitats from windward inshore and windward shallow geomorphic zones of the inner shelf, respectively. Data for each of the two cross-shelf habitat combinations are pooled from two transects conducted in November-December, 1998.

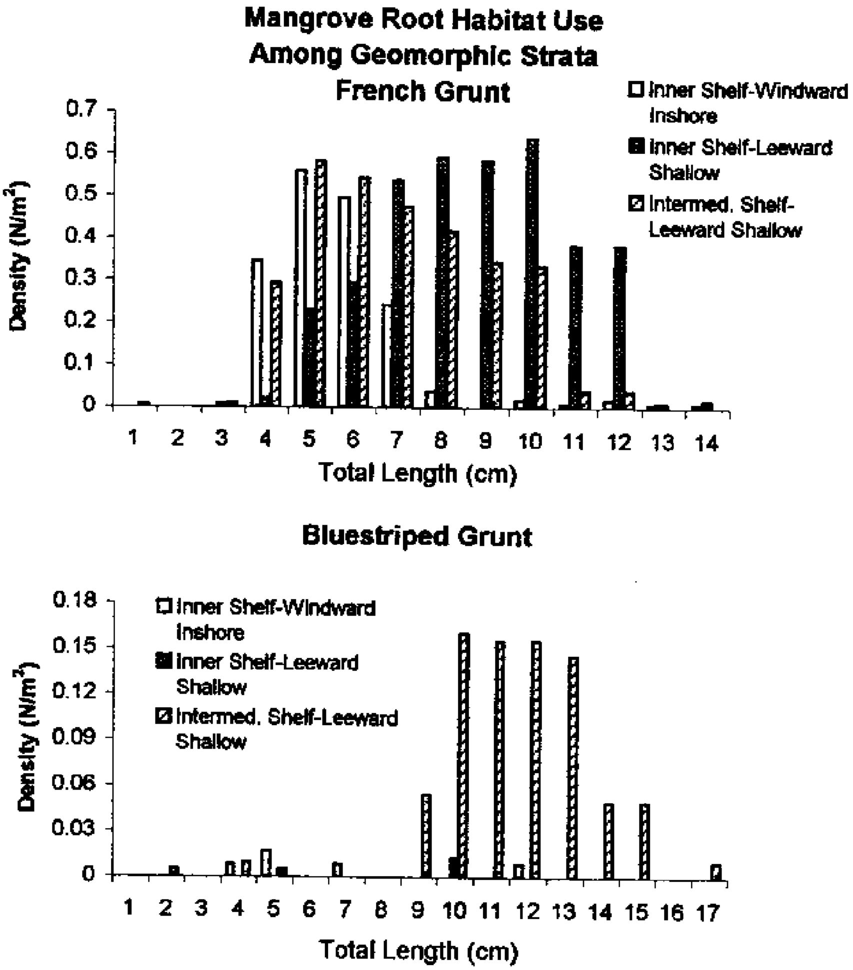


Figure 6. Densities of French and bluestriped grunt size classes from mangrove root habitats among three geomorphic strata. Data were collected in November-December 1998, and were pooled from three, four, and four transects per stratum, respectively.

Our approach builds upon preliminary applications of cross-shelf habitat frameworks by incorporating density estimates by size-class. Based on data availability, these density estimates may be collapsed or expanded among habitat types and geomorphic zones to assess variations in distribution among key spatial scales or to derive biomass estimates for mapped areas (Appeldoorn et al., in press). Preliminary analyses focused on mangrove root and dead coral habitats and showed potential ontogenetic habitat shifts among structures and depth zones (Figure 5). French grunts were more abundant in visual surveys of mangrove habitats than bluestriped grunts (Figure 6), a pattern also seen in the La Parguera area by Rooker and Dennis (1991). This contrasts with visual surveys and trapping in northeast Florida Bay, where bluestriped grunts dominated haemulid abundances, while French grunts were absent (Ley et al. 1999). Abundance estimates of haemulid early life stages within other combinations of habitat and geomorphic zones defined by cross-shelf habitat matrices are available (e.g., Lindeman and Snyder 1999). Comparisons of cross-shelf habitat use patterns that can be stratified according to data availability are underway among several geographic regions.

As fishery reserves become increasingly accepted management tools, research efforts are shifting in part from rationale for justification to guidelines for actual design (Recksiek and Appeldoorn, 1998). As reserves are ideally configured in interconnected networks (Roberts et al. 1995), broad design criteria can benefit from standardized spatial frameworks. In addition, a variety of new approaches that take advantage of advanced modeling and visualization tools have applications to both essential fish habitat identification (Rubec et al. 1999, Ault et al. 1999) and marine reserve design (Meester 2000). The hybridization of geographically standardized spatial frameworks and new analytic tools will yield practical methods for designing fishery reserves that protect both key habitats and essential demersal life stages.

ACKNOWLEDGMENTS

This project was funded by the University of Puerto Rico Sea Grant Program (R-31-1-98). We thank the faculty and students who worked together for several days during the summer of 1998 at the Isla Magueyes field station to identify the structural habitat categories and geomorphic zones used to assemble a cross-shelf habitat framework for the La Parguera area. We also thank Jerry Ault, Graciela Garcia-Moliner, Matt Kendall, Joe Kimmel, and Mark Monaco for their input on these issues.

LITERATURE CITED

Appeldoorn R.S., C.W. Recksiek, R.L. Hill, F.E. Pagan, G.D. Dennis. 1997. Marine protected areas and reef fish movements: the role of habitat in controlling ontogenetic migration. *Proc. 8th Int. Coral Reef Symp.* **2**: 1917-1922.

Appeldoorn, R.S., K. Foley, E. Molina, and C.W. Recksiek. Benthic mapping from fish and habitat transect data using GIS technology. *Proc. Gulf Carib. Fish. Inst.* Vol. **52**. In Press.

Ault, J.S., J. Luo, S.G. Smith, J.E. Serafy, J. D. Wang, R. Humston, and G. Diaz. 1999. A spatial dynamic multistock production model. *Can. J. Fish. Aquat. Sci.* **56**:1-22.

Ley, J.A., C.C. McIvor, and C.L. Montague. 1999. Fishes in mangrove prop-root habitats of northeastern Florida Bay: distinct assemblages across an estuarine gradient. *Estuar. Coast. Shelf Sci.* **48**:701-723.

Lindeman, K.C., G.A. Diaz, J.E. Serafy and J.S. Ault. 1998. A spatial framework for assessing cross-shelf habitat use among newly settled grunts and snappers. *Proc. Gulf Caribb. Fish. Inst.* **50**:385-416.

Lindeman, K.C. and D.B. Snyder. 1999. Nearshore hardbottom fishes of southeast Florida and effects of habitat burial caused by dredging. *Fish. Bull.* **97**(3):508-525.

Lindeman, K.C., R. Pugliese, G.T. Waugh, and J.S. Ault. Developmental patterns within a multispecies reef fishery: management applications for essential fish habitats and protected areas. *Bull. Mar. Sci.* In Press.

Meester, G.A. 2000. *A mathematical programming and simulation-based approach to determining critical factors in the design of optimal marine reserve plans for reef fish.* Ph.D. Dissertation. Univ. of Miami, Coral Gables, FL.

Recksiek, C.W., and R.S. Appeldoorn. 1998. In pursuit of design criteria for marine fishery reserves. *Proc. Gulf Carib. Fish. Inst.* **50**:372-384.

Roberts, C., W.J. Ballantine, C.D. Buxton, P. Dayton, L.B. Crowder, W. Milon, M.K. Orbach, D. Pauly, and J. Trexler. 1995. Review of the use of marine fishery reserves in the U.S. Southeastern Atlantic. *NOAA Tech. Mem.* NMFS-SEFSC-376, 31 pp.

Rooker, J., and C.W. Recksiek. 1992. The effects of training with fish models in estimating lengths of fish underwater. *Proc. Gulf Carib. Fish. Inst.* **41**:321-331.

Rooker, J.R. and G.D. Dennis. 1991. Diel, lunar and seasonal changes in a mangrove fish assemblage off southwestern Puerto Rico. *Bull. Mar. Sci.* **49**(3):684-698.

Rubec, P.J., J.C. W. Bexley, H. Norris, M.S. Coyne, M.E. Monaco, S.G. Smith, and J.S. Ault. 1999. Suitability modeling to delineate habitats essential for sustainable fisheries. *Amer. Fish. Soc. Symp.* **22**:108-133

"Do No Harm" Versus "Stop the Bleeding" in the Establishment of Marine Reserves for Fisheries Management

RICHARD S. APPELDOORN
Department of Marine Sciences
University of Puerto Rico
Mayagüez, Puerto Rico 00681-9013

ABSTRACT

It has been argued that, in analogy to medical science, the first rule of fishery management is "Do no harm". That is, when treating human patients, doctors should not act to worsen the patient's condition. Application of this analogy to fisheries management suggests that management should not take actions that may result in permanent resource and economic decline. Based on this analogy, it was recently argued that the establishment of marine reserves could be deleterious if placed in "sink" areas, as opposed to "source" areas. The implication is that management must await further scientific studies. However, this argument is flawed on several accounts. First, management cannot wait indefinitely for scientific proof. Management must consider the weight of scientific opinion and be flexible in response to increasing knowledge. A more appropriate medical analogy here is "Stop the bleeding", that is, stabilize the situation. Many fishery resources are in severe decline and actions are needed in the short term. Second, it is unlikely that scientific studies will ever be able to identify source and sink areas on the scale needed for multispecies fishery management. Third, there is now substantial empirical evidence that marine reserves create source areas by increasing spawning stock. Fourth, and perhaps most fundamental, the argument fails to understand the nature and importance of marine reserve networks. Marine reserves serve multiple management objectives in addition to enhancing spawning stock (e.g., provide control areas and buffers against management failure, maintain system integrity, diversify benefits). The principles of reserve replication and network design specifically serve to counter unknowns in siting reserves relative to source and sink areas, while further accounting for the obvious fact that species will differ in this regard. Following basic guidelines, marine reserves can be established now, for the betterment of resource managers and users.

KEY WORDS: Fisheries management, marine reserves, reserve location

INTRODUCTION

The use of marine reserves in fisheries management represents a partial paradigm shift. The shift is partial because, at present, marine reserves are seen as a necessary component of management, but one that also requires the more traditional quantitative stock assessment methodologies and resulting management options, especially quotas. This has been termed the "numerical approach" (Acheson and Wilson 1996). A full paradigm shift would move fisheries science and management further away from the numerical approach to what is termed the "parametric approach", where emphasis is placed on maintaining the parameters of system production. The goal is not to regulate or control the yield of fish, but to keep "the system in a state where the normal range of variability is preserved" (Acheson and Wilson 1996). Emphasis on marine reserves, as well as essential fish habitat, are clearly movement in the direction of parametric management, but many scientists and managers have not been able to fully embrace the new concept. Kuhn (1962) argued that new paradigms triumph not because they convince existing practitioners to change, but by the eventual replacement of existing practitioners with a new cadre of different minded professionals. Given current rates of fishing pressure and coastal development, any delay in adopting management measures designed to promote system productivity will be costly, both to local economies and the resources they support.

It has been argued that, in analogy to medical science, the first rule of fishery management is "Do no harm". That is, when treating human patients, doctors should not act to worsen the patient's condition. In particular, doctors should not conduct experiments of dubious outcome on their patients when known procedures may provide some degree of relief. Application of this analogy to fisheries management suggests that management should not take actions that may result in permanent resource and economic decline when other proven management options are available. Based on this analogy, it was recently argued that the establishment of marine reserves could be deleterious if placed in "sink" areas, as opposed to "source" areas. Sink areas are those that receive fish (through larval dispersal or adult migration) but do not, in turn, provide fish to other areas. Source areas are those that provide fish to other areas. Source areas may be self-seeding or may depend on recruitment from another source area. The concern is that, if reserves are placed preferentially in sink areas, fishing will be more concentrated in source areas, thus worsening the overall productive capacity of the system. The implication of the "Do no harm" dictum is that management must await further scientific studies. Specifically, do not establish marine reserves until we know all the answers. However, this argument is flawed on several accounts.

MANAGEMENT AND SCIENCE

There are fundamental differences between science and management. While it is generally agreed that mangement must have a sound scientific basis, this has not always been the case. In fact, for much of history, management was based more on religious practice than on science, with many cases of successful resource conservation (Johannes 1981, Acheson and Wilson 1996). In the end, management differs from science in two respects. First, management is a political process, science is not. Second, management must deal with current problems, whereas there are areas where science cannot make effective contributions, either because of excessive costs or times scales, or because effective methodologies are not currently available. Management cannot wait indefinitely for scientific proof. Management must consider the weight of scientific opinion and be flexible in response to increasing knowledge.

At present, the scientific model for fisheries management, that is, the one that most scientists and managers would argue they are using, is based on a decision making process called "rational judgement". Within this process an "optimal" judgement is reached statistically weighing and comparing all relevant information. However, use of rational judgment can become ponderous when the optimal solution is too difficult to calculate due to complexity or when no single best solution exits. In fact, most management decisions, in fisheries and elsewhere, are not made in this manner. Few managers have the scientific training or the time to carefully amass all scientific data and analyze them within the context of optimality models and decision theory, no less explain the outcome to their nonscientific constituency. Instead, most decisions are made within a process called "bounded rationality". Within this system, decisions are made within the context of limited, but well chosen bits of information and a set of simple rules for comparing or classifying them. These are variously called "heuristics", "rules of thumb", or in the case of marine reserves, "design criteria" (Recksiek and Appeldoorn 1998). Science can facilitate management by providing and verifying rules of thumb that can be used for decision making in complex situations, in the present, with limited information. For marine reserves in coral reef areas, a preliminary set of design criteria was devised by Appeldoorn and Recksiek (1998) (Table 1).

MANAGEMENT AND REALITY

Is "Do no harm" the right metaphor? Many fishery resources are in severe decline due to overfishing and due to habitat degradation associated with this and coastal development. The future of this course unchecked is easy to see: collapse. This has been adequately demonstrated both globally and regionally, and the economic and social dislocations are severe. Therefore, the option of no action is definitely one of doing harm in the long run, to both the industry and the resource. Actions are needed in the short term. A more appropriate medical metaphor here is "Stop the bleeding", that is, stabilize the situation. The more appropriate medical analogy recognizes that not only is the patient (fishery) sick, treatment is needed to prevent further decline and past remedies have rarely been effective, although they may have slowed its progress. Novel and/or drastic measures are needed. Marine reserves are novel, they need not be drastic, and they have been shown to be effective in practice.

Table 1. Biological guidelines for designing marine reserves (from Appeldoorn and Recksiek 1998).

1) All marine habitats in each biological region should be represented.
2) All habitats must be replicated.
3) Areas should be linked in a self sustaining network.
4) Network should eventually contain 20-30% of the management area.
5) Priority should be given to protecting spawning stock (min. pop. size = 500)
6) Emphasis should be given to large predators.
7) Protect locations of large spawning aggregations
8) Include areas of high habitat diversity.
   a) where possible, include habitats to be connected into a single closed area.
   b) size permitting, close off entire geomorphological units, such as reefs.
   c) If not, maximize geomorphic habitat diversity (e.g., fore & back reef plus reef channel).
9) Errors in siting specific reserves are buffered by replication of reserves.
10) Include areas known to serve as nursery areas.
11) Include upstream areas (receiving larval import).
12) Include downstream areas (greatest probability of retaining larvae).
13) Locate "No-take" zones within existing MPA's

Would there be any advantage to waiting for more scientific information on the siting of marine reserves relative to sources and sinks? Management must routinely contend with tens to over a hundred commercially important species. In coral reef areas there is an added degree of complexity due to the spatial heterogeneity of the environment on a variety of scales. This effectively splits a stock into a number of subpopulations, each with its own local dynamics. While there are some constants that one can apply across species to determine potential source areas, e.g. direction and strength of dispersing currents, these are greatly modified by the individual behaviors of each species, e.g. season of spawning, larval behavior, larval duration, preferred settlement habitat, etc. The expectation is that the many species in any given area arose from a wide variety of source areas, and that any specific area may serve as a source for some species and a sink for others. In addition, there is a general inverse relationship between the complexity of the biological resources and the societal resources available for research. Thus, it is unlikely that scientific studies will ever be able to identify source and sink areas on the scale needed for multispecies fishery management, particularly in tropical areas.

The only obvious source areas of importance, and the exception to the above, are areas of spawning aggregations. Spawning aggregations are unique and relatively short tem events. The history of fishing has shown that these must be protected if they are to persist. While it is desirable to include such aggregations within the boundaries of marine reserves, this is not always possible. Fortunately, there are other ways of protecting such aggregations, such as seasonal or area closures.

Would poorly located marine reserves "cause harm" by focusing fishing on source populations? There are two arguments against this. The first compares marine reserves with other potential management measures. Any measure that restricts fishing on some part of the resource will increase it on the remaining portion. For example, an increases in mesh size or size limits are another way to increase spawning biomass. Like marine reserves, they will reduce the exploitable stock in the short run, but they differ in that they force fishermen to concentrate on larger individuals. Ironically, it is those individuals that are most valuable as spawners, and their removal due to increased fishing pressure would have a significant deleterious impact. On the other hand, the short-term decline in catch might also drive some individuals out of the fishery, thus offsetting the impact of the regulations. In comparison, then, marine reserves may not have the conceived deleterious effects, and may be better than alternative and more accepted management measures.

However, the best argument in favor of marine reserves is the substantial empirical evidence that marine reserves create source areas by increasing spawning stock (Roberts in press). For example, the effects of the Exuma Keys Land and Sea Park in the Bahamas were studied by Sluka et al. (1997) for Nassau groupers and Stoner and Ray (1996) for conch. The former study shows that a 20% areal closure produces 60% of the eggs. The latter study showed adult density in the closed area to be 4.6 times that outside and that resulting densities of early stage larvae inside the closed area were typically an order of magnitude greater that outside.

MANAGEMENT AND MARINE RESERVES

The deepest and most severe criticism of the source-sink argument against marine reserves is that it fails to understand the fundamental nature and importance of marine reserve networks. First and foremost, marine reserves serve multiple management objectives in addition to enhancing spawning stock. Some of these management goals (e.g., provide control areas and buffers against management failure, maintain system integrity, diversify benefits) are simply not achievable through any other mechanism. Furthermore, marine reserves have benefits that transcend fisheries management alone, providing additional scientific, educational, economic and social benefits (Bohnsack, 1998). Second, the principles of reserve replication and network design specifically serve to counter uncertainty in our knowledge and management of biological systems. This specifically includes unknowns in siting reserves relative to source and sink areas, while further accounting for the obvious fact that species will differ in this regard. Assuming detailed data will not be available in most cases, the replication criterion of marine reserve networks becomes paramount as protection from mistakes in siting any particular reserve.

Following basic guidelines such as given in Table 1, marine reserves can be established now, for the betterment of resource managers and users. This is said despite the certainty that future scientific work will enhance our knowledge of how reserves work and where they may be best located. Developing a network of reserves is a long term, often multidecadal process (Ballantine 1992). Indeed, getting just one reserve established frequently takes years. It requires a fundamental shift in management approach, often obtained only by slowly building a popular constituency to demand change. The start of this social process should not be hindered by science, when science can, in fact, provide suitable guidelines for making good decisions.

ACKNOWLEDGMENTS

The ideas expressed in this paper were developed within the context of the University of Puerto Rico Sea Grant College's Marine Fisheries Reserve Initiative and specifically grant R-31-1-98.

LITERATURE CITED

Appeldoorn, R.S. and C.W. Recksiek. 1998. Guidelines for the design of marine fisheries reserves, with emphasis on Caribbean reef systems. Presented at the Mote Symposium on Essential Fish Habitat and Marine Reserves, November 1998, Sarasota, FL.

Acheson, J.M. and J.A. Wilson. 1996. Order out of Chaos. The case for parametric fisheries management. *Amer. Anthropol.* 98: 579-594.

Ballantine, W.J. 1991. Marine reserves for New Zealand. *Univ. Auckland, Leigh Lab. Bull.* 25, 200 pp.

Bohnsack, J.A. 1998. Application of marine reserves to reef fisheries management. *Aust. J. Ecol.* **23**:298-304.

Johannes, R.E. 1981. *Words of the Lagoon. Fishing and Marine Lore in the Palau District of Micronesia.* Univ. California Press, Berkeley, CA. 207 pp.

Kuhn, T.S. 1962. *The structure of scientific revolutions.* Univ. of Chicago Press, Chicago. 172 pp.

Recksiek, C.W. and R.S. Appeldoorn. 1998. In Pursuit of Design Criteria for Marine Fishery Reserves. *Proc. Gulf Carib. Fish. Inst.* **50**:372-384.

Roberts, C. Selecting marine reserve locations: optimality vs opportunism. *Bull. Mar. Sci.* In press.

Sluka, R., M. Chiappone, K.M. Sullivan and R. Wright. 1997. The benefits of a marine fishery reserve for Nassau grouper *Epinephelus striatus* in the central Bahamas. *Proc. 8th Int. Coral Reef Sym.* 2:1961-1964.

Stoner A.W. and M. Ray. 1996. Queen conch, *Strombus gigas*, in fished and unfished locations of the Bahamas: effects of a marine fishery reserve on adults, juveniles, and larval production. *U.S. Fish. Bull.* 94:551-165.

Benthic Mapping from Fish and Habitat Transect Data Using GIS Technology

RICHARD S. APPELDOORN[1], KIMBERLY FOLEY[1]
EDWIN MOLINA[1], and CONRAD W. RECKSIEK[2]

[1]*Department of Marine Sciences*
University of Puerto Rico
Mayagüez, Puerto Rico 00681-9013
[2]*Department of Fisheries, Animal and Veterinary Science*
University of Rhode Island
Kingston, RI 02881 USA

ABSTRACT

The mapping of benthic habitats and corresponding fish fauna is essential for understanding the role of habitat in controlling the spatial distribution of fish abundance, diversity and production. Any approach to mapping must first standardize habitat definitions and classification, as well as a data collection methodology. Visual transects are a standard tool for quantifying fish abundance and habitat characteristics. Yet, for purposes of mapping and analysis these data must be scaled up to their relative areas. We report on a methodology using stratified sampling and Geographic Information System (GIS) technology to convert transect data into larger-scale habitat maps, illustrating the procedure with data from La Parguera, Puerto Rico. Sampling strata are determined using two criteria. The first is Lindeman's cross-shelf habitat (CSH) classification system adapted to the local insular shelf; the main determinants are position in the cross-shelf direction and depth. The second criterion is based on a visual inspection of the array of different habitat mosaics (100's m^2) present within any CSH stratum (e.g., grassbed, sand/algae plain, gorgonian/coral field). Representative transects, replicated where possible, are placed in the different habitat-mosaics, and associated fishes and habitats are quantified. Areas within each habitat-mosaic stratum can be determined using a variety of approaches such as field mapping with Global Positioning System (GPS) coordinates, or remote sensing using aerial photography or side-scan sonar. Use of GIS technology allows transect data to be overlain on the habitat-mosaic and CSH strata and facilitates further analyses such as calculation of total abundances and estimation of both fish and habitat diversity.

KEY WORDS: Habitat, Fish Abundance, GIS

INTRODUCTION

Fisheries managers are now recognizing the importance of habitat in understanding and managing marine resources. Habitat plays a crucial role in the spatial distribution of fishes (Green 1996, Appeldoorn et al. 1997, Friedlander and Parrish 1998). The types, abundance and arrangement of habitats are also important for determining the fisheries productivity of a system. Specific locations act differentially as nursery and feeding grounds for juvenile and adult marine organisms, and as sites for spawning aggregations. Maintaining the integrity of these areas is necessary for the persistence of these functions. The flow of energy between habitats and to higher trophic levels is in large part controlled by the movement of fishes, and this, in turn, is influenced by the distribution of habitat (Appeldoorn et al. 1997, Kramer and Chapman 1999). Whether fishes move through and across habitats in feeding migrations, ontogenetic migrations, or predatory movements, they are bringing about a net transfer of energy. This provides the energetic link between one habitat and another. Understanding the role of habitat in controlling these important processes allows for their protection. Furthermore, with this understanding, easily collected habitat data may serve as a surrogate for more detailed life-history data, thus facilitating management. For example, distributions of habitat may be useful for determining those areas that are truly critical for the system ("Essential Fish Habitat") or for siting marine reserves (e.g., Recksiek and Appeldoorn, 1998; Recksiek et al., 2000).

Habitat classification systems provide a logical framework for organizing the collection and interpretation of data on both habitat and fish distributions (Lindeman et al. 1998, Recksiek and Appeldoorn 1998, Mumbry and Harborne 1999). In Lindeman's Cross-Shelf Habitat (CSH) matrix (Lindeman et al. 1998), each cell describes a unique combination of specifically defined habitat types (at 1 m^2 resolution) and geomorphic position. The main determinants of the latter are position in the cross-shelf direction and depth. Diver-based habitat characterization and visual census are commonly applied field methodologies, but they are spatially limited in both the area covered and the area over which their results can be extrapolated. Application of results to a CSH matrix allows one to determine where species exist in matrix space and if habitat use is broad or narrow, and hence potentially limiting. However, matrix space is not geographic space, and application of the CSH matrix alone cannot determine how much area is occupied by each cell, how this area is divided (number of patches, size distribution of patches) and how this area is arranged in the larger landscape. Thus, there needs to be a mechanism to allow detailed studies to be scaled up to shelf-wide dimensions. This is true not only for habitat data but for fish sampling as well. Since species distributions will be related to habitat distributions, one must sample the array of habitats and expand these to larger scales to estimate overall abundances and diversities.

Here we report on a methodology for expanding small-scale estimates of habitat and species abundances and distributions. For purposes of illustration, this methodology is applied to the forereef of an outer emergent reef off the southwest coast of Puerto Rico.

METHODS

Basic input data for our approach are maps of habitat and fish distributions. These were obtained by divers visually mapping distributions across 24 x 4 m transects, using the methodology of Recksiek et al. (2000). Habitats are described on a 1-m^2 basis from a pre-defined list of 18 potential habitat types based on predominant benthic cover and its relief (sand, algae, coral, rubble, etc.), while the position of individuals or schools of fish from 45 species are recorded onto the habitat map. Also recorded are the number of fish at each location and their size range (see Recksiek et al., 2000 for details and examples). In order to relate individual transects to the distribution of fishes and habitats across a larger spatial scale, transect positions must be determined within some kind of sampling framework. We use a two-stage, stratified sampling protocol. Strata are determined using two criteria. The first is a cross-shelf habitat (CSH) classification system (Lindeman et al. 1998) adapted to the local insular shelf (Figure 1). The second criterion is based on a visual inspection of the array of different habitat mosaics (100's m^2) present within any CSH stratum (e.g., grassbed, sand/algae plain, gorgonian/coral field). Use of this second stratification procedure ensures that the complete variety of habitats within an area will be sampled and, thus, that the vertical components of the CSH matrix will be identified for each area. This procedure allows us to relate habitat abundances on a spatial scale, thus allowing for estimation of scale specific indices of habitat diversity and the description of the arrangement of habitats on a scale larger than that of the original transects.

In practice, a given geomorphic area is chosen as the first stage for sampling, based in general terms on the CSH strata. For example, on Laurel, a large emergent offshore reef off La Parguera, Puerto Rico (Figure 2), we sampled three general areas: the forereef, the backreef and the eastern end of the reef. These each constitute a first stage of our multistage approach. Within each area fall several CSH strata. Each general area is then surveyed to assess the kinds of different habitat mosaics present. Assessments are qualitative, but are made on the basis of visually distinct boundaries. For example, on the forereef of Laurel, four such zones were identified (from shallow to deep): a fire coral zone with high relief, a low relief barren hard bottom zone, a gorgonian zone and a mixed coral gorgonian zone. Representative transects, replicated where possible, are then placed in the different mosaics. If a habitat mosaic crosses a CSH stratum boundary, then transects would normally be located in both areas. The locations of transects are determined with differentially corrected Global Positioning System (GPS).

We use a Geographic Information System (GIS) to organize, integrate, manage and analyze all data. The GIS software used was Intergraph's GeoMedia Pro v3.0x, chosen for its shallow learning curve, its full compatibility with multiple GIS platforms (e.g., ArcInfo, ArcView, etc.) and full integration with Microsoft's popular Operating Systems. To insure spatial accuracy, the 24 x 4-m transect grid was recreated using Computer Aided Design (CAD) software. The grid was georeferenced following the State Plane Coordinate System (zone number 5200: Puerto Rico and US Virgin Islands) as the projection parameter standard and used as a basic template to recreate the transects in the GIS. Habitat and fish location maps were scanned and digitized. Data on fish species, number and size of individuals were entered into the GIS database.

For each transect, the digitized maps were vectorized and other data (habitat, fish) were linked to their respective polygonal areas. A color-coded scheme was used to depict each of the 18 habitats types, while fishes were placed as either points, depicting individuals, or as polygons, depicting aggregations (see Recksiek et al., 2000 for examples). The GIS calculated habitat areas, perimeters and relationships among fishes and the habitats they occupied.

For the expansion of transect data to a larger scale, it is necessary to determine the spatial extent of the strata sampled. We used a bathymetric map to determine the location of the CSH strata (Figure 2). This was scanned and digitized, then geo-referenced and vectorized within the GIS. Within the general area of the transects (selected area), the locations of the 9 and 18-m depth contours (boundaries of the deeper CSH strata) were determined using a fathometer and GPS. These were entered into the GIS to aid in geo-referencing the CSH map. It was also necessary to determine the spatial extent of the habitat mosaic strata. Again, within the selected area encompassing all transects, boundaries of the habitat mosaics were marked at intervals with surface buoys, and their positions were determined with GPS. These points were entered into the GIS database and used to describe polygons for each habitat mosaic. For comparative purposes, we also examined the possibility of using aerial photographs to determine the extent of habitat mosaics. An aerial photograph (LARSIP NASA Depository, 1994) of Laurel was digitized, geo-referenced and vectorized in the GIS. Easily distinguished features were converted into polygons. Their biotic characteristics were determined in the field by divers. The GIS was used to calculate the areas of the cross-shelf and habitat-mosaic strata and their intersections.

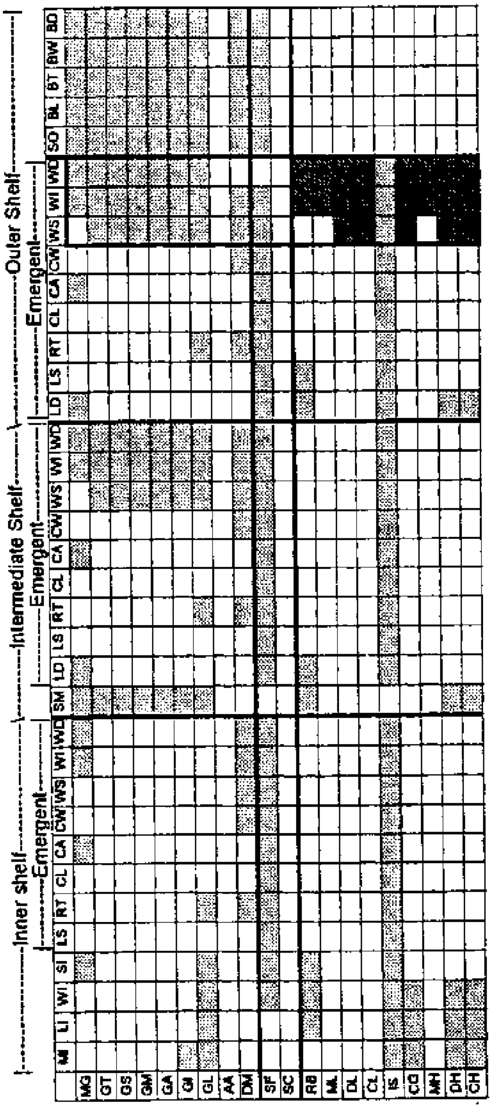


Figure 1. Cross-shelf habitat matrix for La Parguera, Puerto Rico. Vertical axis is habitat type; horizontal axis is cross-shelf stratum. Light gray boxes indicate habitat-cross shelf combinations thought not to occur. The box in bold shows the area sampled: forereef of Laurel. WS: Windward Shallow, WI: Windward Intermediate, WD: Windward Deep. Dark gray boxes indicate habitat-cross shelf combinations found in the habitat transects. See Figure 4 for habitat codes.

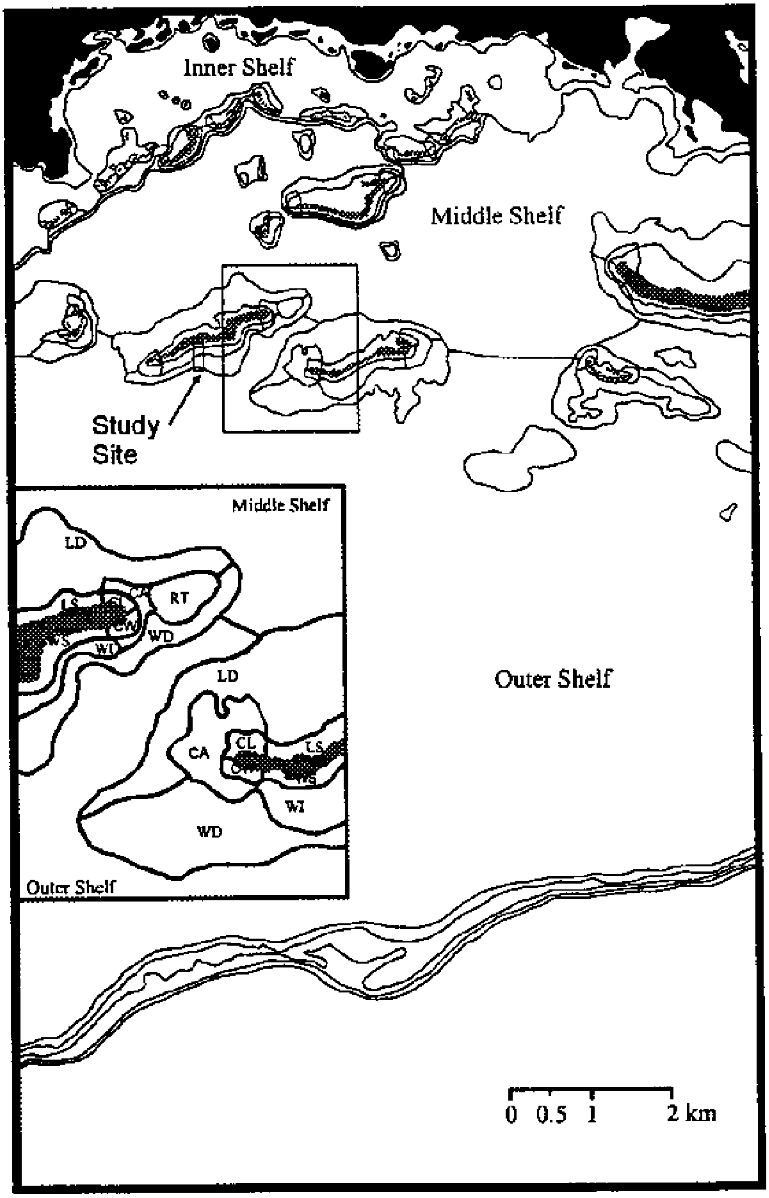


Figure 2. Cross-shelf strata for La Parguera, Puerto Rico based on bathymetry. Study site on Laurel is the small box indicated by the arrow.

A first estimation of habitat diversity was calculated by applying Shannon's index (Shannon and Weaver 1949)

$$H' = -\sum p_i \operatorname{Log} p_i$$

where p_i is the probability of encountering a patch of habitat type i. Probabilities were calculated as the number of patches of habitat type i in a stratum divided by the total number of habitat patches in that stratum. Thus, this formulation does not directly account for differences in the size of individual patches, nor the proportion of area occupied by a particular habitat type.

Calculations of fish abundances and diversity follow directly from the expanded strata areas using similar calculations. For brevity we report only the results for habitat.

RESULTS

On the forereef of Laurel, habitat and fish distribution data were collected from 8 transects, representing two replicates in each of the four types of habitat mosaics. These are shown in Figure 3. The area studied covered three cross-shelf strata: Windward Shallow (WS), Windward Intermediate (WI) and Windward Deep (WD) from an outer emergent reef; their position on the CSH matrix is outlined in Figure 1, while their geographic position is given in Figure 2. For logistical reasons, no transects were located in WD; but since the transects in WI were located near the WI-WD boundary (Figure 3), and because they shared the same habitat mosaic, we extrapolated the results of the deepest two transects into the WD stratum. Table 1 gives the areas of the cross-shelf and habitat-mosaic strata and the their intersections, as calculated within the GIS.

The basic unit for the expansion of the habitat areas to larger scales was the sum of each pair of replicate transects (192 m^2); this consisted of the number of different habitat types, the number of patches of each habitat and their respective percent areas. To expand transect results, the following procedure was used. If a cross-shelf stratum consisted of a single habitat mosaic, the percent area of each habitat type from the transects was multiplied by the area of the stratum to get the total area of each habitat type. The total stratum area divided by the area of the transects (192 m^2) was used to multiply the number of patches within the transects to get the total number of patches of each habitat within the stratum. When the cross-shelf stratum consisted of more than one habitat mosaic, the *selected area* of each mosaic within the stratum was divided by the area of the transects. This number was then used to multiply the transect results (number of patches and their proportional areas) for each mosaic. The results from the two habitat mosaics were then added to obtain the results for the stratum as a whole within the selected area. This was then used to expand the results for the entire stratum.

For example, two habitat mosaics (fire coral, bare hard bottom) were found in the WS stratum (Figure 3). Within the selected area around the transects, the WS stratum occupied 6,934 m^2, of which 1,365 m^2 was of the fire coral habitat mosaic and 809 m^2 was of the bare hard bottom mosaic (Table 1). Within the fire coral habitat mosaic four habitat types were found (Table 2). The area of this habitat mosaic within the selected area of the WS stratum (1,365 m^2) divided by the area of the transects (192 m^2) is 7.1. Thus, the number of patches and areas of each habitat within the fire coral mosaic were multiplied by 7.1 to get the total number for each habitat within the selected areas of the WS stratum.

Table 1. Areas (m^2) of cross-shelf and habitat-mosaic strata and their intersections. WS: Windward Shallow, WI: Windward Intermediate, WD: Windward Deep

	Total	WS	WI	WD
Selected Area	16,815	2,174	6,934	7,708
Total Area	446,641	54,242	146,264	246,130
Fire Coral	1,365	1,365		
Bare Hard Bottom	1,050	809	241	
Gorgonian	1,766		1,704	
Coral-Gorgonian	12,695		4,988	7,708

Table 2. Spatial coverage of habitats within transects located within the fire coral habitat mosaic. Area is in m^2. N is number of patches of each habitat observed in the transects.

Habitat Type	Area	% Area	N
Coral-High Relief	41	20.8	19
Coral-Low Relief	1	0.6	1
Dead Coral-High Relief	27	14.3	18
Dead Coral-Low Relief	123	64.3	4

A similar set of calculations was made for the bare hard bottom habitat mosaic, and the sum of these were added to the results obtained for the fire coral mosaic. The proportion of each habitat thus obtained within the selected area of the WS stratum would be multiplied by the total area of the stratum to get the final estimates of the total area of each habitat within the WS stratum.

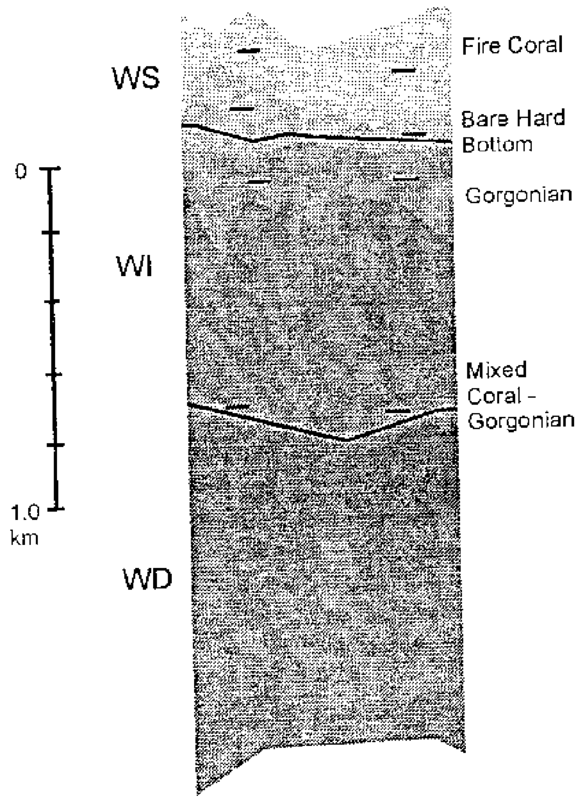


Figure 3. Selected area of the forereef of Laurel. Short black lines show position of transects (drawn larger than scale); long black lines show the boundaries of the cross-shelf strata: WS - Windward Shallow, WI - Windward Intermediate, WD - Windward Deep. Different colored areas represent the four habitat mosaics.

The results of the above procedure applied to all strata are given in Figure 4, which shows the percent area for each combination of habitat and cross-shelf stratum over the entire forereef. The cell areas in Figure 4, then, represent the relative areas of the cells in the CSH matrix (Figure 1). Thus, we have converted matrix space into true geographic space. The figure shows how the percent areal coverage of the different habitat types changes among the cross-shelf strata. For example, dead coral-low relief is the habitat most abundant in the WS stratum, but its importance is low in the remaining cross-shelf strata and for the forereef as a whole.

Habitat diversities (H') calculated for each cross-shelf stratum were as follows: WS = 0.559, WI = 0.806, WD = 0.747. These results compare well with the habitat distributions shown in Figure 4. The shallow stratum has fewer habitat types and a more uneven distribution, both contributors of lower diversity. Interestingly, Figure 4 is based on total area coverage, while the Shannon index is based on number of patches of habitat.

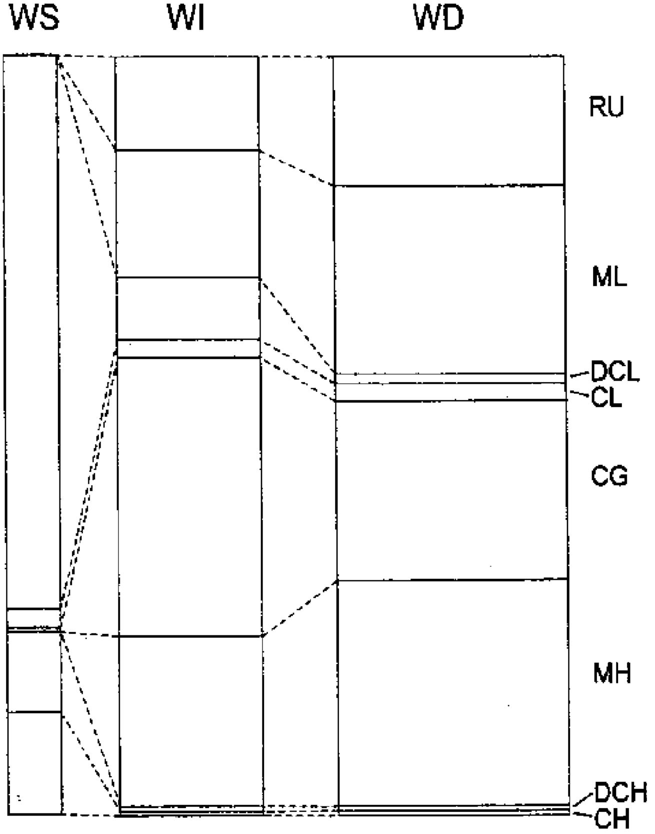


Figure 4. Bottom habitat cover within the three windward cross-shelf strata of Laurel (WS - Windward Shallow, WI - Windward Intermediate, WD - Windward Deep). Within each cross-shelf stratum, the area of each habitat type represents it proportional to bottom cover. RB: rubble, ML: mixed invertebrate-low relief, DL: dead coral-low relief, CL: coral-low relief, CG: gorgonian, MH: mixed invertebrate-high relief, DH: dead coral-high relief, CH: coral-high relief.

DISCUSSION

There are several advantages to the approach outlined here. One of the principal ones is that it allows use of transect data. Such data are often available, either from past studies or because it is easily collected and does not require expensive or complicated technologies. Such data are easily placed within the logical framework of the CSH matrix.

Using our double stratification method (cross-shelf strata and habitat mosaics) increases the probability of sampling the range of existing habitats and species, while still allowing expansion of results to larger scales based on easily obtainable bathymetric data. Because species distributions are related to habitat distributions, this approach can be used to reduce the variance on estimates of population abundance. Species-habitat locations in the CSH matrix can identify critical habitats, and the methods used here can show the availability and distribution of such critical habitat.

The method is flexible and can use different approaches to define habitat mosaics. Remote sensing (e.g., aerial photography, side scan sonar) is often used to map habitat mosaics. This usually has a coarser resolution but broader areal coverage. In our case, a georeferenced aerial photograph of Laurel, when overlain on the diver-generated map of the habitat mosaics, clearly showed the boundary between the bare hard bottom and gorgonian habitat mosaics. However, it failed to differentiate between the fire coral and bare hard bottom mosaics or between the gorgonian and mixed coral-gorgonian mosaics. The best method will, of course, depend on the goals of the study and the resources available.

ACKNOWLEDGEMENTS

This project was funded by a grant from the University of Puerto Rico Sea Grant College Program (R-31-1-98).

LITERATURE CITED

Appeldoorn, R.S. 1997. Dispersal rates of commercial important coral reef fishes: what do tagging studies tell us about potential emigration from marine reserves. *Proc. Gulf Carib. Fish. Inst.* **49**:54-63.

Appeldoorn R.S., C.W. Recksiek, R.L. Hill, F.E. Pagan, and G.D. Dennis. 1997. Marine protected areas and reef fish movements: the role of habitat in controlling ontogenetic migration. *Proc. 8th Int. Coral Reef Symp.* **2**:1917-1922.

Friedlander, A.M., J.D. Parrish, J.L. 1998. Habitat characteristics affecting fish assumblages on a Hawaiian coral reef. *J. Exp. Mar. Biol. Ecol.* **224**:1-30.

Green, A.L. 1996. Spatial, temporal and ontogenetic patterns of habitat use by coral reef fishes (Family Labridae). *Mar. Ecol. Prog. Ser.* **133**:1-11.

Kramer, D.L., and M.R. Chapman. 1999. Implications of fish home range size and relocation for marine reserve function. *Env. Biol. Fish.* 55: 65-79.

Lindeman, K.E., G.A. Diaz, J.E. Serafy and J.S. Ault. 1998. A spatial framework for assessing cross-shelf habitat use among newly settled grunts and snappers. *Proc. Gulf Carib. Fish. Inst.* **50**:385-416.

Mumby, P.J., and A.R. Harborne. 1999. Development of a systematic scheme of marine habitats to facilitate regional management and mapping of Caribbean coral reefs. *Biol. Cons.* **88**:155-163.

Recksiek, C.W., and R.S. Appeldoorn. 1998. In pursuit of design criteria for marine fishery reserves. *Proc. Gulf Carib. Fish. Inst.* **50**:372-384.

Recksiek, C.W., R.S. Appeldoorn and B.R. Murphy. 2001. Integrating fish fauna and habitat assessments: A first step in developing marine fishery reserve design criteria. *Proc. Gulf Carib. Fish. Inst.* **52**:654-666.

Roberts, C.M., R.F.G. Ormond. 1987. Habitat complexity and coral reef fish diversity and abundance on Red Sea fringing reefs. *Mar. Ecol. Prog. Ser.* **41**:1-8.

Shannon, C.E., and W. Weaver. 1949. *The Mathematical Theory of Communication.* Univ. Illinois Press, Urbana, IL.

Movement Patterns of Red Snapper (*Lutjanus campechanus*), Greater Amberjack (*Seriola dumerili*), and Gray Triggerfish (*Balistes capriscus*) in the Gulf of Mexico and the Utility of Marine Reserves as Management Tools

G. WALTER INGRAM, JR.[1] and WILLIAM F. PATTERSON, III[2]
[1]*Department of Marine Sciences*
University of South Alabama
Mobile, AL 36688 USA
[2]*Coastal Fisheries Institute*
Louisiana State University
Baton Rouge, LA 70803 USA

ABSTRACT

In the Gulf of Mexico, reef fishes represent important commercial and recreational resources, and recently many concerns have arisen over the future of several reef fish stocks. The Reef Fish Fishery Management Plan was implemented in November 1985 to rebuild declining reef fish stocks, including red snapper (RS), greater amberjack (GAJ), and gray triggerfish (GTF).

Based on tagging studies conducted in the northern Gulf of Mexico, the movement patterns of the above three species were compared. Of the 2,932 RS and 206 GTF tagged off the Alabama coast (1995 - 1998), 427 and 42 have been recaptured, with 195 (46% of recaptures) and 28 (67% of recaptures) being recaptured at the site of release, respectively. The greatest effect on magnitude of RS and GTF movement was that of tropical cyclones; the magnitude of movement of fish not at liberty during tropical cyclones were significantly less than those at liberty during storms. Of 564 and 614 GAJ tagged off Pensacola and Panama City Beach, Florida (1989 - 1995), 178 (32%) and 118 (19%) were recaptured, with 173 (97% of recaptures) recaptured in the permit (release) area and 105 (90% of recaptures) recaptured within 30 km of the point of tagging and release, respectively.

We reason that reef species exhibiting high area fidelity may profit from the shelter of a no-take marine reserve, which may lead to an increase in spawning stock biomass (SSB). Since each of these species have pelagic larvae, which may be transported throughout the Gulf by oceanic currents, an increase in SSB in a reserve may increase the production and export of propagules. Also, tropical cyclones may facilitate the export of adult biomass from a no-take marine reserve, therefore restocking surrounding areas. However, due to interspecific differences in site fidelity, the relative effectiveness of marine reserves were evaluated on a species specific basis.

KEY WORDS: Marine reserves, movement patterns, reef fisheries

INTRODUCTION

Reef fishes represent important commercial and recreational resources in the Gulf of Mexico (Gulf), and within the past decade and a half many concerns have arisen over the future of several reef fish stocks. These concerns include declining recreational and commercial landings of many reef species and spawning stock biomass (SSB) estimates which, in some cases have fallen below threshold values (GMFMC 1989). The Reef Fish Fishery Management Plan was implemented in November 1985 (GMFMC) to rebuild declining reef fish stocks. There are numerous species managed under this plan, including the three species which are the subject of this paper.

A large problem with fishery management today is the practice of development of unique regulations for each species, without regard for environmental variability and interactions between species or a species complex (Bohnsack 1993, Lauck et al. 1998). While community-level dynamics may never be fully understood, there is little chance for effective management given this practice (Lauck et al. 1998). Also, it is not uncommon for fishing effort to shift from one species, on which regulations have been imposed, to another species, requiring future management regulations on this newly fished species (Lauck et al. 1998). Marine reserves may provide an effective management tool to overcome the weaknesses of current management practices, especially with regard to ecological complexity and the effects of shifts in fishing effort between species due to imposition of regulations (Bohnsack 1993, Lauck et al. 1998).

The utility of a marine reserve depends on the life histories and behavior of the species under management. We reason that reef species exhibiting high site and/or area fidelity may profit from the shelter of a no-take marine reserve, which may lead to a local increase in SSB. Also, we reason that tropical cyclones may facilitate the export of adult biomass from a no-take marine reserve by causing an increase in movement, therefore restocking surrounding areas. However, due to interspecific differences in area fidelity, the relative effectiveness of marine reserves must be evaluated on a species specific basis. Based on data gathered from the northern and western Gulf, movement patterns are analyzed and compared, and the utility of marine reserves to the management of these three species is discussed.

Red snapper (RS) support one of the most important commercial and recreational fisheries in the Gulf (Schirripa and Legault 1997). Despite the findings of Gold et al. (1997), indicating a single panmictic stock of RS in the Gulf, most tagging studies have indicated that snapper have high site fidelity, with most tag recoveries occurring within 10 km of the site of tagging and release (Beaumariage 1969, Beaumariage and Bullock 1976, Fable 1980, Szedlmayer and Shipp 1994). However, some movement on the order of ten to hundreds of kilometers has been reported (Beaumariage 1969, Beaumariage and

Bullock 1976, Szedlmayer and Shipp 1994, Watterson et al. 1998). More recently, Watterson et al. (1998) reported that RS showed high site fidelity above artificial reefs in the northern Gulf; however, the authors also reported movement of some fish over 100 km. They also reported that Hurricane Opal had a significant effect on snapper movement. The study described herein was a continuation of the study reported by Watterson et al. (1998).

Greater amberjack (GAJ) support important recreational and commercial fisheries in the Gulf (Burch 1979, McClellan and Cummings 1996). There have been a number of tagging studies conducted concerning the movement of GAJ. Since 1954, 5,643 GAJ have been tagged and 642 recaptured due to the efforts of the Cooperative Game Fish Tagging Program in the Atlantic Ocean, Gulf of Mexico, and Caribbean Sea, which is managed by the National Marine Fisheries Service's Southeast Fisheries Science Center (Scott et al. 1990). Scott et al. (1990) speculate that the high tag recovery rate of 11.4% is due to the abundance of amberjacks in inshore waters and the active recreational fishery. This fishery is responsible for the majority of the tagging effort. They report the longest distance traveled by a GAJ was about 2,400 km from central west Florida to Venezuela, and the record time at large for this species is just over 10 years for a fish that was tagged and recaptured off Jacksonville, Florida. Also, a tagging study was conducted by Mote Marine Lab, targeting cobia, dolphin and GAJ, from October, 1990 to July, 1992 (Burns and Neidig 1992). During this project a total of 785 GAJ were tagged and released from many locations on the Gulf and Atlantic coasts of Florida. Of these, 46 were recaptured, with only 12 showing any net movement. One of the 12 which moved did so from Florida to Texas (~850 miles) in 252 days. These results are consistent with the findings of Burch (1979), who suggests that GAJ may temporarily leave then return to the area on an annual basis.

Until recently gray triggerfish (GTF) were not prized by recreational and commercial fishermen, unlike the two previously described species. Hence, the GTF did not support an economically important directed fishery in the Gulf. Increases in regulations on other reef species and increased awareness of the excellent flesh of gray triggerfish have created a demand for this previously underutilized fishery resource.

To date, however, there have been few studies concerning movement of adult GTF. Ofori-Danson (1990) reported that GTF off the West African coast in the eastern Atlantic Ocean move from cold coastal waters as a result of seasonal upwelling (during the third quarter of the year) to join an offshore stock which is present throughout the year. In contrast, Beaumariage (1964) and Johnson and Saloman (1984) reported that Gulf GTF may be less migratory. For example, Beaumariage (1964) found that 38 of 103 GTF, which were tagged and released in the northeastern Gulf, were recaptured by investigators (36.9% recovery rate) in the approximate area of initial capture and release.

MATERIALS AND METHODS

The data sets used in this paper come from two different sources. The first source of data comes from a tagging study, which was part of a larger dissertation project concerning the population dynamics of RS (Patterson 1999). During this tagging study, GTF were opportunistically tagged during the research cruises. RS and GTF were captured at nine sites in the northcentral Gulf of Mexico off the coast of Alabama in the Hugh-Swingle General Reef Permit Area, between 20 and 30 km south-southeast of Dauphin Island, Alabama between March 22, 1995 and July 20,1998 (Figure 1). The sites, which were artificial reef structures, were arranged in a 3 × 3 grid; three reefs were in each of three depth strata: 20, 26, and 35 m. Fish were taken by hook and line, measured for length and tagged abdominally with an individualized Floy® internal anchor tag. Recaptures by researchers were recorded and the fish were re-released, while recaptures by fishermen were reported via the phone number provided on each tag.

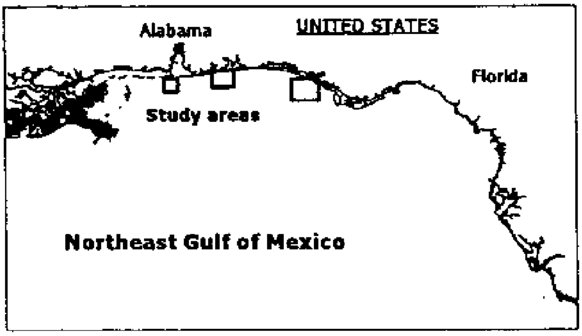


Figure 1. Study areas in the northeast Gulf of Mexico

Recreational fishermen from Pensacola and Panama City Beach, Florida provided the second source of data, which concerned GAJ tagging and movement. The fishermen used individualized dart tags, with which they dorsally tagged GAJ over artificial reefs off the coast of Pensacola and Panama City Beach, Florida from 1989 to 1995 (Figure 1). As each GAJ was tagged the total length of each fish was taken, and the release site and date were recorded. When recaptures were made, a phone number provided on each tag enabled the location of recapture and size of the GAJ to be reported.

A number of statistical procedures were used to describe and compare movement patterns of RS, GAJ, and GTF. The distance traveled from the point of tagging and release, if any, and the days at large of all recaptured fish were recorded. Polar graphs of distances moved from the point of tagging and release were developed for each species for visual analysis of movement patterns. Unbiased estimates of mean distances moved and mean velocities (distance moved/days at large) were computed for each species using the delta method due to the abundance of zero values in the data (Pennington 1983). Rayleigh's test for randomness in direction of movement was employed to determine if there was significant directional movement in each of the three species (Batschele 1981). Parametric and non-parametric analyses of variance were employed to characterize the effects of tropical cyclones, fish size, time at large, etc., on the movement patterns of each species at an $\alpha = 0.05$. Finally, graphs of cumulative frequency distributions of each species recaptured within three years of tagging versus the distance moved from the point of tagging and release were developed to analyze the frequency of recaptures within certain radii of distances moved.

RESULTS

Mean total length (± SE) of RS tagged in this study was 335.1 mm (± 1.34). Of the 2,932 RS tagged, 80% were shorter than the legal size limit of 380 mm TL; 427 were recaptured. Of the 427 RS recaptured, 195 were recaptured at their site of release and 232 were recaptured at some other location. The largest distance moved was 344 km for a RS that was at large for 400 days. The maximum time at large for a recaptured RS in this study was 1,142 days for a RS that moved 55 km to the southwest of where it was released off the Alabama coast. The mean time at large for recaptured RS was 313 days.

Statistical analysis performed by Patterson (1999) found days at large after tagging and release ($p < 0.001$), exposure to hurricanes ($p < 0.001$), residuals of the relationship between days at large and TL at recapture ($p = 0.001$), and transportation prior to release ($p = 0.033$) were significant effects in a model with the ranked distance of RS movement as the dependent variable. Similar effects were found on the ranked velocity of RS movement using the same model, without days at large as a covariate. RS that were at large during Hurricane Opal, which moved over the tagging stations on October 4, 1995, or Hurricane Georges, which moved over the tagging stations on September 27, 1998, moved greater distances and had greater velocities of movement than did RS that were not at large during hurricanes. Unbiased estimates of the mean distance (± SE) that RS moved away from the point of tagging and release and mean velocities (± SE) calculated using the delta method are as follows: overall mean distance, 12.78 km (± 0.13); overall mean velocity, 47 m/day (± 29); mean distance moved during tropical cyclones, 48.98 km (± 3.72); mean velocity during tropical cyclones, 136 m/day (± 11); mean distance moved not during tropical cyclones, 5.93 km (± 0.11); mean velocity not during tropical cyclones, 36 m/day (± 0.9). Also, Rayleigh's test for randomness in direction of movement indicted directional movement of RS during hurricanes ($p < 0.001$) and random movement in RS not a large during hurricanes ($p > 0.25$). The resultant vector of RS at large during tropical cyclones was 37.4 km at 8.0° (0° = due east). Polar graphs of RS movement patterns during and not during tropical cyclones can be seen in Figure 2.

The mean fork length (± SE) of GTF tagged was estimated to be 346.7 mm (± 4.0). Of the 206 gray triggerfish tagged, 50 recaptures were made of 42 fish, with 28 (67% of recaptures) recaptured at the site of release. The mean time at large for GTF in this study was 190 days. The longest time at large for a GTF in this study was 949 days for a fish that moved 4.1 km. The greatest distance move by a GTF in this study was 8.6 km for a fish that was at large for 54 days during which Hurricane Opal passed over the tagging stations.

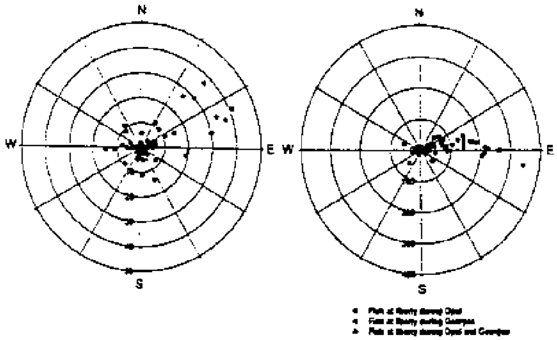


Figure 2. Movement patterns of red snapper not during (left graph: n = 284; 163 fish recaptured at release site) and during (right graph: n = 143; 24 fish recaptured at release site) tropical cyclones. Scale in km.

The greatest effect on magnitude of movement of GTF was that of Hurricane Opal, which passed over the tagging stations on October 4, 1995 (Figure 3). A Kruskal-Wallis One Way Analysis of Variance on Ranks indicated that the differences in the median values among the treatment groups (i.e., magnitude of movement of gray triggerfish tagged and recaptured before Opal, tagged before and recaptured after Opal, and tagged and recaptured after Opal) are statistically significant ($p < 0.001$). Dunn's Method, a multiple comparison procedure, showed that the magnitude of movement of gray triggerfish tagged and recaptured before Opal were not significantly different from that of fish tagged and recaptured after Opal ($\alpha = 0.05$). However, the magnitude of movement of fish tagged before and recaptured after Opal was significantly different from that of fish tagged and recaptured before Opal and tagged and recaptured after Opal ($\alpha = 0.05$). Unbiased estimates of the mean distances (± SE) and mean velocities (± SE) moved by GTF calculated using the delta method are as follows: overall mean distance, 1.64 km (± 0.40); overall mean velocity, 13 m/day (± 5); mean distance moved during tropical cyclones, 7.45 (± 0.65); mean velocity during tropical cyclones, 82 m/day (± 10); mean distance moved not during tropical cyclones, 0.72 km (± 0.20); mean velocity not during tropical cyclones, 4 m/day (± 1). The Rayleigh test for randomness indicated that direction of movement of GTF not at large during Hurricane Opal is random in nature ($p > 0.25$) and directional during Hurricane Opal ($p < 0.05$) with a resultant vector of 3.86 km at 166° (0° = due east).

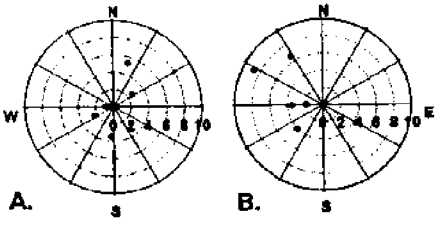


Figure 3. Movement patterns of gray triggerfish not during (A: n = 32; 25 fish recaptured at release site) and during (B: n = 5; no fish recaptured at release site) tropical cyclones. Scale in km.

The GAJ tagged in artificial reef permit areas off Pensacola and Panama City Beach, Florida (1989 - 1995) were done so opportunistically by recreational fishermen. The majority of the GAJ tagged were fish which were under the legal size limit (28 inches = 711 mm) and had to be released. There were very few GAJ tagged that were larger than the legal limit. Five hundred sixty-four GAJ were tagged by recreational fishermen in the reef permit area south of Pensacola, Florida. Thirty-two percent (178) of those GAJ tagged were recaptured. Of these recaptures, 97% (173) were recaptured within the reef permit (release) area.

An additional 614 GAJ were tagged by recreational fishermen in the reef permit area southwest of Panama City Beach, Florida. These recreational fishermen provided very detailed coordinates of tagging and recapture sites of GAJ allowing a more in depth analysis of GAJ movement (Figure 4). Nineteen percent (118) of those GAJ tagged off Panama City Beach were recaptured. Of these recaptures, the greatest time at large was 1,210 days for a GAJ that moved 92.3 km. The greatest distance moved was 403.6 km from off Panama City Beach, Florida to off Fouchain, Louisiana for a GAJ that was at large for 396 days. The mean time at large for the GAJ tagged off Panama City Beach was 200 days. There was no significant directional movement or effect of tropical cyclones on the movement of GAJ off Panama City Beach found during statistical analysis. The only large tropical cyclone that occurred during this study was Hurricane Andrew, and this storm passed well south of the tagging area. However, the GAJ which moved the greatest distance was at large during this storm. Days at large after tagging and release ($p < 0.0001$) was a significant effect in a model, with the ranked distance of GAJ movement as the dependent variable. As days at large increased so did the value of the ranked distance. Unbiased estimates of the mean distance moved (±SE) and mean velocity (±SE) of GAJ off Panama City Beach calculated using the delta method was 10.8 km (± 8.9) and 1.07 km/day (± 0.88), respectively.

DISCUSSION

Before we compare movement patterns of these three species and begin the discussion of the utility of marine reserves, we first discuss the validity of these tagging studies. To be valid these studies have to meet certain assumptions. First, we address the assumption of no effect of tagging on mortality for each species. Estimates of acute mortality rate caused by tagging were calculated for both RS and GTF using a scale of relative condition of the fish at release. Assuming only fish released in good condition survived, acute mortality rates for RS and GTF due to tagging were estimated to be 15% and 2%, respectively. No direct estimate of acute mortality rate due to tagging of GAJ was calculated during this study. However, during the tagging study conducted by Burns and Neidig (1992) condition of GAJ after tagging and release was recorded with 4.3%

in a condition other than excellent or good. As seen by the above estimates of acute mortality, these studies basically met the assumption of no effect of tagging on mortality with the possible exception of RS.

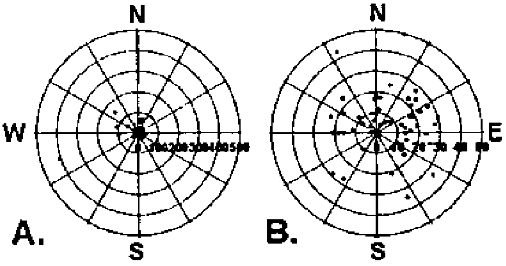


Figure 4. Movement patterns of greater amberjack tagged off Panama City Beach, Florida. A: Scale to 500 km. B: Scale to 50 km

Next, we address the assumption that all the tagged fish were recognized, which depends largely on tag retention. Using logistic regression, retention rates of the external portion of internal anchor tags were estimated for both GTF and RS. Retention rates (± 95% C.I.) were estimated to be 93% (88 - 96%) and 92% (82 - 97%) for fish at large for six months, 77% (68 - 86%) and 86% (66-95%) for fish at large for one year, and 18% (7-40%) and 38% (0.1-99%) for fish at large for two years, for RS and GTF, respectively. As for GAJ, no direct estimation of tag loss was made. However, Dunning et al. (1987) conducted a double tagging study on striped bass, *Morone saxatilis*, and estimated that retention rates of internal anchor tags was 98% after two years, while the retention rate of t-bar anchor tags placed dorsally on the striped bass, similar to those used to dorsally tag GAJ, was only 42% after one year. Tag loss, or the failure to recognize a tagged fish, can have significant impact on results. Due to tag loss in this study, it is possible that we are underestimating movement in each species, especially RS and GAJ, which tend to move greater distances the longer these species are at large.

Finally, the assumption of equal catchability is addressed. The first part of this assumption made is that tagged and untagged fish have the same probability of being captured. This implies that tagging a fish did not alter its behavior. No tests of this part of the assumption were conducted for any of the species in this paper, but 30 tagged RS were held for six months in the laboratory by Szedlmayer and Shipp (1994), and no changes in behavior were recorded. The second part of this assumption concerns seasonal trends in catch rates. Catch rates for both RS and GTF were higher in the fall than other seasons, which can be explained by seasonal stratification of the water column. In the late spring and summer, stratification of the water column leads to depleted bottom dissolved oxygen. Patterson (1999) found that catch per tagging station was significantly correlated with bottom dissolved oxygen, and when wind mixing broke down stratification in the fall, catch per tagging station increased. Similar trends were also found with GTF. For GAJ, Thompson et al. (1999) reported monthly changes in size and availability of GAJ, which may be due to seasonal migrations of larger fish. These seasonal patterns of catch rates could bias estimates of equal catchability low for larger fish.

One of the most important uses of a marine reserve is as a refuge from fishing pressure for some or all of the species in the area. By releasing such fishing pressure, a population may then become structured by natural mortality instead of fishing mortality (Bohnsack 1992). Thus, reserves may increase the density, average size, and SSB of species in the reserve, even large predatory species like carangids and lutjanids (e.g., Russ and Alcala 1996, Polunin and Roberts 1993, Roberts 1995). Because larger individuals tend to have higher reproductive output, a population now structured by natural mortality with more larger and older individuals in a marine reserve may have a much higher reproductive output (Bohnsack 1992). Hence, marine reserves could possibly enhance fisheries by acting as centers of dispersal of propagules and adults into the areas surrounding the reserve.

However, the utility of a marine reserve would depend on the ecology of each species in question. In this discussion we characterize the utility of a marine reserve with area fidelity using unbiased estimates of mean distance and velocity moved, coupled with analysis of the cumulative frequency of recaptures versus the distance from the point of tagging and release to point of recapture. Also, we mention the relative effective size of a reserve for each species in the context of area fidelity and reproductive patterns.

GTF were estimated to have extremely high area fidelity. This can be seen in the estimates of mean distance moved and velocity (i.e., 1.64 km and 13 m/day, respectively). Also, as illustrated by the cumulative frequency graph (Figure 5) 100 % of all GTF were recaptured within nine kilometers of the point of tagging and release. Even though there was a significant effect from exposure of GTF to Hurricane Opal and a pulse of GTF movement to the west-northwest was observed, area fidelity remained high. GTF are very territorial, have an early age at maturity, build nests during the spawning season, and have larvae and juvenile life history stages that are pelagic through the first year of life (Harper and McClellan 1997). Therefore, we reason that territorial species like GTF may benefit from small no-take marine reserves; SSB may increase in the face of no fishing pressure, since highly fidelic species exposed to fishing pressure may more vulnerable to overexploitation (Bohnsack 1989).

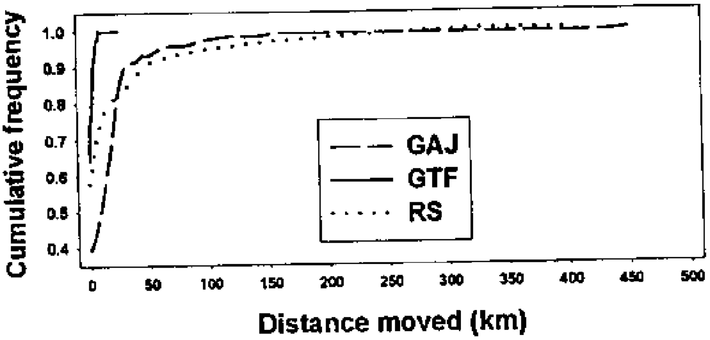


Figure 5. Cumulative frequencies of fishes recaptured versus distance from the point of tagging to the point of recapture

Eighty percent of the RS tagged in this study had total lengths less than the legal size limit of 380 mm at the time of the study. Therefore, the portion of the population that is analyzed by this study consists of relatively small fish. Patterson (1999) reports greater distances moved by RS in this study than in previous studies. However, of these small RS, 58% were recaptured at the site of release and 80% were recaptured within 20 km of the site of tagging during this study (Figure 5). Therefore, we reason that red snapper exhibit moderate area fidelity, relative to the high area fidelity of GTF, with the probability moving out of an area increasing over time. Hence, at the very least these undersized RS may benefit from escape of release mortality. Also, due to the fact that RS mature later than do GTF, do not reach their peak fecundity until around 15 years of age, and tend to move greater distances over time, RS may require a marine reserve covering a much larger area than GTF.

Surprisingly, the relatively small GAJ (< 711 mm TL) tagged off Florida were found to have a higher area fidelity than red snapper. Off Pensacola, 97%

of GAJ tagged were recaptured within the reef permit (release) area. Off Panama City Beach, there were less recaptures at the site of release than RS (39% < 58%). However, 90% of GAJ recaptures were made within 30 km of the point of tagging, while 90% of RS recaptures were made within 50 km (Figure 5). While mean velocity of GAJ is much greater than RS and GTF (i.e., 1.07 km/day, 47 m/day, and 13 m/day, respectively), the mean distance moved by GAJ is less than that of RS (i.e., 10.8 km < 12.8 km). This implies that young GAJ move around a great deal but may do so in a relatively small area. Therefore, like RS, GAJ exhibit moderate area fidelity, with the probability moving out of an area increasing over time. This last relationship may be partially explained by the findings of Thompson et al. (1999), which suggested that larger GAJ may undergo seasonal migration. However, the size at which GAJ begin to migrate is unknown (Burch 1979). Again, like RS, the relatively smaller GAJ in this study may benefit from escape of release mortality. Also, due to the facts that GAJ mature and reach peak fecundity earlier and appear to have higher area fidelity, GAJ may benefit from a relatively smaller marine reserve than RS.

We conclude that marine reserves can benefit the species addressed in this paper. However, caution needs to be used when forming marine reserves in the future. All of the fish in these studies were tagged over the shallow shelf of the northern Gulf of Mexico. RS and GTF, exhibited greater distances and velocities moved during tropical cyclones. It is reasonable to assume that RS and GTF inhabiting a marine reserve placed in a similar area would also show an increase in movement. Therefore, there may be a reduction in the effectiveness of a marine reserve placed in relatively shallow areas susceptible to perturbation by tropical cyclones with an increase in export of adult biomass to surrounding areas. Hence, due to the high frequency of tropical cyclones in the northern Gulf, offshore placement of a marine reserve may be more advantageous to these species.

LITERATURE CITED

Batschelet, E. 1981. *Circular Statistics in Biology*. Academic Press. New York. 371 pp.

Beaumariage, D.S. 1964. Returns from the 1963 Schlitz tagging program. *Florida Board of Conservation Technical Series.* **43**:1-34.

Beaumariage, D.S. 1969. Returns from the 1965 Schlitz tagging program. *Florida Department of Natural Resources Technical Series.* **59**:1-34.

Beaumariage, D.S. and L.H. Bullock. 1976. Biological research on snappers and groupers as related to fishery management requirements. Pages 86 - 94 in: H.R. Bullis, Jr. and A.C. Jones, (eds.) *Proceedings: Colloquim on snapper-grouper fishery resources of the wastern central Atlantic Ocean.*

Florida Sea Grant Colloquium Report 17.

Bohnsack, J.A. 1989. Are high densities of fishes at artificial reefs the result of habitat limitation of behavioral preference? *Bull. Mar. Sci.* **44**:631-645.

Bohnsack, J.A. 1992. Reef resource habitat protection: the forgotten factor. *Marine Recreational Fisheries.* **14**:117-129.

Bohnsack, J.A. 1993. Marine reserves: they enhance fisheries, reduce conflites, and protect resources. *Oceanus.* **Fall**:63-71.

Burch, R.K. 1979. The greater amberjack, *Seriola dumerili*: its biology and fishery off southeastern Florida. M.S. Thesis, University of Miami, FL. 112 pp.

Burns, K.M. and C.L. Neidig. [1992] Cobia (Rachycentron canadum), amberjack (Seriloa dumerili) and dolphin (Coryphaena hippurus) migration and life history off the southwest coast of Florida. MARFIN Final Report. 88 pp.

Dunning, D.J., O.E. Ross, J.R. Waldman, and M.T. Mattson. 1987. Tag retention by, and tagging mortality of, Hudson River striped bass. *N. Am. J. Fish. Manag.* 7:535-538.

Fable, W.A., Jr. 1980. Tagging studies of red snapper (*Lutjanus campechanus*) and vermillion snapper (*Rhomboplites aurorubens*) off south Texas coast. *Cont. Mar. Sci.* **23**:115-121.

Gold, J.R., F. Sun, and L.R. Richardson. 1997. Population structure of red snapper from the Gulf of Mexico as inferred from analysis of mitochondrial DNA. *Trans. Am. Fish. Soc.* **123**:386-396.

GMFMC (Gulf of Mexico Fishery Management Council). [1989] Amendment number 1 to the reef fish fishery management plan. Tampa, FL.

Harper, D.E. and D.B. McClellan. [1997] A review of the biology and fishery for gray triggerfish, *Balistes capriscus*, in the Gulf of Mexico. National Marine Fisheries Service, Miami. MIA-96/97-52.

Johnson, A.G. and C.H. Saloman. 1984. Age, growth, and mortality of gray triggerfish, *Balistes capriscus* , from the northern Gulf of Mexico. *Fish. Bull.* **82**(3):485-492.

Lauck, T., C.W. Clark, M. Mangel, and G.R. Munro. 1998. Implementing the precautionary principle in fisheries management through marine reserves. *Ecol. Appl.* **8**(1):S72-S78.

McClellan, D.B. and N.J. Cummings. [1996] Stock assessment of Gulf of Mexico greater amberjack through 1995. NMFS, SEFSC, Miami Laboratory, Miami. MIA-97/98-03. 69 pp.

Ofori-Danson, P.K. 1990. Reproductive ecology of the triggerfish, *Balistes capriscus*, from the Ghanian coastal waters. *Tropical Ecology.* **31**:1-11.

Patterson, W.F., III. 1999. *Aspects of the population ecology of red snapper,* *Lutjanus campechanus, in an artificial reef area off Alabama*. Ph.D. Dissertation. University of South Alabama, Mobile. 164 p.

Polunin, N.V.C., and C.M. Roberts. 1993. Greater biomass and value of target coral-reef fishes in two small Caribbean marine reserves. *Mar. Ecol. Prog. Ser.* **100**:167-176.

Russ, G.R., and A.C. Alcala. 1996. Marine reserves: rates and patterns of recovery and decline of large predatory fish. *Ecol. Appl.* **6**(3):947-961.

Roberts, C.M. 1995. Rapid build-up of fish biomass in a Carribean marine reserve. *Conservation Biology*. **9**:815-826.

Schirripa, M.J. and C.M. Legault. [1997] Status of the red snapper stock in U.S. waters of the Gulf of Mexico: updated through 1996. NMFS, SEFSC, Miami Laboratory, Miami. MIA-97/98-05.

Scott, E.L., E.D. Prince, and C.D. Goodyear. 1990. History of the Cooperative Game Fish Tagging Program in the Atlantic Ocean, Gulf of Mexico, and Caribbean Sea, 1954-1987. *Am. Fish. Soc. Symposium*. 7:841-853.

Szedlmayer, S.T. and R.L. Shipp. 1994. Movement and growth of red snapper, *Lutjanus campechanus*, from an artificial reef are in the northeastern Gulf of Mexico. *Bull. Mar. Sci.* **55**:887-896.

Thompson, B.A., M. Beasly, and C.A. Wilson. 1999. Age distribution of greater amberjack, *Seriola dumerili*, from the north-central Gulf of Mexico. *Fish. Bull.* **97**:362-371.

Watterson, J.C., W.F. Patterson, III, R.L. Shipp, and J.H. Cowan, Jr. 1998. Movement of red snapper, *Lutjanus campechanus*, in the north central Gulf of Mexico. *Gulf of Mex. Sci.* **16**:92-104.

Assessment of the Reef Fish Community, Habitat, and Potential for Larval Dispersal from the Proposed Tortugas South Ecological Reserve

CRAIG P. DAHLGREN[1], JACK A. SOBEL[1] and DOUGLAS E. HARPER[2]
[1] *Center for Marine Conservation*
1725 DeSales St. NW, Suite 600
Washington, DC 20036 USA
[2] *National Oceanic and Atmospheric Administration*
National Marine Fisheries Service
Southeast Fisheries Science Center
75 Virginia Beach Drive
Miami, FL 33149 USA

ABSTRACT

The Florida Keys National Marine Sanctuary is in the process of creating one or more no-take marine reserves in the Dry Tortugas region. Initial recommendations call for the creation of two reserves: Tortugas North, encompassing the northern parts of Dry Tortugas National Park, Tortugas Bank, and adjacent areas; and Tortugas South, encompassing Riley's Hump and deepwater habitats to the south. These areas are expected to be important because they include reefs with high coral cover and fish spawning sites upstream from the Florida Keys, yet little is known about the habitat characteristics, reef fish community, and larval dispersal from these areas, especially the proposed Tortugas South reserve. We conducted visual surveys in the Riley's Hump area of the proposed Tortugas South reserve to characterize its habitat composition and reef fish community. Several surveys during the full moon of late May - early June 1999, a time of expected spawning activity, were focused on areas identified as potential spawning aggregation sites for mutton snapper, *Lutjanus analis*. Satellite-tracked drifter buoys, released at the expected time and location of spawning, were used to examine potential larval dispersal pathways.

Results suggest that, although coral cover is relatively low on Riley's Hump, reef fish diversity is high and includes species that are rare elsewhere in the Florida Keys. Snapper species occurred in relatively high densities, and several female mutton snapper with ripe eggs were collected, suggesting spawning occurred at this time. Larvae spawned on Riley's Hump at this time may have been transported to nursery areas in the Florida Keys and Biscayne Bay, possibly even as far north as Palm Beach. The information we collected is essential to understanding the potential importance of the proposed reserve area for conserving biodiversity, building spawning stock biomass, and supporting fisheries throughout the Florida Keys via larval replenishment.

KEY WORDS: Habitat, larval transport, reef fish

INTRODUCTION

Studies of no-take marine reserves around the world suggest they have the potential to provide a variety of benefits related to protecting ecosystem structure and function, enhancing our understanding of natural processes influencing marine ecosystems, enhancing non-consumptive economic opportunities, and maintaining or enhancing fisheries (Sobel 1996, Bohnsack 1998). In light of this potential, marine reserves are gaining popularity around the world for both conservation and fisheries management. In the United States, the Florida Keys National Marine Sanctuary (FKNMS) contains several small reserves that protect vulnerable areas with essential reef habitats, but are expected to provide limited benefits to fisheries (Dahlgren 1999, Dahlgren and Sobel in press). The FKNMS is also proposing to establish a much larger marine reserve, the Tortugas Ecological Reserve. Because of its proposed upstream location and large size, such a reserve is expected to protect essential habitats, and provide numerous benefits related to conservation and fishery management (Dahlgren 1999, Dahlgren and Sobel in press).

Initial recommendations for the Tortugas Ecological Reserve include establishing two marine reserves in the Dry Tortugas region (Figure 1). The larger proposed area, Tortugas North (ca. 429 km^2), encompasses waters within the FKNMS, Dry Tortugas National Park, and waters under state and federal jurisdiction. The smaller proposed reserve area, Tortugas South (ca. 206 km^2), is almost entirely within federal waters and contains deep reef habitats (30+ m) on Riley's Hump and deeper habitats off of the shelf edge to the south (Figure 1). Mutton snapper, *Lutjanus analis*, spawning aggregations have been documented on Riley's Hump (Domeier and Colin 1997) and reports from commercial fishermen indicate that other snapper species spawn there (Lindeman et al. in press). At present, a 38.2 km^2 area encompassing Riley's Hump is closed to fishing in May and June to protect mutton snapper spawning aggregations. A permanent closure of Riley's Hump is hoped to protect reef ecosystems there, conserve reef fish biodiversity, and benefit fisheries in the Florida Keys. Despite the expected importance of Riley's Hump, little is known about the habitats and reef fish community there, or the potential for spawning on Riley's Hump to replenish depleted stocks elsewhere in the Florida Keys and south Florida.

The primary objectives of this study were to describe the habitat characteristics and the reef fish community of Riley's Hump, and identify potential pathways for larval transport from spawning sites. Such information is essential to determine the potential for the Tortugas South reserve to provide the anticipated conservation and fisheries management benefits. Moreover, quantitative information on the reef fish community and habitats before reserve protection is critical to establishing a baseline for evaluating reserve effects over time. We focused specifically on the Riley's Hump area of the proposed Tortugas South reserve because it is expected to be the most important part of the proposed area from a fisheries management perspective (Lindeman et al. 1999).

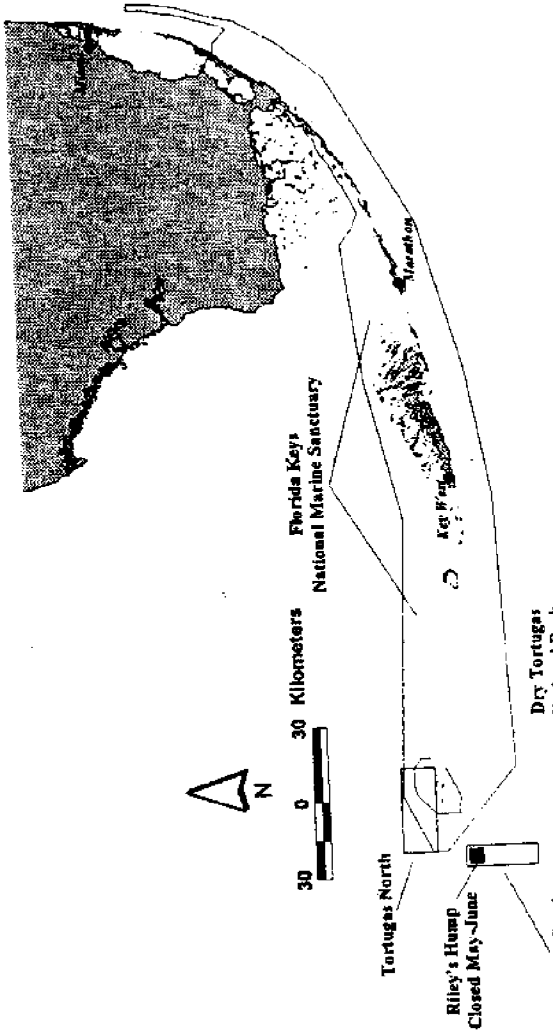


Figure 1. Chart showing the proposed boundaries of the proposed Tortugas Ecological reserve and existing protected areas in the region.

METHODS

Reef fish and Habitat

Divers quantitatively assessed the reef fish community and habitat composition of Riley's Hump at randomly selected sites. The Riley's Hump area was divided into 0.25 x 0.25 minute grid cells (cell area ca. 0.2 km^2). Bathymetric maps and information provided by local commercial fishermen were used to identify grid cells with water depths less than 33 m. Of the 50 grid cells identified, 19 were selected at random for fish and habitat surveys. Within each randomly chosen cell, divers were dropped where water depths were >33 m, and censuses conducted wherever the divers landed on the sea floor.

At each site, two divers (three divers at two sites) conducted quantitative visual censuses of fish and habitats in adjacent areas. Visual censuses followed a stationary census technique described in detail by Bohnsack and Bannerot (1986). Briefly, stationary divers identified all reef fish within a 7.5 m radius (ca. 177 m^2 area), and the cylindrical volume of water above this area, for five minutes, then estimated the abundance and size (minimum, maximum and average fork length) of each species observed. Species observed after the initial five minutes were noted, but were neither counted nor measured. Size estimates were made with the assistance of a ruler attached to the end of a meter stick. Species density (abundance per 177 m^2 census area) and mean size data were later averaged between censuses at each site.

After completing reef fish censuses, divers estimated the percent cover of all habitats within their survey area. Habitats were grouped into six categories: living coral, rock, unconsolidated rubble, gorgonians (and seafans), sponge, and sand. All other rare habitats were lumped into an "other" category. Percent cover of each habitat was later averaged between surveys at each site.

Quantitative fish censuses were supplemented by qualitative fish censuses using a roving diver technique (Schmitt et al. 1998). Supplementary censuses were conducted to note the presence of relatively rare species, or those whose appearance and/or behavior made them difficult to detect in stationary censuses. Roving censuses also increased the area surveyed for potential spawning aggregations.

Spawning and Larval Transport

Because a mutton snapper spawning aggregation has been documented on Riley's Hump, and other snapper species are expected to spawn there (Domeier and Colin 1997, Lindeman et al. in press), we were particularly interested in identifying sites where spawning aggregations occur, and examining characteristics of the site and fish at the aggregation. Potential spawning aggregation sites were identified based on information provided by commercial fishermen who have fished the area for over 20 years. For several days around the late May, 1999 full moon (expected to be the time of peak spawning for mutton snapper, Domeier and Colin 1997) divers conducted roving surveys of potential sites, specifically to determine if snapper were present at unusually high abundance. At potential spawning sites, several snapper were caught using hand line and spear to examine their gonads and determine whether or not spawning was likely to occur at this time.

The potential transport of larvae spawned on Riley's Hump during the May, 1999 full moon was examined using satellite-tracked drifter buoys (Argo-drifter, Technocean Inc.). Shortly after dark on the evening of May 30, 1999, the night of the full moon we released two satellite-tracked drifter buoys (Argo-drifter, Technocean Inc) at potential spawning sites on Riley's Hump. Drifters floated just below the sea surface, with only their antennae exposed to the air. Drifter positions were determined several times each day by Argos satellites until the drifters were retrieved or their batteries died. Drifters were retrieved after 38 days to encompass the expected mutton snapper planktonic larval duration of three to five weeks (Clarke et al. 1997, Watanabe et al. 1999, Lindeman et al. in press).

RESULTS

Reef Fish and Habitat

Fish and habitat censuses ranged in depth from 26 to 33 m and covered over 7,065 m^2 of the seafloor and the water column above it. In general, the areas censused were low relief areas of carbonate rock or sand, with varying amounts of rubble (primarily small pieces of dead coral), small coral heads (1 - 2 m diameter mounding coral heads), sponges, gorgonians or seafans. Rock and rubble were frequently covered with low growing macroalgae. Overall, rock and sand were the dominant habitats (Figure 2), covering over 65% of the area censused on average, but coral, rubble, sponges and gorgonians had high percent cover (up to 50%) at some sites.

During quantitative stationary censuses, 8,654 fish ($\bar{x}$ = 199.3 ± 69.7 per census) belonging to 97 species (30 families) were observed. A mean of 25.8 (± 5.3) species were observed at each site (17.1 ± 4.6 per individual census). The most frequently observed species were: bicolor damselfish, *Pomacentrus partitus*; bluehead wrasse, *Thalassoma bifasciatum*; yellowhead wrasse *Halichoeres garnoti*; gray snapper, *Lutjanus griseus*; saucereye porgy, *Calamus calamus*; and blue tang, *Acanthurus coeruleus*. A total of 11 families occurred in more than 50% of observations with damselfish (pomacentridae), snappers (lutjanidae), wrasses (labridae), and jawfish (opistognathidae) having the highest density (Figure 3). The high mean jawfish density was primarily due to an extremely high average density (191.7/census) of yellowhead jawfish, *Opistognathus aurifrons*, at a single site. Several infrequently sighted families also had high mean densities. For example, goatfish (mullidae) had the highest mean densities of any family ($\bar{x}$=64.7 ± 269 fish/census), but were observed at only 10.5% of study sites. High goatfish density resulted from a single site containing a large school of yellow goatfish, *Mulloidichthys martinicus*, (average site density: 1,175 fish/census).

During qualitative fish censuses, including fish observed after five minutes in stationary censuses, a number of species and families were recorded that were not observed in quantitative censuses. For example, sargassum triggerfish, *Xanthichthys ringens*, were not detected in quantitative censuses, but were locally abundant in qualitative censuses at one of the study sites. Similarly, 11 different jewfish, *Epinephelus itajara*, were observed and several tagged (up to 4 on a single dive) near a single large coral head over three days, but were not observed in quantitative censuses and rarely observed in qualitative censuses at other sites.

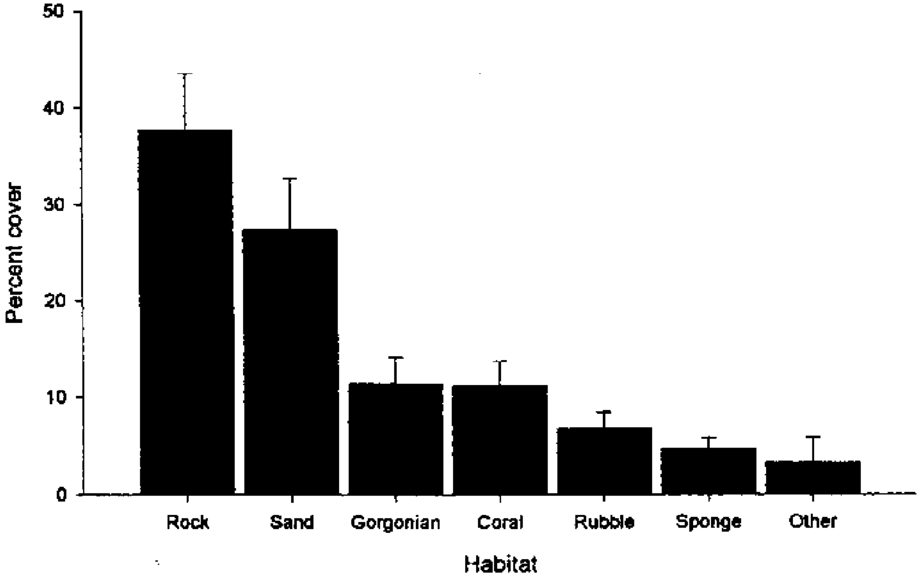


Figure 2. Percent cover of habitat categories in visual censuses on Riley's Hump (n = 19). Means represent site averages and error bars show standard deviation.

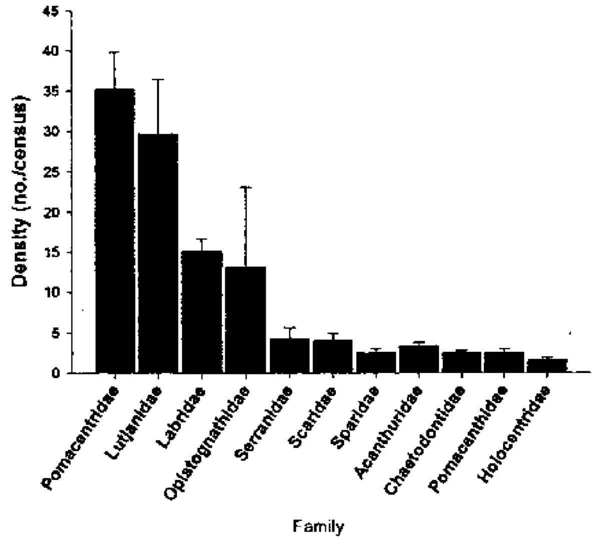


Figure 3. Density of fish (mean number observed per 177 m^2 census) belonging to all families observed at over 50% of study sites on Riley's Hump (n = 19). Means represent average site densities and error bars represent standard deviation.

Spawning and Larval Transport

At one site suggested by commercial fishermen as a potential spawning site, divers observed high abundance of both gray and mutton snapper. The site was near the edge of the hardbottom and coral reef area, where a relatively flat bottom surrounded a few large coral heads (>2 m diameter) that provided structure. On one late afternoon dive, >100 mutton snapper were observed congregating at the interface between the reef and sand plain. Nearby, schools of several hundred gray snapper were observed over the reef on several dives. Preliminary analysis of the gonads from several female mutton snapper caught at the site indicates that they contained mature eggs (M. Burton, National Ocean Service, Beaufort, NC, personal communication). The reproductive state of gray snapper was not examined, but several male gray snapper caught by handline released milt before being returned to the water.

One of the two satellite-tracked drifter buoys released at the time and location of expected spawning by mutton snapper, lost contact with the satellite within 24 hours of release due to low battery power. Twenty-one days after its release, however, a research vessel spotted this drifter offshore from Pompano Beach, FL (26° 12.385' N, 80° 04.165' W; Figure 4). The other drifter maintained contact with the satellite until it was retrieved on July 7, after drifting for 38 days. Upon release, this drifter began a slow drift to the west, followed by a gradual turn to the south. After 12 days, it was entrained in the Florida Current and began moving rapidly to the northeast. Three weeks after its release, the drifter entered Biscayne Bay, and remained in shallow waters of the Biscayne National Park until it was retrieved.

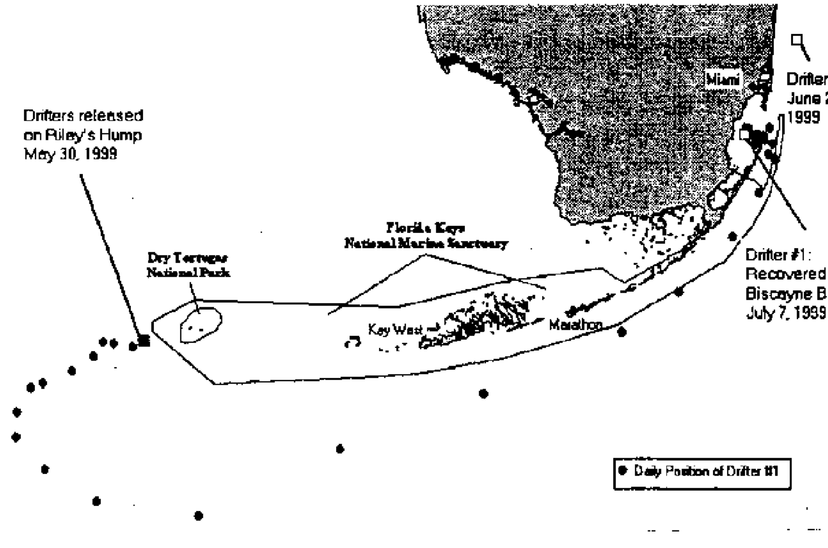


Figure 4. Daily position of a satellite-tracked drifter buoy deployed at the time and location of expected snapper spawning on May 30, 1999. The position of a second buoy, which immediately lost contact with the satellite but was later re-sighted, is also shown.

DISCUSSION

Quantitative assessment of habitat and living resources within a proposed reserve area is essential to determine potential benefits of reserve protection, and to establish a baseline for evaluating reserve effects. Despite its apparent importance to fishermen (Lindeman et al. in press), and seasonal closure, our study was the first to quantitatively assess the habitat and reef fish community of Riley's Hump. Our study also identified potential spawning aggregations and larval dispersal pathways from this area.

Riley's Hump can generally be characterized as low relief rocky reef mixed with pockets of sandy areas. Rock was often overgrown with corals, gorgonians, and sponges, which provided some structural complexity, but corals and rock ledges with vertical relief greater than two meters were rare. Thus, Riley's Hump is a different type of reef from others in the proposed Tortugas Ecological Reserve. The Tortugas North reserve contains many high relief (>10 m) coral pinnacles and reef with high coral cover and a variety of shallower reefs in the Dry Tortugas National Park (Rydene and Kimmel 1995, Authors' personal observations).

Despite the lack of vertical relief, low habitat complexity, and low coral cover, Riley's Hump had a relatively high diversity of reef fish, with 117 species belonging to 32 families observed overall. Although our visual censuses of Riley's Hump are likely to have missed many small, cryptic and rare species, our estimate of species richness for Riley's Hump is comparable to several other coral reef areas within the Florida Keys, where similar visual techniques were used (Jones and Thompson 1978, review by Chiappone and Sluka 1996). The mean number of species observed per individual census on Riley's Hump was comparable to reefs within Looe Key National Marine Sanctuary (now part of the FKNMS) and Biscayne National Park, where similar surveys were conducted (Bohnsack et al. 1987, 1992). Species-area curves derived from similar sampling techniques within Dry Tortugas National Park indicate that species diversity there is also comparable to our findings at Riley's Hump (Rydene and Kimmel 1995). Fish density at Riley's Hump was also similar to that reported from Biscayne National Park and most reef habitats within Looe Key National Marine Sanctuary (Bohnsack et al. 1987, 1992).

Despite similarities in fish density and species richness between Riley's Hump and other reefs, reef fish community composition differed between Riley's Hump and other reefs in the Florida Keys. For example, other studies using visual censuses reported that grunts (haemulidae) comprised between 15 to 20%, and snappers less than 5%, of fish on reefs throughout the Florida Keys (Bohnsack et al. 1987, 1992, Thompson et al. 1990, Rydene and Kimmel 1995). On Riley's Hump, however, less than 5% of all fish observed were grunts, and snappers comprised 15% of all fish. Qualitative censuses on Riley's Hump also included fish species, such as the sargassum triggerfish, *Xanthichthys ringens*, not reported in visual censuses from other parts of the Florida Keys (review by Chiappone and Sluka 1996). Differences between the reef fish community Riley's Hump and other parts of the Florida Keys are most likely due to several factors, including differences in depth, habitat, proximity to shallow nursery areas, and fishing pressure. Moreover, density of some species, such as gray snapper, *Lutjanus griseus*, may have been high at Riley's Hump during our study because they aggregate there to spawn (Lindeman et al. in press).

Although our study provided no direct evidence that spawning occurred, high abundance of mutton and gray snapper, and individuals in advanced reproductive states provided indirect evidence of spawning. If this was the case, the spawning aggregation that we encountered was much smaller than those reported for mutton snapper in elsewhere and on Riley's Hump in previous years (Domeier and Colin 1997). The reason for the small size of this aggregation is unknown. Reports from fishermen indicate that mutton snapper may not have formed typically large spawning aggregations during this full moon throughout the Florida Keys (P. Gladding, Key West commercial fisherman, personal communication). Alternatively, fishing pressure, particularly following the seasonal closure (Dahlgren, personal observation) may have reduced the abundance of mutton snapper on Riley's Hump.

Currents at the time and place of expected spawning until the time of expected settlement, suggest that larvae spawned on Riley's Hump during the late May-early June, 1999 full moon may have been transported throughout Florida Keys, Biscayne Bay, and along the east coast of Florida to Pompano Beach. Larval transport at this time appears to have been influenced by the Tortugas Gyre initially, then the Florida Current. A companion study, in which neutrally buoyant bottles were released at the same time and place as our drifters, produced similar results, with bottle recoveries ranging from the upper Florida Keys to Palm Beach 17 - 35 days after their release (M. Domeier, Pfleger Institute for Environmental Research, personal communication).

Using current patterns to assess larval transport, however, assumes that larvae are transported passively. In reality, larvae may be capable of swimming, and may exhibit behavioral patterns that limit dispersal distances of fish larvae (Sale and Cowen 1998). Thus, our results should be interpreted as a potential range of recruitment. Larval transport may also vary within and between years due to spatiotemporal variability in regional current patterns and local oceanographic or meteorological features (e.g., Shenker et al. 1993). Nevertheless, the regular occurrence of the Tortugas gyre, Pourtales gyre, and other features suggest that larvae spawned at Riley's Hump are likely to be retained in the Florida Keys and South Florida system frequently (Lee et al. 1994, Lee and Williams 1999).

Our study provides before-protection information on reef fish and habitats of Riley's Hump that is essential for the evaluation of reserve efficacy in the future. The results of our study also suggest that the protection of Riley's Hump within the proposed Tortugas South reserve may serve several purposes. Because the habitats and reef fish community of Riley's Hump differ from other reef areas proposed for protection in both the Tortugas North reserve, and deepwater areas of the Tortugas South reserve, protecting Riley's Hump is important for the conservation of biodiversity. The relatively high abundance of overexploited commercial and recreational fish species (e.g., snappers) on Riley's Hump may also contribute towards rebuilding these overfished stocks, or at least help to safeguard them against the possibility of management failure (Ault et al. 1998, Dahlgren and Sobel in press). The potential to provide fisheries benefits is further supported by direct and indirect evidence that the area is important for snapper spawning, and current patterns may transport larvae spawned there throughout the Florida Keys and south Florida. Thus, Riley's Hump may serve as an important source population for many of south Florida's overexploited fish stocks.

ACKNOWLEDGEMENTS

Quantitative and qualitative censuses of reef fish and habitat were conducted with the assistance of M. Burton, G. Davenport, D. DeMaria, and K. Marks and M. Domeier. Additional diving and photo/video support was provided by S. Brown, D. Kincaid, P. Molyneaux, and R. Buckley. The crew of the *M/V Spree* provided logistical support. P. Gladding provided invaluable knowledge and fishing support. T. Lee and K. Lindeman provided drifter buoys, and E. Williams assisted with tracking drifters. This research was funded by the Center for Marine Conservation and a Challenge Cost-share grant from the U.S. Fish and Wildlife Service awarded to the Key West National Wildlife Refuge.

LITERATURE CITED

Ault, J.S., J.A. Bohnsack, and G.A. Meester. 1998. A retrospective (1979-1996) multispecies assessment of coral reef fish stocks in the Florida Keys. *Fish. Bull.* **96**:395-414.

Bohnsack, J.A. 1998. Application of marine reserves to reef fisheries management. *Aust. J. Ecol.* **23**:298-304.

Bohnsack, J.A., D.E. Harper, D.B. McClellan, M.W. Hulsbeck, T.N. Rutledge, M.H. Pickett, and A.M. Eklund. [1992] Quantitative visual assessment of fish community structure in Biscayne National Park. DOC/NOAA/NMFS/SEFSC/Miami lab. Unpubl. MS. 45 pp.

Bohnsack, J.A., D.E. Harper, D.B. McClellan, D.L. Sutherland, and M.W. White. [1987] Resource Survey of Fishes within Looe Key National Marine Sanctuary. NOAA Tech. Mem. NOS MEMD 5. 108 pp.

Bohnsack, J.A. and S.P. Bannerot. [1986]. A stationary visual census technique for quantitatively assessing community structure of coral reef fishes. NOAA Tech. Rep. NMFS 41. 15 pp.

Chiappone, M, and R. Sluka. 1996. *Site characterization for the Florida Keys National Marine Sanctuary and environs volume 6: Fishes and fisheries.* The Nature Conservancy. Miami, FL. 149 pp.

Clarke, M.E., M.L. Domeier, and W.A. LaRoche. 1997. Development of larvae and juveniles of the mutton snapper (*Lutjanus analis*), lane snapper (*Lutjanus synagris*) and yellowtail snapper (*Lutjanus chrysurus*). *Bull. Mar. Sci.* **61**:511-537.

Dahlgren, C.P. 1999. Replenishing marine populations. *Science* **284**:49.

Dahlgren, C.P. and J. Sobel. Designing a Dry Tortugas Ecological Reserve: how big is big enough...to do what? *Bull. Mar. Sci.* In press.

Domeier, M.L. and P.L. Colin. 1997. Tropical reef fish spawning aggregations: defined and reviewed. *Bull. Mar. Sci.* **60**:698-726.

Fratantoni, P.S., T.N. Lee, G.P. Podesta, and F. Muller-Karger. 1998. The influence of Loop Current perturbations on the formation and evolution of Tortugas eddies in the southern Straights of Florida. *J. Geophys. Res.* **103**:24,759-24,779.

Lee, T.N., M.E. Clarke, E. Williams, A.F. Szmant, and T. Berger. 1994. Evolution of the Tortugas gyre and its influence on recruitment in the Florida Keys. *Bull. Mar. Sci.* **54**:621-646.

Lee, T.N. and E. Williams. 1999. Mean distribution and seasonal variability of coastal currents and temperature in the Florida Keys with implications for larval recruitment. *Bull. Mar. Sci.* **64**:35-56.

Lindeman, K.C., R. Pugliese, G.T. Waugh, and J.S. Ault. Developmental pathways within a multispecies reef fishery: management applications for essential habitats and reserves. *Bull. Mar. Sci.* In press.

Rydene, D.A. and J.J. Kimmel. [1995] A five year assessment (1990-1994) of coral reef fish assemblages within Dry Tortugas National Park, Florida using visual census techniques. Florida Marine Research Institute. 81 p. Unpubl. MS.

Shenker, J.M., E.D. Maddox, E. Wishinski, A Pearl, S.R. Thorrold, and N. Smith. 1993. Onshore transport of settlement stage Nassau grouper, Epinephelus striatus, and other fishes in Exuma Sound, Bahamas. *Mar. Ecol. Prog. Ser.* **98**:31-43.

Sobel, J.A. 1996. Marine reserves: necessary tools for biodiversity conservation? *Global Biodiversity* **6**(1): 8-18.

Thompson, T.E., D.G. Lindquist, I.E. Clavijo, S.K. Bolden, S.W. Burk, and N.C. Drayton. 1990. Assessment of reef fishes at Sombrero Key, Florida. Pages 375-383 in: W.C. Jaap (ed.). *Diving for Science...1990.* Proceedings of the American Academy of Underwater Sciences, 10th Annual Scientific Diving Symposium, October 4-7, St. Petersburg, FL.

Watanabe, W.O., E.P. Ellis, S.C. Ellis, J. Chaves, and C. Manfredi. 1998. Artificial propogation of mutton snapper, Lutjanus analis, a new candidate marine fish species for aquaculture. *J. World. Aquacult. Soc.* **29**:176-187.

The Origin of Florida Fish and Fisheries

R. GRANT GILMORE, JR.
Senior Aquatic Ecologist
Dynamac Corporation
Mail Code: DYN-2
Kennedy Space Center, FL 32899 USA

ABSTRACT

Today the Florida peninsula provides habitat for one of the most complex ichthyofaunas in the western Atlantic. High Floridian biodiversity and habitat complexity is principally due to Florida's geomorphology resulting in transitional climates associated with regional oceanographic currents which vary significantly around the peninsula. The majority of Floridian fish species are tropical, having migrated to the peninsula only recently, during the Holocene submergence of the Florida platform. These tropical invaders now form a heterogeneous group that often numerically dominate fish communities in southeastern Florida. Many tropical species form the basis for major recreational and commercial fisheries. Most fishery species life histories require estuarine, coastal or shelf edge spawning often with passive/active movement of pelagic eggs and larvae within estuaries, or across continental shelves and into, or from oceanic waters. Entrainment within oceanic currents allows larval transport for considerable distances downstream, loss of larvae from Florida waters, larval recruitment from Caribbean Provincial waters to Florida, or recruitment from Florida back to Floridian waters via eddies, gyres and meanders.

The differential distribution of suitable habitats and hydrological/climatic environments around the Florida peninsula creates differential faunal associations with the Caribbean Province, consequently, successful fishery larval recruitment from Caribbean. The Floridian marine and coastal environment can be divided into 26 faunal - biogeographic regions which support distinct communities of warm temperate, eurythermic and stenothermic tropical faunas. Using the speciose family serranidae as an example, the origination of various species from various regions of the western Atlantic can by hypothesized. These analyses give additional perspective on the evolution of the Florida fish fauna and its relationship with the Caribbean Sea. The differential association of Florida fishery species with the Caribbean Sea has major implications for regional and local fishery management and critical fishery habitat assessment.

KEY WORDS: Florida fisheries, ichthyofauna, biogeography

INTRODUCTION

The Florida peninsula provides habitat for one of the most complex and richest ichthyofaunas in the western Atlantic principally due to its geomorphology and transitional climate associated with regional oceanographic currents (Gilmore 1995). There are many historical accounts on the faunal composition of the tropical western Atlantic. Few new fish species are presently being described. Except for a few very rich families (ex. Gobiidae) most regional shallow water (< 200 m depths) species have been described and are well known. This allows the majority of the Floridian ichthyofauna to be adequately described and enumerated. However, considerable disagreement exists in the distribution and ecology of this fauna as seen in recent publications on the distribution of major fisheries groups such as the groupers and snappers.

In the past the normal historical procedure for describing the distribution of coastal marine fish species was to give their range in association with their collection site. However, typical marine fish life history strategies along with the mobility of water allows species to be distributed for hundreds and often thousands of miles beyond the range of their typical breeding habitats and ecosystems to which they have adapted. The dots on the range maps are misleading if they do not take into account species preferred habitat and breeding range. Capture records actually mean little to the life history of the animal. The often stated range for tropical species, "from Nova Scotia to Rio de Janeiro" can give a very erroneous picture of organismic biology, biogeography and critical habitat. This then creates gross errors in fishery species distribution or assessment of critical habitat. There are examples today of fisheries literature and public information sites (World Wide Web network) placing species in entire ocean basins (Gulf of Mexico or Caribbean Sea) where they are considered extremely rare or absent. World wide coverage of regional fisheries through FAO volumes on groupers (Heemstra and Randall 1993) and snappers (Allen,\ 1985) show false species range information in the tropical western Atlantic. An important fishery species, the mutton snapper, pargo criollo, *Lutjanus analis*, is listed as common in the Gulf of Mexico. The northern and eastern Gulf has been listed as critical habitat for this species by the NOAA Critical Fisheries Habitat program. Unfortunately, the mutton snapper does not occur in this region of the Gulf of Mexico. Other common valuable commercial and recreational fishes for which equally erroneous distributional information has been given are the striped mullet, ***Mugil cephalus*** (absent from the Caribbean, but commonly listed as occurring there, Thomson, 1963 Campton and Mahmoudi 1991, Pattillo et al. 1997) and black grouper, ***Mycteroperca bonaci***, as common in the northern and eastern Gulf of Mexico but absent there. These species are the most conspicuous examples revealing erroneous distribution presentations in the literature.

Due to the inaccuracies in describing fish species distribution an effort is made here to describe basic fish biogeographic distribution patterns for the Floridian ichthyofauna based on species evolutionary history, eco-physiologies, comparative ecology, and life histories. This treatment attempts to take these biological and environmental factors into account, particularly sensitivities to hypothermia. However, the only comprehensive data is from repetitive quantitative fish collections and observations made by many investigators over the past fifty years. Phyletic relationships and sibling species associations are also important in determining the evolutionary history and distribution of fishes in the tropical western Atlantic. This analysis is done relative to oceanic as well as coastal environments which play a vital role in marine species survival and distribution. Data from quantitative ecological studies through the Gulf and Caribbean region are used for these analyses as well as information on hypothermal mortalities recorded from the Florida peninsula.

The central hypothesis that governs these analyses is that the Florida ichthyofauna consists of definable species groups or faunas, each fauna having similar biological, ecological and eco-physiological adaptations. Some species may have co-evolved showing similar distribution tracks while others reveal different evolutionary histories causing them to occupy different oceanographic, hydrological and climatic regions around the Florida peninsula.

There are a variety of important fishery questions that can be answered by a detailed study of regional biogeography. The basic objectives relevant to regional fisheries are to determine:

i) Which fishery species are in Florida?
ii) Why other species are not in Florida?
iii) How did the species that are in Florida, arrive in Florida?
iv) What allows these species to survive in Florida?

Methodologies entail a thorough review of coastal and continental shelf studies within the region as well as a thorough review of individual species ecologies from published and unpublished sources. In this presentation the range of the species is limited to its known breeding range, or where adults and juveniles predictably and commonly occur on at least an annual basis.

MIGRATORY STRATEGIES AND TRANSPORT MECHANISMS: INVASION OF THE FLORIDA PENINSULA

As the Holocene post-glacial sea level rose and warm equatorial current flows changed so did regional and local climates, sea surface temperatures and the size of tropical aquatic habitats in the western Atlantic. Tropial habitats were not limited to the open ocean but included complex coastal ecosystems. As tropical coastal ecosystems developed in Florida fishes were able to reach them by a wide variety of mechanisms.

Invasion Mechanisms

It is now well documented that besides actively swimming, often for substantial distances, fishes also passively utilize ocean currents, eddies and gyres, prevailing winds, storms, freshwater runoff and tides for short and long distance transport as well as local larval retention. Passive propagule, egg and larval movement is pervasive in the ocean and allows larvae to travel tens to thousands of miles with minimal energy expenditures (serranids, gobioids). Other species (ex. spotted seatrout, *Cynoscion nebulosus*) time their spawning activities at specific locations to insure local larval recruitment and retention to parental feeding grounds. Florida resides at the downstream juncture of two major oceanic currents: Caribbean Current via the Yucatan Straits and Gulf of Mexico (as the Loop Current) and the Antilles Current (via the Bahama - Cuba Channel). They combine within the Florida Straits to become the Florida Current passing between Florida, Cuba and the Bahama Platform (Figure 1). All originate in the equatorial North Atlantic before flowing into the Caribbean Sea (Caribbean and Guinea Currents) or along the northern edge of the Antilles and south of the Bahama Islands (North Equatorial and Antilles Currents). The prevailing southeasterly trade winds follow these currents and so typically, do hurricanes and tropical storms. The immense freshwater flows from the Amazon and the Orinoco dominate riverine input to the region impacting mostly the Antilles, but potentially giving propagules of euryhaline tropical peripheral species to the Florida peninsula. Prevailing easterly winds and continental shelf gyres on the west Florida shelf block drifter movement to inshore waters of the western side of the Florida peninsula. Consequently, Caribbean larval transport, to western peninsula Florida does not often occur (Gilmore and Hastings 1983, Rezak et al. 1985). Successful invasion also requires specific biotic and physical conditions as well as adequate habitat for survival at the settlement site. Seasonal hypothermal conditions on the west Florida shelf preclude survival of stenothermic tropical species. During the winter (January - February) shelf waters in the eastern Gulf of Mexico are considerably colder relative to the Florida east coast while the latter locale is under the influence of the warm Florida Current (Gilmore et al. 1978).

Eddies moving off of major coastal currents may carry larvae and juveniles onto adjacent continental shelves where cross shelf transport via Ekman phenomena, upwelling events, and prevailing winds as well as tides bring them onto near shore reef formations or into estuarine waters. Coastal counter-currents have been found to transport larvae from major ocean currents (Florida Current or Gulf Stream; Lee and Williams 1999). Fish larvae and juveniles are transported routinely enough from Caribbean sources to Florida that little genetic variation is seen in a variety of reef fishes known to have long duration 930+ days) pelagic larvae (serranidae and carangidae) examined from Florida and the Caribbean to date (Richardson and Gold 1993, Gold and Richardson 1998). Genetic exchange via passive larval transport may be often enough to reduce local gene pool isolation.

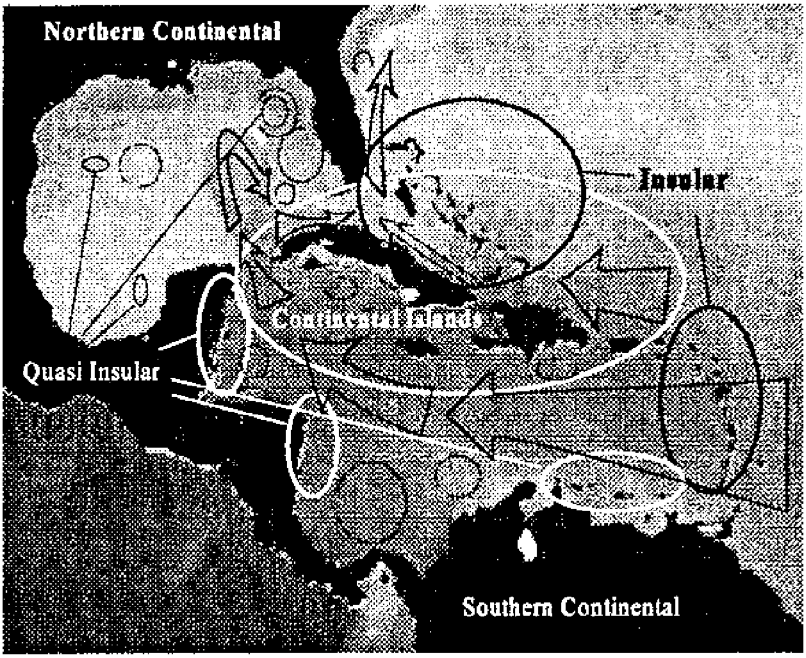


Figure 1. Geographic terrains in the tropical western Atlantic: continents, continental islands, quasi-insular and insular terrains; major ocean currents (arrows), gyres and eddies (dark open broken circles). Dark background water color indicates warmer water (from January 1998 image).

There are over 1,200 fish species recorded from the Florida peninsula. They are too numerous to list in this publication, is one of the most speciose families and is analyzed here as an example. The family serranidae is a basal perciform group which supports major fisheries throughout the region yet contains subfamilies of fishes that have great biogeographic, faunal, ecological and evolutionary significance. An analysis of serranid life histories and distributions is aided substantially by the fact that they are economically important and therefore sought after by fisheries throughout the region. They are of great conservation interest because of their depleted stocks due to overfishing and their popularity to the diving tourist. Serranids offer a wide variety of reproductive patterns and life history strategies. Serranid reproduction includes gonochorism and all varieties of hermaphrodism. Serranid ecological roles vary from small benthic predators to schooling planktivores and large roving predators. Study of this familial diversity presents further insight into regional fish community evolution and distribution.

Dispersal - Migratory Strategies and Faunal Movement Patterns

Major differences between marine and terrestrial, freshwater and estuarine environments with regard to organism isolation, migration and evolution is the potential for broad passive distribution. Many marine and euryhaline freshwater species utilize the mobility of water to complete life histories. The elopomorph fishes, the tarpon (sabalo), *Megalops atlantica*, and the American freshwater eels, *A. rostrata*, represent clear examples of passive oceanic mobility used predictably for coastal recruitment of larvae hatching in oceanic environments and recruiting as juveniles to adult coastal and freshwater habitats. Similar strategies are used by a large number of marine fishes. These species must find critical habitats on a broader geographic scale. This also means that marine species that spend early life histories in the open ocean are unlikely to be genetically isolated. When isolation does occur, it is typically associated with geographic isolation of ocean basins, the North Atlantic versus the South Atlantic or Eastern Pacific.

Limitations in survival will be principally associated with juvenile settlement site characteristics and adult spawning site characteristics. Pelagic environments would certainly influence larval survival, however, they are relatively stable when compared to coastal environments.

Those species which can guarantee adequate juvenile and adult environments downstream will be most likely to survive, and this may require adult migration to the most appropriate spawning location. Active adult migration to upstream spawning sites may allow passive larval migration back to parental habitats downstream. Indian River lagoon snook, *Centropomus undecimalis*, use this strategy for coastal larval recruitment to sites between 27° 00' and 28° 00' N. latitude by migrating to nearshore continental shelf spawning sites south of 27° 00' to 26° 00' N. latitude. In contrast, broadcast spawning into oceanic currents over continental shelf margins insures larval dispersal to potential new habitats on a broader regional scale. Gag grouper, *Mycteroperca microlepis*, spawn at a shelf edge (70-100 m depth) Oculina coral reef near 27° 30' N. latitude. Their larvae may remain in the Florida Current water mass along the shelf edge where they are likely to be entrained in eddies and wind driven shelf waters which move across the shelf to inshore settlement sites and nursery grounds. Tagged adult gag groupers from South Carolina have been recaptured off central Florida revealing a Carolina gag migration south to spawn off Florida. This would insure recruitment of larvae via north-northeasterly Florida Current - Gulf Stream flow (Van Sant et al. 1994, McGovern et al. 2001). Serranid larvae are morphologically adapted for pelagic dispersal. Therefore, larval entrainment in coastal pelagic currents is likely an evolutionary strategy for dispersal and survival in epinepheline serranids.

If these dispersal/migration strategies work, then many marine species should use them - and they do. Notable fishery species groups that use this strategy are the tarpons, or sabalos, megalopidae; snook or robalos, centropomids; groupers, serranidae; snappers, lutjanidae; porgies, sparidae; and jacks, carangidae. The most abundant pelagic open ocean larvae of coastal or continental margin fishes in the tropical western Atlantic are serranids, particularly anthiine serranids, gobioid fishes, carangids, clupeids and syngnathids (Richards et al. 1993). If all these families are casting larvae to ocean currents why are species within these families often limited in distribution? The most likely explanation is that juveniles and adults of these widely dispersed larvae are often stenoecious, requiring narrowly defined ecological and environmental conditions for survival. They have evolved under a variety of environmental and biological conditions associated with past climatic, hydrological and geological conditions in addition to biological interactions at population and community levels (trophic, spatial and spawning competition, and predation mortality). Many species showing the same distribution have apparently co-evolved, consistently occuring in the same habitat and represent the same fish community type.

Species Eco-physiology, Thermal Preferences and Biogeographic Distribution

Survival after Arrival: Historical work (Gilmore 1977, Gilmore et al. 1978, Gilmore and Hastings 1982, Gilmore 1985, 1995) has classified the Florida fish fauna into a variety of groups based on preferred temperature ranges and habitat preferences. Unfortunately, thermal preferences of many species have not been examined empirically through physiological experiments. However, many field observations have been made of thermal mortalities around the Florida peninsula over the past century (Gilmore et al. 1978, Snelson 1978). These observations have revealed differential temperature tolerances and lethal minimums for many tropical western Atlantic species. This allows a definition of thermal tolerance based on survival temperatures during natural hypothermal events. These thermal responses coupled with known abundance in preferred habitats and broad geographic distribution allow species to be classified as stenothermic tropical, eurythermic tropical, warm temperate and temperate.

Florida-tropical western Atlantic species can also be classified as principally "insular" or "continental" in provincial or sub-regional habitat preference. Regional insular species are recorded from the Lesser Antilles, Bahama Islands and Bermuda. "Quasi- insular" species are recorded from islands adjacent to continental margins (islands off the coast of Venezuela and Belize) and from continental islands (the Greater Antilles)(Fig. 2). Continental species occur primarily in continental habitats and only occur in insular ecosystems where adequate continental habitat types occur. Many continental species are found in the Greater Antilles.

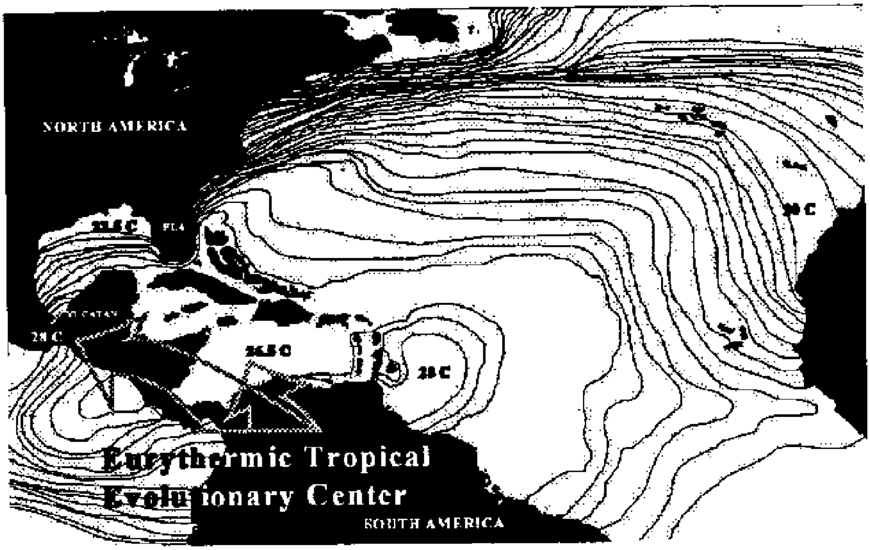


Figure 2. Glacial topography and hypothesized temperature patterns showing mean summer surface temperatures in the North Atlantic.

The preferred definitions for the fish biogeographic guilds used here are: *Stenothermic Tropical* — Species that cannot withstand prolonged exposure (> 48 hours) to ambient water temperatures below 13 - 14°C without significant mortality at polyhaline/marine salinities, 20 - 35‰. They prefer water temperatures above 18 - 20°C. These species are more typical of tropical insular environments which typically do not reveal significant water temperature or salinity variation. These species are typically distributed to the margins of the tropical/subtropical climatic zones (25° 00 to28° 00 N. lat.) only in association with warm coastal currents. This is the typical West Indian or Antillean Provincial fauna (Province = 10% endemism, Briggs 1974). These species were probably limited to the windward Antilles and the South American coast southeast of the Orinoco River during glacial periods. Epinepheline serranid examples are: Nassau grouper, *Epinephelus striatus*; black grouper, *Mycteroperca bonaci*; Comb grouper, *M. rubra*, and Tiger grouper, *M. tigris*.

Table 1. Eco-physiology and habitat distribution in Serranidae. (#) = Total Western Atlantic. species.

	Eurythermic (47 Sp)				
	Euryoecious	Continental	Insular	Total	
Epinephelinae					
Epinephelini	2	8	5	15	-24
Liopropomini	1	2	0	3	(6)
Grammistini	1	1	0	2	(8)
Anthiinae	1	9	1	11	(11)
Serraninae	6	10	0	16	(35)
Totals	11	30	6	47	(84)
	Stenothermic (37 Sp)				
	Euryoecious	Continental	Insular	Total	
Epinephelinae					
Epinephelini	0	6	3	9	(24)
Liopropomini	0	3	0	3	(6)
Grammistini	2	3	1	6	(8)
Anthiinae	0	0	0	0	(11)
Serraninae	0	5	14	19	(35)
Totals	2	17	18	37	-84

Eurythermic Tropical — Species that can withstand ambient water temperatures to 6-8°C for 24 - 48 hrs,without significant mortality at polyhaline/marine salinities, 20-35‰. They prefer water temperatures above 14°C. These species are more often continental in their range and have evolved the physiological capability to withstand greater temperature and salinity variation than stenothermic insular species. They may even do well at locations which have seasonally warm/temperate to temperate climates as far north as 35° 00' N. lat. They also penetrate depths greater than 100 m, survive annual upper slope cold upwellings (6 - 8°C), and may penetrate the epibathyal region along island and continental escarpments at least to 700 m depths. This is the typical Caribbean or Central American Provincial fauna (Briggs 1974), divided by Miller (1969) up into the "Southern Cool Tropical Subregion" found in inshore waters along the coasts of northern South and Central America and the "Northern Cool Tropical Subregion" found along the southern Gulf of Mexico and southeastern Florida to 28° 00 N. However, we extend this fauna further north into continental shelf environments along the Atlantic coast to Cape Hatteras (35° 00 N) and on the continental shelves of the northern Gulf of Mexico. It is likely that this fauna originated during glacial periods in the southern Gulf of Mexico off Laguna de Terminos, in southern Cuba, the southeastern Bahamas and the Caribbean Sea where ambient water temperature means were 5°C below today's mean sea surface temperatures (Emiliani 1971, 1972).

Epinepheline serranid examples are: Red grouper, *Epinephelus morio*; Scamp grouper, *Mycteroperca phenax;* Snowy grouper, *E. niveatus..*

Warm-Temperate — Species which readily withstand ambient water temperatures below 6°C for prolonged periods (several days) without significant mortality. They prefer water temperatures below 35°C. These species are primarily continental in habitat preference and do not typically occur commonly in tropical geographic regions. However, they also do not prefer temperate ecosystems (north of 37° 00' N. latitude) and prolonged (multi-seasonal) significantly lower ambient water temperatures < 14°C. These species may have a phyletic relationship with tropical families, but are usually differentiated at the generic level. This is the typical Carolinian Provincial Fauna. Epinepheline serranid examples: Speckled hind, *Epinephelus drummondhayi*, Gag, *Mycteroperca microlepis, Black seabass, Centropristis striata..*

Temperate — Species which have a decidedly glacial heritage and withstand ambient water temperatures below 10°C. for months at a time. These are continental species which are often euryhaline but do not do well at ambient water temperatures above 30°C. Though these species may have an ancient phyletic association with tropical fish families they are often differentiated greatly having evolved in cold ambient water ecosystems. This is the typical Virginian and Acadian (Cold - Temperate) Provincial Fauna. Epinepheline serranid examples: None

Sibling species have formed from vicariant events associated with climatic change, glacial and inter-glacial periods and geological formations, isolation associated with greatly differing habitats, such as insular versus continental ecosystems. These species pairs are found in all subfamilies and tribes within serranidae. Apparent sibling (geminate) species examples are the red and Nassau groupers (*E. morio/E. striatus*), the warsaw and misty groupers (*E. nigritus/E. mystacinus*), the wrasse and lineside basses (*Liopropoma eukrines/L. abberans*), the whitespotted and greater soapfishes (*Rypticus maculatus/R. saponaceous*), the swallowtail and nickol's basslets (*Anthias woodsi/A.nicholsi*), the belted sandfish and serrano de arena (*Serranus subligarius/S. flaviventris*).

Table 2. Habitat and sibling species pairs (geminate species) evolution in Serranidae

	Euryoecious	Continental	Insular	Siblings
Epinephelinae				
Epinephelini	2	11	11	6
Liopropomini	1	2	3	2
Grammistini	3	2	3	2
Anthiinae	1	9	1	1
Serraninae	6	15	14	7+
Totals	13	39	32	

PALEONTOLOGY OF THE FLORIDA PLATFORM AND THE EVOLUTION OF ITS ASSOCIATED ICHTHYOFAUNA

The geological boundaries of the Florida platform are considerably larger than the terrain that is presently above sea level. During the last glaciation, 72-12,000 ybp, the large (250 km wide) west Florida shelf had emerged and the Florida coastline was typified by sharp escarpments or steep slopes around most of the peninsula. Continental shelves were largely absent (Figure 2). The central plateau elevation was 100 m higher and colder. The climate of the peninsula and surrounding waters was temperate or warm-temperate. Habitat heterogeneity was considerably lower than that seen today. Consequently, the glacial Floridian fauna was more depauperate, containing many of the species associated with the Carolinian and Virginian Provinces today (Atlantic sturgeon, *Acipenser oxyrhynchus*; striped bass, *Morone saxatilis*; Atlantic shad, *Alosa sapidissima*; Atlantic menhaden, *Brevoortia tyrannus*; black seabass, *Centropristis striata*; weakfish, *Cynoscion regalis*). These species and others with similar temperate physiologies remain at isolated sites in northern Florida as glacial relicts.

The glacial ichthyofauna of the southern Gulf of Mexico was more diverse, particularly around the Laguna de Terminos. Paleoclimatological evidence reveals a warmer glacial climate west of the Yucatan peninsula than the remainder of the western Gulf of Mexico (Figure 2). This region probably acted as a thermal refugia for species preferring warmer waters during several Pleistocene glacial periods. The Gulf endemics with warm temperate affinities may have evolved here, such as the yellowfin menhaden complex, *Brevoortia gunteri/smithi*; goldspotted killifish, *Floridichthys carpio;* diamond killifish, *Adenia xenica* as well as certain warm temperate species with tropical family affinities, pinfish, *Lagodon rhomboides*; sheepshead, *Archosargus probatocephalus*; spotted seatrout, *Cynoscion nebulosus*. This area may have also acted as a evolutionary center for eurythermic tropical species that range into warm temperate waters today, Gag grouper, *Mycteroperca microlepis*, scamp, *M. phenax*, red grouper, *Epinephelus morio*; gray snapper, *L. griseus*; lane snapper, *L. synagris*; spottail porgies, *Diplodus holbrooki*; mojarras, *Eucinostomus gula, E. harengulus;* cocoa damselfish, *Stegastes variabilis*; slippery dick wrasse, *Halichoeres bivittatus;* emerald parrotfish, *Nicholsina usta*, Gulf surgeonfish, *Acanthurus randalli*, scaled sardine, *Harengula jaguana*.

Not only was the terrestrial Florida platform twice as large during the glacial periods, but so was the Yucatan platform, the Bahama platform and shelf areas off Nicaragua/Honduras and the Windward Islands of the West Indies. The Aves Ridge was closer to the surface also. Warm current flows into the Caribbean from the equatorial Atlantic were not as effective as they are today in entering the Sea. The Caribbean basin cooled down to a mean surface seawater temperature 5°C. lower than today's mean. Eurythermy in truly tropical species could have developed in the Caribbean basin during these glacial periods.

The more sensitive stenothermic tropical species were most likely limited to the Windward Islands and coastal South American from Trinidad south: Nassau grouper, *Epinephelus striata*; black grouper, *Mycteroperca bonaci*; tiger grouper, *M. tigris*; mahogani snapper, *Lutjanus mahogoni*; mutton snapper, *L. analis*; silver porgy, *Diplodus argenteus*; and many tropical coral reef fishes, as well as the euryhaline tropical peripherals: the fat snook, *Centropomus parallelus*, tarpon snook, *C. pectinatus*; Poey's snook, *C. poeyi*; burro grunt, *Pomadasys crocro*; opossum pipefish, *Mycrophis brachyurus lineatus*; bigmouth sleeper, *Gobiomorus dormitor*; esturarine stenothermic tropicals: patao, *Diapterus auratus*; slender mojarra, *Eucinostomus jonsei*; sea bream, *Archosargus rhomboidalis*; emerald parrotfish, *Cryptotomus roseus*; sailor choice, *Haemulon parra*.

With post-glacial global warming sea level rose, tropical western Atlantic land masses declined, warm ocean currents flowed more freely between these land masses and islands and tropical species were distributed to more northerly latitudes, including the Florida peninsula and survived. The majority of Florida fish species are tropical, having migrated to the Florida peninsula only recently, during the Holocene submergence of the Florida platform. These tropical invaders now form a heterogeneous group that often numerically dominate Florida fish communities, particularly those dependent on tropical habitats and ecosystems such as mangrove forests and coral reef formations. Many of these tropical species form the basis for major recreational and commercial fisheries. Most fishery species life histories require estuarine, coastal or shelf edge spawning often with passive/active movement of pelagic eggs and larvae into oceanic waters. Entrainment within oceanic currents allows larval transport for considerable distances downstream and, consequently, loss of larvae from Florida waters, conversely, larval recruitment from Caribbean biogeographical provincial waters.

Post-Glacial Faunal Movement

With the end of the Wisconsin glacial period and warming of waters around the Florida peninsula there was a regression of temperate and warm-temperate faunas northward. The Florida species most obviously impacted were those that prefer temperate climates and have life history strategies that require temperate climatic patterns or temperate ecosystems and habitats. Examples are the striped bass, *Morone saxatilis;* American shad, *Alosa sapidissima,* weakfish, *Cynoscion regalis,* Atlantic menhaden, *Brevoortia tyrannus*, black seabass, *Centropristis striata,* Atlantic sturgeon, *Acipenser oxyrhynchus,* and the sandtiger shark, *Carcharias taurus*. The only Atlantic sturgeon, striped bass and American shad populations remaining in Florida are found within extreme northern Florida, tributaries to the St. Johns River and the Appalachicola River. Striped bass are thought to be practically extinct from Florida today, resident populations being largely maintained by fish hatchery stocking (Dr. James Estes, Florida Fish and Wildlife Conservation Commission, pers. communication).

Warm temperate species still remain as important components of the Florida fauna on both coasts. However, they are interacting and competing with eurythermic and stenothermic tropical species particularly in estuaries from Florida Bay north to the southern Indian River Lagoon. Continental shelf faunas have undergone substantial change from temperate/warm-temperate glacial escarpment faunas to broad sub-tropical shelf faunas as far north as the northern Gulf of Mexico and Cape Hatteras.

With the recession of the temperate species came an expansion of eurythermic tropical species that may often be syntopic with warm-temperate species in estuarine or continental shelf environments. Notable eurythermic tropicals that have invaded the Florida peninsula after glaciation are the lane snapper, *L. synagris*, mangrove snapper, *L. griseus*, gag grouper, *M. microlepis*, scamp grouper, *M. phenax*, and red grouper, *E. morio*. Stenothermic tropical species have invaded continental shelves, estuaries and freshwater tributaries of tropical Florida. Examples are the mutton snapper, *L. analis*, tarpon snook, *C. pectinatus*, fat snook, *C. parallelus*, patao, *Diapterus auratus*, Mayan cichlid, *Cichlasoma uropthalmus*, black grouper, *M. bonaci*, Nassau grouper, *E. striata*, Cubera snapper, *Lutjanus cyanopterus*, and the dog snapper, *L. jocu*. The anthiine serranids and the *Epinephelus nigritus/niveatus/flavolimbatus/mystacinus* grouper complex, along with the *Lutjanus campechanus/purpureus* complex may have remained in deeper waters around the Florida peninsula and Gulf of Mexico during glacial periods as they would be buffered from surface glacial temperatures to a degree and can withstand ambient temperatures down to 6 - 8°C. today.

MODERN FISH FAUNA OF FLORIDA: FAUNAL COMPLEXITY FROM A COMPLEX ENVIRONMENT

Physical Complexity: Geomorphology, Transitional Climate, Differential Regional Oceanographic Currents

Figures 1 and 3 illustrate the present Florida peninsula physiography and climatic complexity, and associated oceanic current patterns as well as various hydrographic eco-regions. These eco-regions each have a fairly consistent fish community associated with them. These communities can be defined by their complement of temperate, warm-temperate, eurythermic tropical and tropical species as defined above.

In order to analyze the quantitative contribution of various species to the Florida inshore/estuarine fauna nekton surveys which required repetitive long term quantitative sampling programs, typically using trawl and haul seine techniques were used. Many studies have now been conducted throughout the Florida peninsula and elsewhere in the Caribbean which now allow quantitative regional ichthyofaunal assessment. At least three to four separate quantitative surveys were used for each estuary. The extensive reference list used to produce Table 3 are given in Gilmore (1987, 1995). Only the top ten species ranked by number of individuals captured are listed. They are then classified to three thermal and provincial association categories and percentage contribution to the total site fauna calculated.

No stenothermic tropical species made it into the top ten species along the Florida Gulf coast north of Cape Romano to Apalachee Bay. The faunas north of Marco Island and Cape Romano were dominated by warm temperate species (60 - 80 %). Eurythermic tropicals made up 14-40 % of the fauna, north Marco

Island, 54% at Marco/Cape Romano.

In contrast, the southeast Florida, Florida Bay to southern Indian River Lagoon estuarine faunas contained stenothermic tropical species that were abundant and often numerically dominating the local ichthyofauna. Eastern Florida Bay and the southern Indian River Lagoon contained more stenothermic tropicals than warm-temperate or eurythermic species (38-64 % of the fauna). Biscayne Bay and southern Indian River Lagoon revealed a greater biodiversity containing more species in the top ten categories (34 for each estuary) than other Florida estuaries.

North of the southern Indian River Lagoon the estuarine fish faunas are more like those of the Florida Gulf coast. The northern Indian River Lagoon does not have the stenothermic tropical component in its top ten species, 80% being warm temperate. An out-group comparison with the extensive studies done in Pamlico Sound, North Carolina (35° N latitude) shows no eurythermic or stenothermic tropical species dominating the estuarine fauna.

When continental shelf faunas are compared around the state of Florida there is broader distribution of stenothermic tropical species on the Florida east coast, eurythermic tropicals extending to North Carolina on the mid and outer shelf, while Gulf shelf faunas are dominated by warm-temperate and eurythermic tropicals.

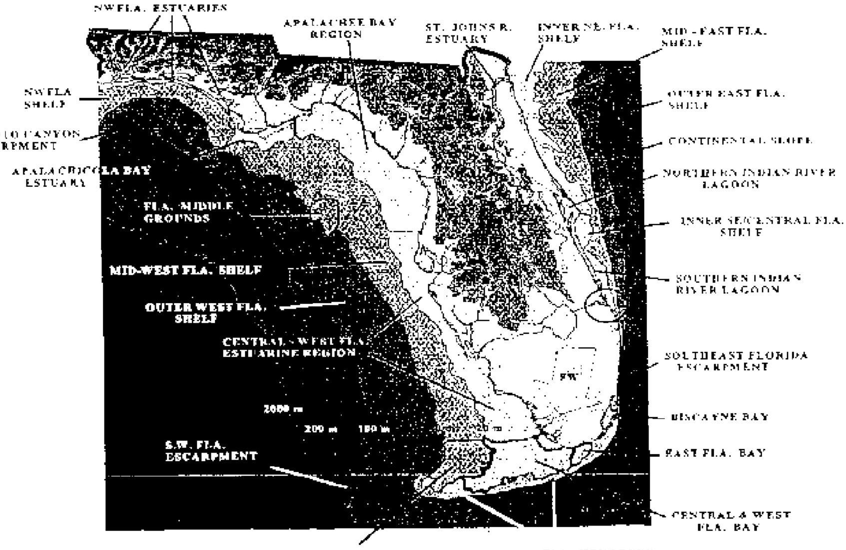


Figure 3. Twenty six fish communities and eco-regions for the Florida peninsula.

Table 3. Stenothermic tropical fishes contribution to Florida estuaries based on top ten species in 3-5 quantitative long term surveys per estuary or region. From studies cited in Gilmore 1987 and 1995. (IRL = Indian River Lagoon; Pam. Snd. = Pamlico Sound, NC)

Thermal Eco-physiological Status		**East**	**Gulf**	**Estuaries**		
	Apalachee (%)	Anclote Cedar K. (%)	Tampa (%)	Charlotte Harbor	Romano Marco (%)	Florida Bay (%)
Warm Temperate	6 (60.0)	11 (78.57	14 (66.67)	8 (80.00)	6 (46.15)	8 (34.78)
Eurythermic Tropical	4 (40.0)	2 (14.29)	7 (33.33)	2 (20.00)	7 (53.85)	9 (39.13)
Stenothermic Tropical	0 (0.)	0 (0.00)	0 (0.00)	0 (0.00)	0 (0.00)	6 (26.09)
Total Species	10	14	21	10	13	23
		Atlantic	**Estuaries**			
	E. Fla Bay	Biscayne	S. IRL	N. IRL	Pam. Snd.	
Warm Temperate	0 (0.0)	5 (14.71)	11 (32.35)	12 (80.00)	15 (100.00)	
Eurythermic Tropical	4 (36.36)	18 (52.94)	10 (29.41)	3 (20.00)	0 (0.00)	
Stenothermic Tropical	7 (63.64)	11 (32.35)	13 (38.24)	0 (0.00)	0 (0.00)	
Total Species	11	34	34	15	15	
	West/south	Estuaries				
	Laguna De Terminos	Tx				
Warm Temperate	8 (42.11)	9 (69.23)				
Eurythermic Tropical	3 (15.79)	4 (30.77)				
Stenothermic Tropical	8 (42.11)	0 (0.00)				
Total Species	19	13				

FISHERIES IMPLICATIONS OF BIOGEOGRAPHIC PATTERNS

Effects on Fisheries and Fishery Management

Knowing the preferred habitat, climatic and hydrological setting for all fish species allows the fishery manager to realistically assess the regional survival and production of a specific species, or guild of species. Florida fishery management is complex due to its complex fish/fishery fauna. This also holds true for much of the Gulf of Mexico and Caribbean Sea. The stocking of cultured fish for fishery purposes often has not recognized the indigenous range and life history of the stocked species. This not only could jeopardize the survival of the stocked species wasting agency funds on low survival rates, but also deleteriously impacting fish communities with which the species is interacting as an alien. Classification of entire faunas based on biogeographic distributions, preferred habitat and climatic/hydrological zones, life histories and ecology is required for realistic and cost-effective regional fishery management.

The Florida fish fauna is constantly changing with global warming and, consequently, habitat change. Temperate salt marshes are being displaced by tropical mangrove communities along both coasts of Florida. Management for temperate fish species will eventually be replaced by management for tropical species in the northern estuaries of Florida. The common snook, *Centropomus undecimalis*, rarely landed north of 28.5 - 29°N. latitude on both coasts of Florida will eventually be commonly landed as far north as 30°N. Cichlids will play a greater role in both estuarine and freshwater ecosystems, though, unfortunately, many of these cichlid species are from the tropical western Atlantic, thus not having previously invaded the Florida peninsula during prior interglacial periods. Cichlids are already commonly consumed and sold in regional markets. Eventually it may no longer be feasible and cost effective to stock cold water temperate species such as striped bass in Florida. For these reasons it is important that fisheries managers throughout Florida utilize the most accurate fish distribution and life history data available before making broad management decisions

LITERATURE CITED

Allen, G.R. 1985. *FAO Species Catalogue, Vol. 6: Snappers of the World.* FAO Fisheries Synopsis No. 125, Vol.6: 208 pp, 28 plates

Briggs, J.C. 1974. Marine zoogeography. McGraw Hill, N.Y. 475 pp.

Campton, D.E. and B. Mahmoudi. 1991. Allozyme variation and population structure of striped mullet (*Mugil cephalus*) in Florida. *Copeia* **1991**(2):485-492.

Emiliani, C. 1971. The amplitude of Pleistocene climatic cycles at low latitudes and the composition of glacial ice. Pages 183-197 in: L.

Turkian (ed.) *The Late Cenozoic glacial ages.* New Haven: Yale Univ. Press.

Emiliani, C. 1972. Quaternary paleotemperatures and the duration of the high-temperature intervals. *Science* **178**(4059):398-401.

Gilmore, R.G. 1987. Subtropical-tropical seagrass communities of the Southeastern United States: Fishes and fish communities. Pages 117-137 in: M.J. Durako, R.C. Phillips and R.R. Lewis III (eds.) Proceedings of the Symposium on Subtropical-tropical seagrasses of the Southeastern United States. Marine Research Publication 42, Fla. Dept. Nat. Res.

Gilmore, R.G. 1995. Environmental and biogeographic factors influencing ichthyofaunal diversity: Indian River Lagoon. *Bull. Mar. Sci.* **57**(1):153-170.

Gilmore, R.G. and P.A. Hastings. 1983. Observations on the ecology and distribution of certain tropical peripheral fishes in Florida. *Florida Scientist* **46**:31-51.

Gilmore, R.G., K.H. Bullock and F.H. Berry. 1979. Hypothermal mortality in marine fishes of south, central Florida, January, 1977. *Northeast Gulf Science* **2**:77-97.

Gold, J.R. and L.R. Richardson. 1998. Genetic homogeneity among geographic samples of snappers and groupers: evidence of continuous gene flow? *Proc. Gulf Carib. Fish. Inst.* **50**:709-726.

Heemstra, P.C. and J.E. Randall. 1993. *FAO Species Catalogue: Vol 16. Groupers of the World.* FAO Fisheries Synopsis No. 125, Vol 16: 382 pp., 31 plts)

Lee, T.N., E. Williams. 1999. Mean distribution and seasonal variability of coastal currents and temperature in the Florida Keys with implications for larval recruitment. *Bull. Mar. Sci.* **64**(1):35-56.

McGovern, J.C., G.R. Sedberry, H.S. Meister, T.M. Westendorff, D.M. Wyanski and P.J. Harris. 2001. A tag and recapture study of gag, Mycteroperca microlepis, from the Southeastern United States. Analytical Rpt. South Atl. Fish. Manag. Council. 1-24.

Miller, G.C. 1969. A revision of zoogeographical regions in the warm water area of the Western Atlantic. FAO Fisheries Rpt No. 71.1:141 pp.

Pattillo, M.E., T.E. Czapla, D.M. Nelson and M.E. Monaco. 1997. Distribution and abundance of fishes and invertebrates in the Gulf of Mexico estuaries: Vol. II: Species life history summaries. ELMR Rep. No. 11. NOAA/NOS Strategic Environmental Assessments Division, Silver Spring, MD. 377 pp.

Richards, W.J., M.F. McGowan, T. Leming, J.T. Lamkin and S. Kelly. 1993. Larval fish assemblages at the Loop Current boundary in the Gulf of Mexico. *Bull. Mar. Sci.* **53**(2):475-537.

Richardson, L.R. and J.R. Gold. 1993. Mitochondrial DNA variation in red grouper (*Epinephelus morio*) and greater amberjack (*Seriola dumerili*) from the Gulf of Mexico. *ICES J. Mar. Sci.* **50**:53-62

Thomson, J.M. 1963. Synopsis of biological data on the grey mullet, *Mugil cephalus* Linnaeus 1758. Div. Fish and Ocean., Common. Sci. and Ind. Res. Org., Fish Synop. No. 1: 80 pp.

Thomson, J. M. 1966. The grey mullets. *Oceanogr. Mar. Biol. Ann. Rev.* 4:301-335.

Van Sant, S.B., M.R. Collins and G.R. Sedberry. 1994. Preliminary evidence from a tagging study for a gag (*Mycteroperca microlepis*) spawning migration, with notes on the use of oxytetracycline for chemical tagging. *Proc. Gulf Carib. Fish. Inst.* **43**:417-428.

Transport of Larvae Originating in Southwest Cuba and the Dry Tortugas: Evidence for Partial Retention in Grunts and Snappers

KENNETH C. LINDEMAN[1], THOMAS N. LEE[2], W.D. WILSON[3],
RODOLPHO CLARO[4], and JERALD S. AULT[1]
[1]*Div. of Marine Biology & Fisheries*
RSMAS, Univ. of Miami
Miami, FL 33149 USA
[2]*Div. of Meteorology & Physical Oceanography*
RSMAS, Univ. of Miami
Miami, FL 33149 USA
[3]*Physical Oceanography Division*
NOAA, AOML
Miami, FL 33149 USA
[4]*Instituto de Oceanología de Cuba*
Habana, Cuba

ABSTRACT

New and existing information was assembled to parameterize forthcoming models examining transport scenarios of larvae spawned within and outside of protected areas of the lower Florida Keys and southwest Cuba. This information suggests that in some valuable fish families, partial retention of larvae frequently occurs in areas often assumed to primarily export spawning products. Four categories of biotic variables were emphasized: locations and timing of spawning, larval durations, growth rates, and larval behaviors. Over 30 spawning aggregation sites for eight snapper species have been tentatively identified. Based on otolith increment counts, mean larval durations ranged from 31-42 d post-fertilization in six snapper species (*Lutjanus* and *Ocyurus*) and from 13-20 d in four grunt species (*Haemulon* and *Anisotremus*). Behaviors that foster retention may influence recruitment more than larval durations, and vertical and horizontal distributional patterns differ markedly between grunts and snappers. Snapper larvae occur offshore while grunt larvae are rare in offshore or inshore surface waters despite often gregarious settlement events. Based on these information sets, grunt populations my be less subject to advection than snappers. Off southwest Cuba and within the Yucatan channel, many cyclonic and anticyclonic eddies have been recently identified by satellite-tracked drifters and over the horizon radar. Gyres associated with the Dry Tortugas and the Pourtales platform can also be present during spawning seasons. All of these recirculation systems can have residence times exceeding known larval durations. At least four sets of evidence can favor larval retention:

i) the larval biology of grunts practically precludes long-distance transport
ii) the numerous long-term eddy systems off of southwest Cuba
iii) the semi-continuous eddy system in the eastern margin of the Yucatan

Channel and associated backflow and countercurrents; and
iv) the gyres, eddies, and countercurrents along the Florida Reef Tract.

KEY WORDS: Haemulidae, larval transport, lutjanidae

INTRODUCTION

Biotic and abiotic variables influencing larval transport can underlie variability in local recruitment, genetic structure, recent zoogeographic distribution patterns, and optimal management of fishery stocks. At least four categories of biotic variables have a primary influence on larval transport: locations and timing of spawning; larval durations; rates of growth and mortality, and larval behaviors. Temporally-variable interactions between complex current systems and these developmental patterns determine the potential for larval dispersal.

Mean current speeds associated with the Florida Current and surface drifter tracks imply that larvae are transported in rapid, unidirectional manners off the Florida Keys and areas upstream. However, data on pink shrimp larvae have long suggested a moderate degree of larval retention within the Florida Bay/Florida Reef Tract system (Munro et al. 1967, Rehrer et al. 1967). Recently, a variety of large cyclonic and anticyclonic eddies and nearshore counter-currents that may favor retention have been detailed in this area (Lee et al. 1994, Lee and Williams 1999). In other regions, studies using mark and recapture, otolith microchemistry, and physical models have recently documented varying levels of self recruitment in reef fishes (Jones et al. in press, Swearer et al. in press, Cowen et al. in press).

Larval behaviors that foster retention may influence transport more than larval durations or surface current patterns. The rapid evolution of physical features that may promote retention, complex interactions among adjacent water masses, and constraints on the study of larval behavior present serious challenges to standard observational methods. Models which couple primary physical and biological parameters are needed to predict larval transport pathways into, within, and from the Florida Keys, and subsequent patterns of habitat use (Ault et al. 1999).

We assembled new and existing information necessary to parameterize and interpret coupled bio-physical models to predict potential larval transport paths off southwest Cuba and the Florida Keys. This work reflects preliminary efforts to model the release of particle clouds within eddies passing through marine reserve boundaries of these areas during known periods of spawning of snappers and grunts. Parameterization of these models requires empirical physical and biological data or, in their absence, logical proxies. We summarize new and existing information on:

i) spawning aggregation sites
ii) larval durations and growth rates
iii) larval behaviors, and
iv) physical patterns at primary scales

While assembling this information, several biotic and abiotic information sets suggested highly plausible mechanisms for at least partial retention of larvae. Primary examples are summarized here as alternatives to traditional hypotheses of larval transport.

METHODS

Taxa Examined and Spawning Site Identification

We focused on two families to account for taxa-specific differences in spawning, larval durations, growth rates, and larval behaviors. These families, snappers (Lutjanidae: genera *Lutjanus* and *Ocyurus*) and grunts (Haemulidae: genera *Haemulon* and *Anisotremus*), are among the most widespread and economically valuable of western Atlantic reef fishes (Robins and Ray 1986). At least nine of these species are overfished in the Florida Keys (Ault et al. 1998).

Information on spawning aggregations sites and seasonality in five snapper species from Cuba was obtained from Claro (1981, 1982), García-Cagide et al. (1994) and Claro et al. (in press). Similar information was obtained for aggregations of eight species from the Florida Keys (Domeier and Colin 1997, Lindeman et al. in press a). Spawning events in grunts are undocumented. Information on temporal patterns was assembled from studies of recruitment and gonad histology (e.g., McFarland et al. 1985, García-Cagide et al. 1994).

Larval Durations, Growth, and Vertical Positioning

Estimates of larval durations and early growth rates for 10 species of snappers and grunts were based on:

i) validation of daily deposition of otolith increments in several species
ii) identification of transitions in otolith increment spacing that may reflect age of settlement, and
iii) counts of otolith increments up to and beyond the settlement marks (Lindeman et al. in press a).

Direct information on larval behaviors is unavailable for almost all western Atlantic reef fishes. However, relevant information is available from horizontal and vertical distributional patterns in ichthyoplankton surveys (e.g., Richards 1984), and comparative patterns of morphological development and metamorphosis for grunts (Lindeman 1986) and snappers (Lindeman et al. in press b).

Physical Data and Potential Transport Paths

Southwest Cuba — Coarse-scale current systems and the primary physical literature for this area are summarized in Claro et al. (in press). Much new information is becoming available from satellite-tracked drifters deployed as part of the National Ocean Partnership Program - Year of the Ocean (NOPP-YOTO). These drifters were drogued at depths of 10 - 20 m and transmitted positions at sub-daily intervals. We plotted the paths of the drifters entering waters off southwest or south-central Cuba during periods of snapper and grunt spawning activity.

Yucatan Channel — New synoptic measurements of surface currents are being made for the Florida Straits and Yucatan Channel by over-the-horizon radar facilities. This technology can remotely monitor data-sparse areas that would otherwise require hundreds of widely dispersed *in-situ* instruments (Georges et al. 1998). We surveyed surface current maps of the Yucatan Channel based on several Relocatible Over the Horizon Radar (ROTHR) experiments conducted in 1998 and 1999 by the NOAA Environmental Technology Laboratory.

Florida Keys — As part of ongoing studies of current systems of the Florida Keys and interactions with Florida Bay and the Gulf of Mexico, several long-term current moorings are deployed along the Florida Keys outer shelf and the southwest FL shelf. Acoustic Doppler Current Profilers (ADCPs) are also moored in the middle and northern Keys. Via these tools, satellite-tracked surface drifters, and additional studies, considerable information on horizontal and vertical flow-field variation was available for nearshore and offshore components of the area (Lee et al. 1994, Smith 1994, Pitts 1994, Georges et al. 1998, Lee and Williams 1999).

RESULTS

Egg Release Sites

For many snapper species, spawning aggregation sites are not uniformly distributed across the shelves of southwest Cuba and southeast Florida. For example, Claro (1982) and García-Cagide et al. (1994) documented that lane snapper of the Golfo de Batabanó migrate to the east in April, against mean currents, to the margin of a deep and wide trough extending into nearshore waters, the Golfo de Cazones. At night, on days after the full moon, they spawn on this shelf edge at a depth of 30 - 40 m. Each individual spawns 4 - 5 times during one week. At the same time of the year, evidence suggests that a gyre is present in the center of the Golfo de Cazones for at least four week intervals. This gyre abuts the eastern margin of the Golfo de Batabanó, an area of abundant shallow habitats that may be favorable to settlement-stage larvae. A similar pattern is described for mutton snapper of the northeast Cuban shelf (Claro 1981). In total, seven and ten potential aggregation sites have been identified in Cuba for lane and mutton snapper, respectively (Claro et al. in press). In south Florida, probable spawning aggregations of eight snapper species at 25 sites have been identified (Domeier and Colin 1997, Lindeman et al. in press a). Peak spawning in most species occurs between May and September (García-Cagide et al. 1994).

Spawning in most grunts occurs year-round, with spring and summer peaks in many species (McFarland et al. 1985, García-Cagide et al. 1994). Social behavior and habitat use during spawning are completely unknown despite many observation attempts. This suggests spawning occurs well after dusk. High abundances of recruits semi-continuously occur throughout the region and older life stages typically occur in deeper reef habitats. Therefore, we assumed that most egg release sites for grunts are near depths of 20 - 30 m in mid-shelf and shelf-edge areas.

Larval Durations and Growth Rates

Based on otolith increment transitions associated with settlement, estimates of mean planktonic larval durations (PLDs) ranged from 31 to 42 d post-fertilization in six snapper species. No significant differences were found among species and the mean for all was 32 day (Lindeman et al. MSa). Sample sizes were low and these estimates are preliminary. Settlement marks are not documented in grunts and PLDs were based on total daily ring counts from newly-settled individuals. Estimated PLDs ranged from 13 - 20 days in four grunt species. Larval duration estimates are plotted for both families in Figure 1, with groupers, evolutionarily similar percoids, included for comparison. The mean growth rate of all snapper species pooled (0.81 mm/day, n = 63) was significantly higher than grunts (0.47 mm/day, n = 76) (Lindeman et al. MSa). These rates are for sizes ranging from 10 - 70 mm SL. Evidence suggests that larval growth rates may show a similar pattern between families, at least after the flexion stage.

Vertical and Horizontal Distributions of Larvae

PLDs are not the only or best measure of retention/advection potential. Larval behaviors that foster retention can take many forms and may influence recruitment more than larval durations. Direct information on larval behaviors and their size-specific variation is limited. However, considerable amounts of relevant information were available on horizontal and vertical distribution patterns from ichthyoplankton sampling (e.g., Leis 1987, Richards et al. 1993, Paris-Limouzy et al. 1997). Coupled with information on ecological and morphological patterns of metamorphosis (Lindeman 1986, Lindeman and Snyder 1999, Lindeman et al. MSb), it is possible to outline known developmental patterns and to posit logical hypotheses of the behavioral alternatives that explain these patterns.

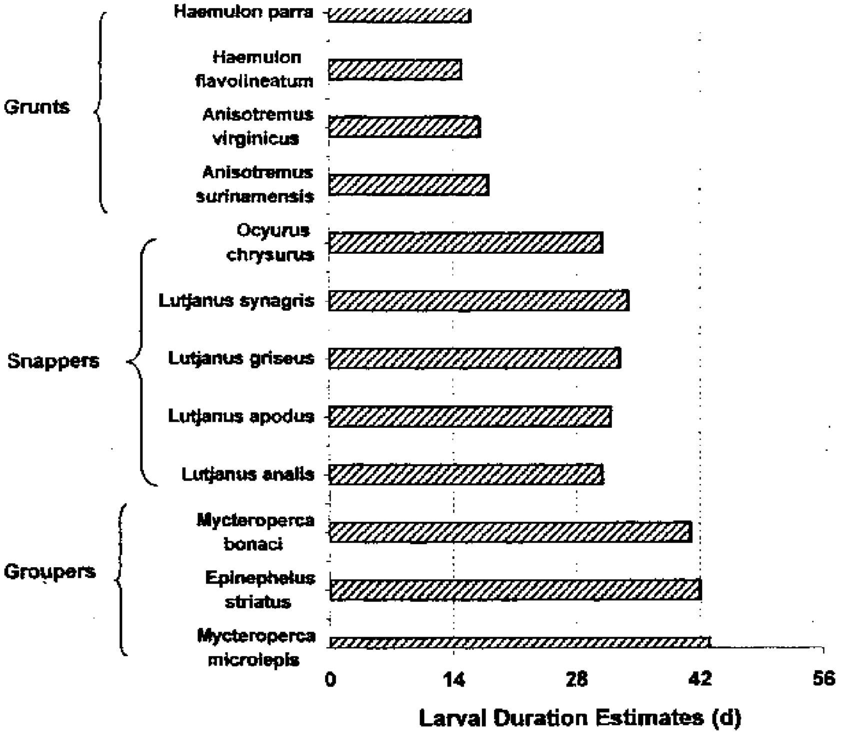


Figure 1. Larval duration estimates for grunt, snapper, and grouper species based on otolith increment counts. Data sources detailed in Lindeman et al. (in press a). Only specimens from southeast Florida and Cuba used for grunts and snappers.

Snapper larvae are commonly known from offshore and surface plankton sampling, whereas grunt early larval stages rarely enter offshore or surface waters. Haemulids are also rare in surface nets or nightlights at inshore stations despite often being the most abundant "settlers" on nearby benthic habitats (Table 1). These consistent patterns and an extremely gradual metamorphosis imply that grunts may associate with the mid or lower water column over continental or insular shelves from the flexion stage or earlier (Lindeman et al .in press a).

Table 1. Comparative patterns of early development among four genera of haemulids and lutjanids. Based on specimens from southeast Florida and Cuba. Modified from Lindeman et al. in press a.

Attribute	Haemulon & Anisotremus	Lutjanus & Ocyurus
	Pre-settlement Stages:	
Offshore larvae	Rare/Absent	Occasional
Surface occurrence	Rare/Absent	Occasional
Diel vertical	Absent?	Present
	Metamorphosis:	
Age range at	13 - 20 d	24 - 42 d
Typical settlement	6.5 - 9 mm	10 - 18 mm
Otolith settlement	No?	Yes
Morphological	Slow (months), subtle	Rapid (days), distinct
	Newly-settled Stages:	
Demersal growth	Slow; 0.27 - 0.65 mm·d-1	Fast; 0.68 - 0.94 mm·d-1
Schooling behavior	Gregarious; often school	Solitary; or in small, non-schooling groups
Adult pigment acquisition	Delayed (months)	Rapid (days/weeks)

Based on the differences identified in Table 1 and the source literature, we hypothesize that positioning and larval durations of post-flexion stages of snappers and grunts can be characterized as follows:

	Larval Duration	Vertical Position		Horizontal Distribution
		Day	Night	
Snappers	30 d	20 - 40 m	0 - 20 m	Offshore/Inshore
Grunts	14 d	Near bottom	Near bottom	Inshore only

Variations in these parameters exist within the families and probably within species (e.g., among geographic regions). However, based on currently available information, these variations are not large enough to contradict these primary hypotheses. The ultimate proportion of larval retention or advection within each family involves the interplay of these parameters with the physical environment of the two study regions during peak spawning periods.

Physical Processes

Characterizations of the primary current systems and their forcing mechanisms in these regions include Sukhovey et al. 1980, Lee et al. 1991, 1994, Smith 1994, Victoria del Río and Penié 1998, and Lee and Williams 1999). We focus here on new information of relevance to larval transport processes.

Southwest Cuba - New information on upper mixed layer current patterns and their variability have recently become available for waters off southwest Cuba based on satellite-tracked drifter paths from 10 m depths. Plots of four drifter paths that entered southwest Cuban waters in 1998 and 1999 are given in Figure 2. Drifter 23472 was released Julian day 291/98 at 10.7 N 76.7 W. It made one cyclonic loop within the Panama-Colombia gyre in approximately 60 days and entered southwest Cuban waters in February 1999. Circulation was then dominated by cyclonic eddies with swirl diameters of 10-100 km (Figure 2). This drifter remained south of Isla de la Juventud for 3 months. Drifter 30659 was released 151/98 at 10.9 N 76.7 W. It moved north and entered the region from the west, south of Cabo San Antonio, in July 1998. It spent eight months in various eddies in the region before exiting the region in March 1999. It subsequently drifted west and north through the Florida Straits in April 1999. Drifter 30660 was released 150/98 at 11.2 N 77.4 W. It made two cyclonic loops of the Panama-Colombia gyre before entering the region in October 1998. It made one large anticyclonic loop with many smaller cyclonic eddies before exiting the region to the west in March (Figure 2) and grounding in Belize in April 1999. Drifter 15592 was released in February 1999 off southwest Cuba. It resided for over 30 d in cyclonic eddies with swirl diameters of 20-50 km (Figure 2).

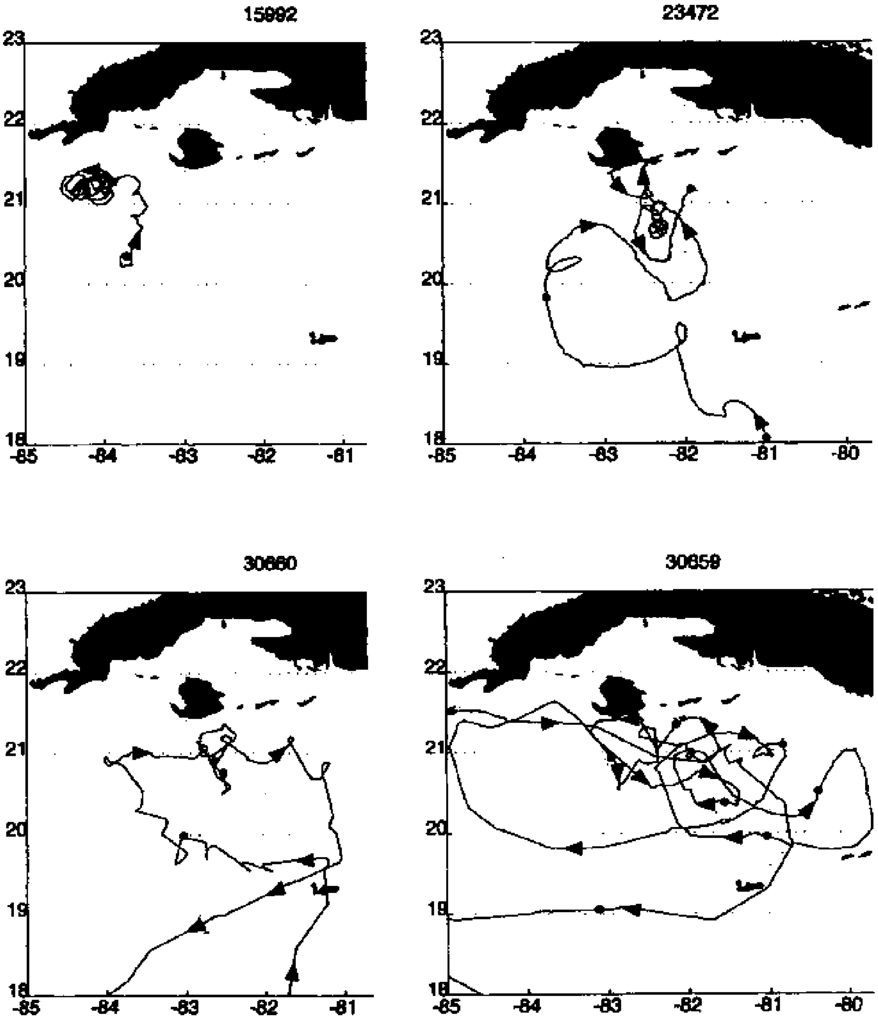


Figure 2. Eddy systems off southwest Cuba. Based on satellite-tracked drifters tracking currents at 10-20 m depths. Dates and durations of drifter paths in text.

Yucatan Channel - ROTHR data for surface currents in the Yucatan Channel were available for 16 days from June 1998 to October 1999. The western margin of the channel consistently showed strong northerly currents at speeds of 0.75-2.0 m/s. However, the central axis and eastern margin of the channel were typically dominated by a large anticyclonic eddy feature (Figure 3). The eddy system showed complex spatial dynamics and appeared to often meander to the north and south of the channel. However, this system was present in all seasons and almost all of the ROTHR images. The system varied from elliptical to spherical in shape with diameters ranging from approximately 50-120 km. Two examples of eddy positioning are given in Figure 3. Data from 6 May and 21 May 1999 show a large eddy southwest of Cabo San Antonio that was generating strong southerly backflow. Other imagery suggests that portions of this backflow may bifurcate and form a substantial component of the easterly Cuban Countercurrent.

Florida Keys — Potential recruitment pathways for larvae spawned in the Florida Keys or from external sources are driven by at least four physical processes: the Florida Current; cyclonic circulation of the Tortugas and Pourtales Gyres; onshore surface Ekman transport; and coastal countercurrents. Physical characteristics of these features relevant to larval retention and advection have been reviewed thoroughly by Lee et al. (1994), Yeung (1996) and Lee and Williams (1999). The shoreward front of the Florida Current is an area of near-surface current convergence. Therefore, larvae and their planktonic foods tend to be concentrated in the frontal zone. Onshore meanders and shoreward displacement of this front transport larvae closer to the coastal zone in the middle and upper Keys. Lobster and conch larvae occur near the outer reefs when the Florida Current is nearshore (Stoner et al. 1997). Small frontal eddies (20 - 50 km diameter) and the larger Tortugas gyre (100 - 200 km) aid the inshore exchange of larvae (Criales and Lee 1995, Limouzy-Paris et al. 1997). The Tortugas Gyre and smaller eddies can entrain larvae for weeks or months and enhance food availability through upwelling and the concentration of zooplankton (Lee et al. 1994).

A westward countercurrent inside the outer reefs results from the influences of downwelling winds and coastal gyres, and can extend from the middle Keys to the Tortugas. Its northern extent is limited by the curving coastline that causes prevailing westward winds to orient alongshore in the lower Keys and onshore in the upper Keys. There is also evidence that the coastal countercurrent extends shoreward to Hawk Channel and other nearshore waters (Lee 1986, Pitts 1994). Thus, larvae that are detrained from the Florida Current can be transported westward and shoreward by the coastal countercurrent, providing opportunities for recruitment to nearshore areas.

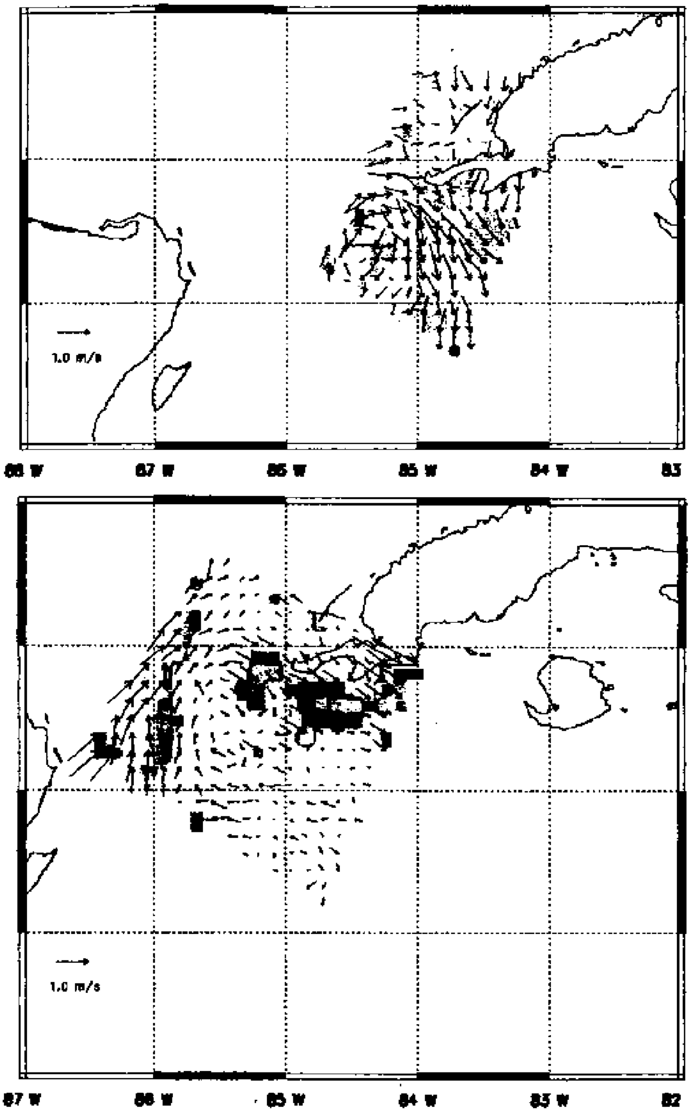


Figure 3. Two representations of anticyclonic eddy systems within the Yucatan Channel. Surface current images from relocatable over the horizon radar (images from T.M. Georges and colleagues). Top: Eddy positioned in the southeast portion of the channel with backflow exceeding 1 m/s, 6 May 1999. Bottom: Centrally-positioned eddy, 13 November 1998.

DISCUSSION

Alternative Larval Transport Scenarios

New subsurface drifter tracks off southwest Cuba and radar data from the Yucatan Channel suggest that larvae spawned from the massive reef system bordering the southern margin of the Golfo de Batabanó are not necessarily entrained into a unidirectional current field flowing towards and through the Yucatan Channel. Larvae may have to traverse a variety of physical retention mechanism before even reaching the axis of the Yucatan Channel. The residence times of the various eddies south of the Golfo de Batabanó easily exceed the PLDs of both snappers and grunts (Figures 1 and 2). Larvae that are not entrained in these eddies and travel west may still encounter the eastward Cuban Countercurrent which is produced in part from backflow from the eddy often at the eastern margin or center of the Yucatan Channel (Figure 3), or be entrained in this eddy itself.

Rapid downstream transport may occur in the western margin of the Yucatan Channel and Florida Current at velocities of 2 m/s. However, mean surface currents do not fully capture the dynamics of larval transport. Vertically stratified flows may transport larvae in manners that contradict assumptions from unidirectional surface flows and PLD estimates. Many current systems commonly thought to be dominated by unidirectional flow frequently generate cyclonic or anticyclonic circulation systems along frontal boundaries in response to a variety of shearing forces. These frontal boundaries are often areas of high plankton density (Richards et al. 1993).

NOPP-YOTO drifter releases of 1998 - 1999 increased the database on Lagrangian flows in the Caribbean by an order of magnitude and their trajectories are revealing very complicated patterns (Figure 2) in areas often considered to have relatively unidirectional flow. The 10 - 20 m operational depths of these drifters may be more representative of larval vertical positioning than surface drifters, thus providing more realistic larval transport trajectories. However, additional information on the vertical stratification of both flow-fields and larval distributions is needed. Recent work in Australia has identified exceptional swimming speeds and maneuvering abilities in late pelagic larvae of many taxa (Stobutzki and Bellwood 1997, Leis and Stobutzki 1999). Laboratory and field investigation of these processes in western Atlantic species, particularly vertical manuvering, will shed considerable light on the capacities of larval fishes to maximize retention or advection, and to traverse frontal boundaries.

Phylogenetic and Geographic Scales

Transport scenarios may vary widely among and within families. Grunts and snappers showed major contrasts in many early demographic parameters, despite close phylogenetic affinities. Along with differing PLDs, these families showed many developmental contrasts that suggest very different transport potential (Table 1). These include differences in:

i) inshore/offshore distribution patterns
ii) vertical distributions
iii) comparative larval morphologies and growth rates
iv) microstructure of otolith increment transitions
v) diel larval behaviors
vi) settlement size; and g) patterns of schooling at settlement.

Given the evidence that snappers are in the plankton for 10 - 20 days longer than grunts and that grunts do not exit the shelve, do not use surface waters, and may engage in near-bottom associations by the flexion stage, it is hypothesized that grunt populations are more likely to self-recruit than snappers. If correct, this hypothesis may have several consequences for the metapopulation structure of these wide-ranging species. For example, grunt stocks may have less potential for genetic exchange with both up- and down-stream populations than snappers.

Considerable evidence now suggests logical alternatives to the assumption that physical transport in the northern Caribbean is largely unidirectional and that larvae behave as passive particles. However, this evidence does not exclude hypotheses of extended transport in some taxa; particularly in our study area which includes adjacent islands and continental shelf subsystems.

Outside of isolated islands, mean larval transport patterns over decadal time scales probably consist of both retention and intermediate-distance advection episodes that vary in frequency within and between years, and as a function of specific locations. In the Caribbean, the distributions of some species may be influenced by larval transport processes operating on the scale of individual basins or areas within. For example, two species of grunts commonly recorded from coastal areas of the Columbian Basin, *Haemulon steindachneri* and *H. bonariense*, are not present or rare in seemingly downstream areas such as Honduras and Belize (Lindeman et al. in press b). A viable hypothesis to explain these disjunct distributions is the combination of the Panama-Columbia Gyre and the apparent lack of larval dispersal capabilities in grunts. This large, cyclonic system may limit the downstream transport of some larvae from the western Columbian Basin. If correct, this hypothesis may not be universal as other *Haemulon* species do not show such disjunct distributions although subject to the same gyre. Explanations for these patterns may involve differences in spawning times, larval behavior, or juvenile ecology at the species level. This highlights the need for more information on geographic variation in population parameters within key species and genera.

ACKNOWLEDGEMENTS

ROTHR imagery was made available through the work of T. M. Georges and colleagues at the NOAA Environmental Technology Laboratory. Many workers have been involved in the NOPP-YOTO drifter program in the Caribbean, in particular we thank Kevin Leaman. Bob Cowen and Bill Richards provided valuable comments on the manuscript. This research received partial support from NOAA Coastal Ocean Program Grant No. NA37RJ0200, United States Man and the Biosphere Marine and Coastal Ecosystems Directorate Grant No. 471014223B, and S. Fl. Ecosystem Restoration Protection Modeling Grant No. NA67RJ0149.

LITERATURE CITED

Ault, J. S., J. A. Bohnsack and G. Meester. 1998. A retrospective (1979-1995) multispecies assessment of coral reef fish stocks in the Florida Keys. *Fish. Bull.* **96**(3):395-414.

Ault, J.S., J. Luo, S.G. Smith, J.E. Serafy, J.D. Wang R. Humston, and G. Diaz. 1999. A spatial dynamic multistock production model. *Can. J. Fish. Aquat. Sci.* **56**:1-22.

Claro, R. 1981. Ecología y ciclo de vida del pargo criollo, *Lutjanus analis* (Cuvier), en la plataforma cubana. *Inf. Cient.-Tec. Acad. Cienc. Cuba.* **186**:1-83.

Claro, R. 1982. Ecología y ciclo de vida de la biajaiba, *Lutjanus synagris* (Linnaeus), en la plataforma cubana. IV: Reproducci\n. *Rep. Invest. Inst. Oceanol. Acad. Cienc. Cuba*, **5**:1-37.

Claro, R., K.C. Lindeman and L.R. Parenti (eds.). *Ecology of marine fishes of Cuba*. Smithsonian Institution Press. In Press.

Cowen, R.K., K.M.M. Lwiza, S. Sponaugle, C. B. Limouzy-Paris, and D.B. Olson. Connectivity of marine populations: open or closed? *Science*. In Press.

Criales, M.M. and T.N. Lee. 1995. Larval distribution and transport pf penaeoid shrimps during the presence of the Tortugas Gyre in May-June 1991. *Fishery Bulletin.* **93**:471-482.

Domeier, M. L. and P. L. Colin. 1997. Tropical reef fish spawning aggregations: defined and reviewed. *Bull. Mar. Sci.* **60**(3):698-726.

García-Cagide, A., R. Claro, and B. V. Koshelev. 1994. Reproducción. Pages 187-262 in: R. Claro (ed.). *Ecología de los peces marinos de Cuba.* Centro de Investigaciones de Quintana Roo and Instituto de Oceanología de Cuba.

Georges, T. M., J. A. Harlan, T. N. Lee, and R. R. Leben. 1998. Observations of the Florida current with two over-the-horizon radars. *Radio Science* **33**(4): 1227-1239.

Jones, G. P., M.J. Milicich, M.J. Emslie, and C. Lunow. Self-recruitment in a coral reef fish population. *Nature*. In Press.

Lee, T. N., J. A. Yoder, and L. P. Atkinson. 1991. Gulf Stream frontal eddy influence on productivity of the southeast U.S. continental shelf. *J. Geophys. Res.* **96**(C12):22,191-22, 205.

Lee, T. N., M. E.Clarke, E. Williams, A. F. Szmant, and T. Berger. 1994. Evolution of the Tortugas Gyre and its influence on recruitment in the Florida Keys. *Bull. Mar. Sci.* **54**:621-646.

Lee, T. N. and E. Williams. 1999. Mean distributions and seasonal variability of coastal currents and temperature in the Florida Keys with implications for larval recruitment. *Bull. Mar. Sci.* **64**(1):35-56.

Leis, J. M. 1987. Review of the early life history of tropical groupers (Serranidae) and snappers (Lutjanidae). Pages 189-237 in: J. J. Polovina, and S. Ralston, (eds.) *Tropical snappers and groupers: biology and fisheries management.* Westview Press, Boulder, Colorado.

Leis, J.M. and I.C. Stobutzki. 1999. Swimming performance of late pelagic larvae of coral-reef fishes: *in situ* and laboratory-based measurements. Pages 1-9 in: B. Séret and J.-Y. Sire (eds.). *Proc. 5th Indo-Pacific Fish Conf.*, Paris.

Limouzy-Paris, C.B., H.C. Graber, D.L. Jones, A.W. Röpke, and W. J. Richards. 1997. Translocation of larval coral reef fishes via sub-mesoscale spin-off eddies from the Florida Current. *Bull. Mar. Sci.* **60**(3):966-983.

Lindeman, K. C. 1986. Development of larvae of the French grunt, *Haemulon flavolineatum*, and comparative development of twelve western Atlantic species of *Haemulon*. *Bull. Mar. Sci.* **39**(3):673-716.

Lindeman, K.C. and D.B. Snyder. 1999. Nearshore hardbottom fishes of southeast Florida and effects of habitat burial by dredging. *Fish. Bull.* **97**(3):508-525.

Lindeman, K.C., R. Pugliese, G.T. Waugh, and J.S. Ault. Developmental patterns within a multispecies reef fishery: management applications for essential fish habitats and protected areas. *Bull. Mar. Sci.* In Press a.

Lindeman, K.C., C. Toxey, and K.E. Carpenter. Haemulidae - Grunts. In: K.E. Carpenter (ed.). *FAO Species Identification Sheets for the Western Central Atlantic (Fishing Area 31).* Food and Agricultural Organization, United Nations. In Press b.

Lindeman, K.C., E.B. Brothers, and J.S. Ault. Comparative growth and life history transitions of newly-settled lutjanids and haemulids (Percoidei) of the western Atlantic. In Prep a.

Lindeman, K.C., W.J. Richards, C. Paris-Limouzy, J. Schultz, D.M. Drass, B.H. Comyns and J.M. Leis. Development of western Atlantic snappers (Lutjanidae). In Prep b.

McFarland, W.N., E.B. Brothers, J.C. Ogden, M.J. Shulman, E.L. Bermingham, and N.M. Kotchian-Prentis. 1985. Recruitment patterns in young French grunts, *Haemulon flavolineatum* (family Haemulidae), at St. Croix, Virgin Islands. *Fish. Bull.* **83**:151-161.

Pitts, P.A. 1994. An investigation of near-bottom flow patterns along and across Hawk Channel, Florida Keys. *Bull. Mar. Sci.* **54**(3):610-620.

Richards, W. J. 1984. Kinds and abundances of fish larvae in the Caribbean Sea and adjacent waters. *NOAA Tech. Rept.* NMFS SSRF-776, 54 pp.

Richards, W.J., M.F. McGowan, T. Leming, J.T. Lamkin and S.Kelley. 1993. Larval fish assemblages at the loop current boundary in the Gulf of Mexico. *Bull. Mar. Sci.* **53**(2): 475-537.

Smith, N. P. 1994. Long-term Gulf-to-Atlantic transport through tidal channels in the Florida Keys. *Bull. Mar. Sci.* **54**:602-609.

Stoner, A. W., N. Mehta, and T. N. Lee. 1997. Recruitment of *Strombus* veligers to the Florida reef tract: relation to hydrographic events. *J. Shellfish Res.* **16**:1-6.

Stobutzki, I. C. and D. R. Bellwood. 1997. Sustained swimming abilities of the late pelagic stages of coral reef fishes. *Mar. Ecol. Prog. Ser.* **149**:35-41.

Swearer, S.E., J.E. Caselle, D.W. Lea, and R.R. Warner. Larval retention and recruitment in an island population of a coral reef fish. *Nature*. In Press.

Sukhovey, V.F., G.K. Korotaev and N.B. Shapiro. 1980. *Hydrology of the Caribbean Sea and Gulf of Mexico*. Leningrado, Gidrometeoizdat. [In Russian], 180 pp.

Wilson, W.D., and W.E. Johns. 1997. Velocity structure and transport in the Windward Islands Passages. *Deep-sea Res.* **44**(3):487-520.

Victoria del Rio, I. and I. Penié. 1998. Hidrología, Pages 117-125 in: M. Vales, A. Alvarez, L. Montes and A. Ávila (eds.), *Estudio Nacional sobre la Diversidad Biológica en la República de Cuba*. PNUMA /CENBIO/IES/AMA/CITMA Rept., Habana.

Yeung, C. 1996. *Larval transport and retention of lobster phyllosoma larvae in the Florida Keys*. Ph.D. Dissertation. University of Miami, Coral Gables, FL. 217 pp.

The Coral Reef Fishes of Broward County Florida, Species and Abundance: A Work in Progress

B.D. ETTINGER,[1] D.S. GILLIAM[1,2], L.K. JORDAN[1],
R.L. SHERMAN[1] and R. E. SPIELER[1,2]
[1] *Nova Southeastern University*
[2] *National Coral Reef Institute*
Oceanographic Center
Dania, FL 33004 USA

ABSTRACT

The inshore environment of Broward County, FL consists of three coral reef/hard bottom reef tracts, separated by sand substrate, running parallel to the coast in sequentially deeper water. At quarter nautical mile intervals, for a five mile coastline section, fishes were censused at western, eastern and crest sites of each of the three reef tracts. On SCUBA and using the Bohnsack/Bannerot point-count method, we recorded: fish abundance, species richness, size, and general habitat of an imaginary cylinder 15 m in diameter. The position of each site was recorded by DGPS after census. One hundred and eighty-one sites were censused during a 10 month period (August 1998 to May 1999).

A total of 16,746 fish belonging to 139 species of 39 families were recorded. There were significant differences ($p < 0.05$, ANOVA) in the species richness and the total abundance of fishes among the three reef tracts. There were significantly fewer total fish ($p < 0.001$) and fewer fish species ($p < 0.001$) on the inshore reef tract as compared to either the middle or offshore reef tracts. The middle and offshore reefs tracts did not differ ($p > 0.05$, SNK). Differences were also found based on the location (edges or crest) on the reef. With all data from the three reef tracts combined, the eastern edge showed significantly fewer total fish ($p < 0.001$) than either the crest of the reef or the western edge, which did not differ significantly ($p > 0.05$). Species richness also varied with the western edges of the tracts having significantly more species ($p < 0.01$) than the crests or the eastern edges, again there was no significant difference between these two ($p > 0.05$). Statistical analysis of a subjective complexity rating taken at each site mirrored the results of fish abundance and species richness. This supports the hypothesis that topographical complexity is, at least in part, a determinant in the differences in fish assemblages among the three reef tracts.

KEY WORDS: Visual census, coral reef fish, complexity

INTRODUCTION

Monitoring of marine resources is becoming more important as the popularity of fishing, diving, boating and development of coastal areas increases. Baseline surveys are a vital part of this monitoring process. With little to no studies conducted in rapidly growing and changing areas such as Broward County, Florida, USA, it is important to determine a baseline inventory and understanding of the fish assemblages. Such data is required to monitor changes resulting from anthropogenic or natural activities and provide useful information for the management of aquatic resources.

The purpose of our research is to conduct a quantitative study of the inshore marine fishes of Broward County. This preliminary study examined both abundance and distribution of reef fishes on Broward County's three natural reef tracts over a stretch of five nautical miles (approximately 25% of Broward County's coast line).

METHODS AND MATERIALS

Study Area

The near-shore marine environment off the coast of Broward County, FL, USA, is characterized by three reef tracts that run parallel to the coast in a north-south direction, in sequentially deeper water (Goldberg, 1973). The individual reef tracts are plateau-like structures with the largest changes in profile usually at either their western or eastern edges or both. Sandy soft sediment separates each reef tract. Hereafter we refer to these three tracts as the inshore, middle, and offshore reef tracts.

The area of this study was located off the coast of Broward County from Port Everglades inlet (265 05.5' N, 0805 06' W) south for five nautical miles and from the western edge of the inshore reef tract to the eastern edge of the offshore reef tract (approximately 1.5 nautical mile). The area was divided into 20, east-west transects, each transect separated from its neighboring transect by one quarter nautical mile. Along each transect were nine sample sites: one on the western edge, one on the reef crest (or midpoint if no crest was determined) and one on the eastern edge of each of the three reef tracts for a total of 180 planned sites.

Sampling Technique

Sampling was conducted using the non-destructive Stationary Visual Census Technique (Point-Count) which censuses the fish in an imaginary 15 m diameter cylinder from substrate to surface (Bohnsack and Bannerot 1986). Individual transects were followed, with the aid of DGPS, from inshore to offshore and using a depth plotter, the bottom composition was observed and topographical characteristics were determined. The gross topographical characteristics were used to determine specific sites along the tract. Sites, on each transect, were chosen to represent the eastern and western edges and crest of each of the three reef tracts. In certain instances there were no obvious crests and a middle point between the eastern and western edges was used as the crest sample site. A total of 20 transects were sampled.

Buoys were deployed to mark each site. Divers were deployed in two man teams consisting of one trained fish counter and a safety diver. Each dive team was equipped with: a 7.5 m line and attached anchor weight; a clipboard with a waterproof data sheet and pencil; an underwater watch; and a one meter fish-stick, used as an aid in measuring fish lengths. At the bottom, the safety diver would attach a 7.5 m line to the buoy line, then swim it out from the center in a straight line before anchoring it on the bottom in an area clear of delicate benthic organisms. This line was used as a reference to define the sample cylinder. The safety diver would then remain outside of that cylinder, within visual contact, while the trained fish counter would complete the sample. Prior to beginning the fish count, the counter recorded the bottom coverage, estimated in percent of 21 different substrates (i.e. coral, sand, sponge etc.),and assigned a complexity rating to the site. The complexity rating was a subjective scale from 1 - 10 which used a local, complex site as a standard. On initiating the census the fish counter would pivot around to scan the entire cylinder and would record all species observed during a five minute period. Following this initial five minute sample, the abundance, mean size, minimum size and maximum size were recorded for each species observed during the initial five minutes. Sample times outside of the five minute initial count were normally kept to no more than 20 minutes. The 20 minute time limit was sufficient to complete abundance and size data collection and allowed divers to complete repetitive dives within a day without decompression.

The data collected was entered into RVC Data Entry System (Weinberger 1998) program, Microsoft Excel and SAS (Statistical Analysis Systems). Microsoft Excel was used to determine general descriptive statistics. The same data entered into SAS was analyzed with non-parametric analysis of variance techniques [PROC GLM of ranked data (≈ Kruskal-Wallis k-sample test), and the Student-Newman-Keuls test between means].

RESULTS

General site description

During data collection, we noted the sample sites included a variety of environments including large reef rock, coral rubble, ledge lines and low profile hard bottom. The inshore reef tract was found in depths ranging from 3.1 m to 9.2 m with the lowest point of the western edge ranging from 4.6 m to 8 m deep. The crest of the inshore reef ranged from 3.1 m to 8 m. The eastern edge of the inshore reef was found between 5.2 m and 9.2 m. Generally, the western edge of the inshore reef tended to be a well defined ledge. The crest of the inshore reef varied from relatively barren hard bottom to diverse coverage with very little to no vertical profile. In most cases, the eastern edge of the inshore reef tapered out from low relief hard bottom to rubble and sand (Figure 1).

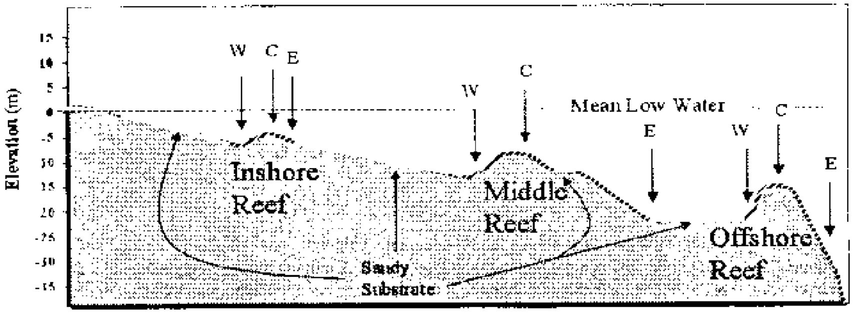


Figure 1. Depth profileof a generalized transect from west to east. Census sites indicated by: W = Western Edge, C = Crests, E = Eastern edges. Diagram not to scale.

The middle reef usually had a well defined western ledge followed by an obvious crest that would taper out to a wide sand flat. The reef extended from 7.4 m to 21.8 m in depth with the western edge 7.7 m to 13.5 m deep; the crest ranged in depth from 7.4 m to 13.2 m; and the eastern edge was between 10.8 m and 21.8 m deep. Unlike the western edge, the eastern edge of the middle reef varied from a ledge to an edge with continual hard bottom which lead onto an initial sandy flat. Following this sand flat there was a narrow downward hard bottom approximately 15.4 m in width, before a well defined edge occurred followed by a sand flat. For this study, the eastern edge of the middle reef was defined as this second edge.

The offshore reef tract ranged in depth from about 14.2 to 31.4 m. The western edge of the offshore reef begins at a depth of about 15.4 m but was found as deep as 20.9 m. The offshore crest was found anywhere between 14.2 and 19.4 m while the eastern edge of the offshore reef ranged from 24.9 m to over 31.4 m deep. The western edge of the offshore reef was generally defined by an obvious edge. The crest of the offshore reef was found to be hard bottom with varying amounts of hard and soft corals. The eastern edge was well defined with, in some cases, the appearance of spur and groove formations. In some instances our definition of the eastern edge was limited by our diving limitations (35 m).

It is important to note that the characteristics for the three reef tracts, outlined above, represent an average impression; even within an edge of a single reef tract there were substantial differences in depth and habitat characteristics.

Study sites were initially sampled between August 1998 and May 1999. A total of 181 sites were surveyed during this period. Although we planned to census 180 unique sites, four planned sites were inadvertently missed and one transect received five additional sites. A total of 16, 746 individual fish made up of 139 species representing 39 families were recorded.

Statistical analysis indicated significantly lower abundance ($p < 0.001$) and lower species richness ($p < 0.001$) on the inshore reef tract as compared to either the middle or offshore reef tracts. The middle and offshore reefs tracts did not appear to differ between each other ($p > 0.05$) (Figures 2 and 3). Interestingly, an analysis of the complexity ratings indicated similar differences among reef tracts. The middle and offshore reefs did not differ from each other ($p > 0.05$) but both were significantly greater than the inshore tract.

Differences were also found based on the site location on the reef. With all data from the three reef tracts combined, the eastern edge showed significantly fewer total fish ($p < 0.001$) than either the crest of the reef or the western edge, which did not differ significantly ($p > 0.05$) (Figure 4). Species richness also varied with the western edge of the tracts having significantly more species ($p < 0.01$) than the crest or the eastern edges, again no significant difference was apparent between these two ($p > 0.05$) (Figure 5). Site complexity also mirrored this pattern with the eastern edge having the lowest complexity ($p<0.05$).

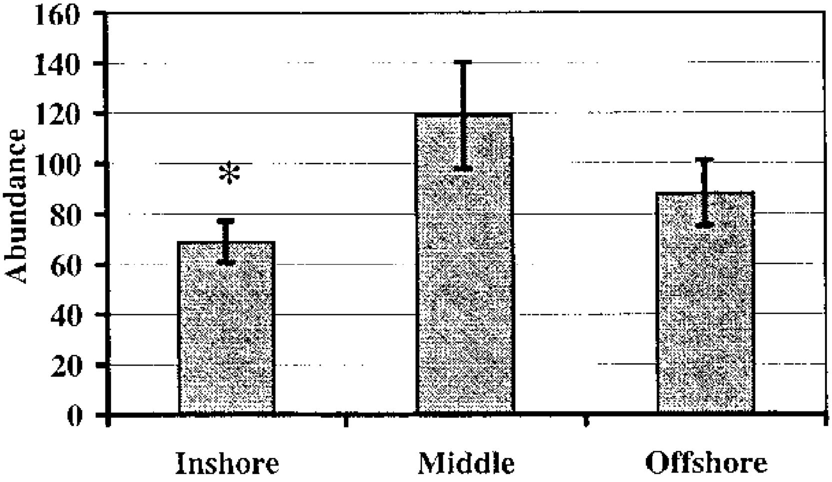


Figure 2. Mean (± 1 SEM) Fish abundance on each of the three reef tracts. *Indicates significantly different from non-asterisked tracts ($p < 0.05$, SNK)

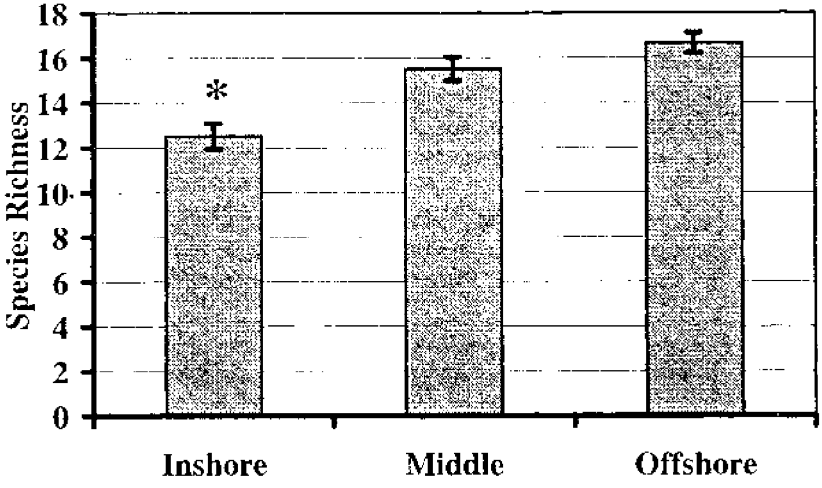


Figure 3. Mean (± 1 SEM) Species richness on each of the three reef tracts. *Indicates significantly different from non-asterisked tracts (p < 0.05, SNK)

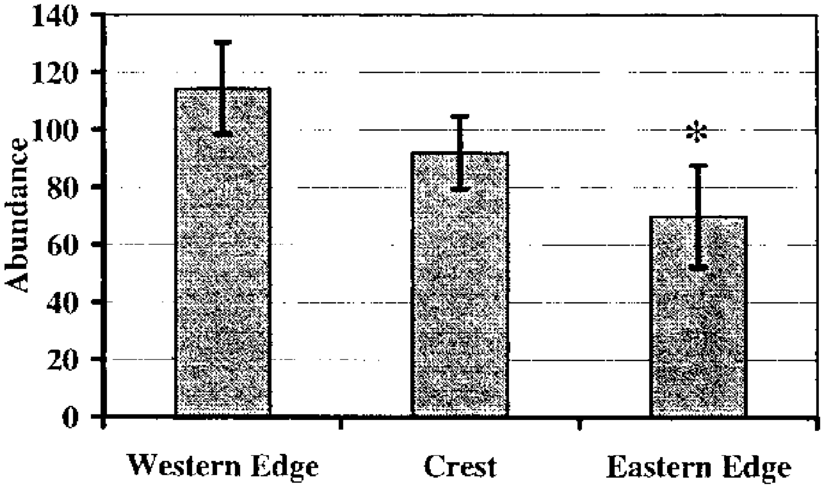


Figure 4. Mean (± 1 SEM) Fish abundance by reef site. *Indicates significantly different from non-asterisked edges (p < 0.05, SNK)

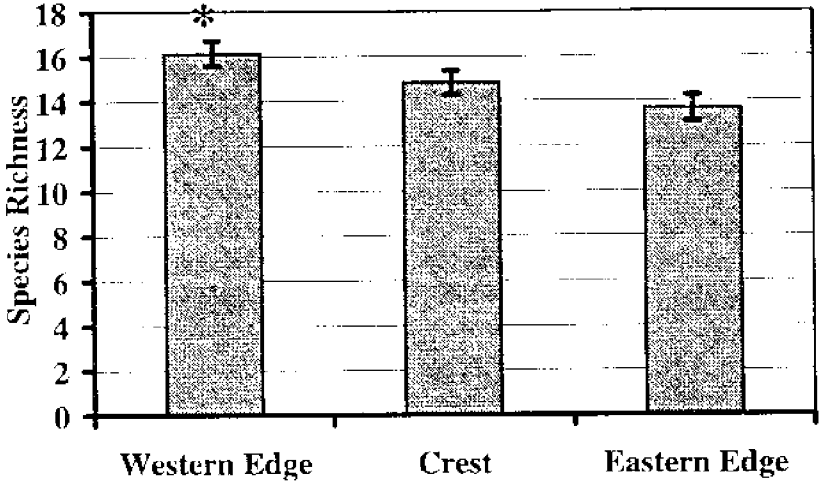


Figure 5. Mean (± 1 SEM) Species richness by reef site. *Indicates significantly different from non-asterisked edges ($p < 0.05$, SNK)

DISCUSSION

Results of this study indicate an overall difference in abundance and species richness among the three reef tracts with both fewer species and total fish on the inshore reef as compared to both the middle and offshore reef tracts. The study was intended mainly to inventory a portion of the Broward County reef tracts. Nonetheless, the differences in fish assemblages among the three tracts warrants some discussion. It is not surprising that we find differing species associated with three reef tracts at differing depths. Most marine fishes appear to have a depth preference, or at least a range in which they are routinely found (Robins et al. 1986, Humann 1994, Froese 1997). Of interest, both from ecological as well as management standpoints, are the biotic and abiotic factors which are involved in forming and maintaining the associations of fish with specific areas of a reef.

Several previous studies have examined the coral reef fish assemblages on inshore and offshore reefs. None of these studies are readily comparable to this study as they differ substantially in water depth (Thompson et al. 1990, McGehee 1994, Chabanet et al. 1997), distances between reefs (Williams 1982), or sampled a variety of habitats (Galzin and Legendre 1987). However, in each case the authors also found differences amongst sites inshore to offshore. These authors ascribed the difference between sites to a host of potential abiotic and biotic variables. Physical factors deemed to be important included: current, wave exposure, shelter, sediment loads, water depth, substrate, topographical complexity; biological factors included predation, larval recruitment, coral and other benthic communities. Certainly, many of these variables may have played a role in establishing the differences noted in our study. Water depths differed among the three tracts, and although we did not take objective data, it is clear from our diving experience there is substantial differences in sediment load and current between the sites (Sherman et al. 1999). Likewise, there were differences in larval recruitment and resident predators between sites (Gilliam 1999, Sherman et al. unpublished data). Although we recorded data on the bottom substrate, both living (i.e. corals, sponges, algae) as well as non-living (i.e. sand, hard bottom, coral rubble), we have not statistically examined any correlations between substrate or benthic communities and fish assemblages. There has been considerable research on the linkage between topographical complexity or refuge and fish assemblages. There are exceptions, but in general, increased complexity has been correlated with increased species richness as well as increased fish abundance (for references see: Spieler et al. in press). The similarities in our results between the areas of greatest complexity and the areas of greatest species richness and fish abundance support a role for topographical complexity in determining the difference noted between the inshore reef tract and the middle and offshore tracts. A more complete determination of the causal agents involved awaits further research.

In conclusion, this study is part of an ongoing survey of the fishes of Broward County. Upon completion, it will provide baseline data for determining changes in local populations of fishes and appropriate management strategies. In addition, because the census data includes substantial habitat information, we anticipate the survey will generate a host of hypotheses regarding the interaction of coral reef fishes with their physical and biotic environment. This information may ultimately prove essential to understanding and managing critical fish habitat.

ACKNOWLEDGMENTS

We thank Jim Bohnsack for instruction in Point-Count methodology and advice during the course of this study. Doug Harper provided hours of essential aid in helping us with the RVC software and moving the data through Excel and into SAS , it is unlikely the study would have been completed without him. We also thank the many NSU graduate students who, as unpaid volunteers, served as safety divers. This research was funded, in part, by NOAA/National Marine Fisheries Service, Southeast Fisheries Science Center, South Florida Ecosystem Restoration Research Program.

LITERATURE CITED

Bohnsack, J.A. and S.P. Bannerot. 1986. A stationary visual census technique for quantitatively assessing community structure of coral reef fishes. U.S. Dept. of Commerce, NOAA Technical Report NMFS **41**:1-15.

Chabanet, P., H. Ralambondrainy, M. Amanieu, G. Faure and R. Galzin. 1997. *Coral Reefs*. **16**:93-102.

Froese, R. 1997. Fishbase, a database with key information on coral reef fish. *Proc. 8th Int. Coral Reef Sym.* **2**:1545-1548.

Galzin, R. and P. Legendre. 1987. The fish communities of a coral reef transect. *Pacific Science* **41**:158-165.

Gilliam, D.S. 1999. *Juvenile fish recruitment processes in south Florida: a multifactorial field experiment.* Ph.D. Dissertation. Nova Southeastern University, Dania, FL. 111 pp.

Goldberg, W. M. 1973. The ecology of the coral-octocoral communities off the southeast Florida coast: geomorphology, species composition, and zonation. *Bull. Mar. Sci.* **23**:467-488.

Humann, P. 1994. *Reef Fish Identification*. New World Pub. Inc. Jacksonville, FL. 406 pp.

McGehee, M.A. 1994. Correspondence between assemblages of coral reef fishes and gradients of water motion, depth, and substrate size off Puerto Rico. *Mar. Ecol. Prog. Ser.* **105**:243-255.

Robins, R.C. and G.C. Ray. 1986. *A Field Guide To Atlantic Coast Fishes Of North America.* Houghton Mufflin Co. Boston. 354 pp.

Sherman, R.L., D.S. Gilliam and R.E. Spieler. 1999. A preliminary examination of depth associated spatial variation in fish assemblages on small artificial reefs. *Jour. Of Appl. Ichth.* **15**:116-121.

Spieler, R.E., D.S. Gilliam and R.L. Sherman. Artificial substrate and coral reef restoration: what do we need to know to know what we need. *Bull. Mar. Sci.* In Press.

Thompson, T.E., D.G. Lindquist, I.E. Clavijo, S.K. Bolden, S.W. Burk and N.C. Drayton. 1990. Assessment of reef fishes at Sombrero Key, Florida. *Proc. AAUS 10th Ann. Sci. Diving Sym.* 375-381.

Weinberger, L. 1998. *RVC User's Guide, Reef Fish Visual Census Version 1.* National Marine Fisheries Service, Miami, FL. 414 pp.

Williams, D. McB. 1982 Patterns in the distribution of fish communities across the central Great Barrier Reef. *Coral Reefs.* **1**:35-43.

The Effects of Natural Foods on the Growth and Development Of Queen Conch Larvae (*Strombus gigas*)

MEGAN DAVIS
Harbor Branch Oceanographic Institution, Inc.
5600 US 1 North
Ft. Pierce, Florida 34946 USA

ABSTRACT

Phytoplankton biomass and quality caused variation in the larval life span of queen conch, *Strombus gigas*. Conch larvae were grown in the laboratory on two sources of natural seawater, Great Bahama Bank and Exuma Sound, and on cultured algae, *Caicos Isochrysis* and *Chaetoceros gracilis*. Larvae were also fed a continuous supply of natural phytoplankton from Bahamian waters in a field-enclosed mesocosm system. Metamorphic competence was achieved in the shortest time (day 16) in the mesocosm system. In the laboratory larvae were competent by day 26 in the Bank water and day 20 in the cultured algal treatments. Veligers fed Sound waters did not achieve metamorphic competence. Food source was best characterized by cell counts and composition, than Chl *a* concentrations. Bank and Sound waters had similar Chl *a* levels, however, Bank water had larger cells, more taxa and higher cell count. Natural foods can be used as an alternative food source for culturing *S. gigas* larvae. Results from feeding studies using natural foods can also be used to predict dispersal potential in the field.

KEY WORDS: Growth, larvae, mesocosm, natural foods, phytoplankton, *Strombus gigas*

INTRODUCTION

The large gastropod *Strombus gigas* (queen conch) inhabits the shallow seagrass beds of Florida and the Caribbean region (Randall 1964). Since the 1970's the queen conch has been an important fisheries species in this region. The recent landings for conch are estimated at 6,000 MT with a value of 60 million dollars US (Chakalall and Cochrane 1997). Overfishing of this species has necessitated the need for management regulations in many countries (Appeldoorn 1994, Chakalall and Cochrane 1997). As of 1992 queen conch was added to Appendix II under CITES (Convention for the International Trade of Endangered Species). Now countries that export conch need permit approval from CITES management to ensure that the species is harvested at a level consistent with its fisheries population. However, management practices may not be enough to bring back overfished stocks or maintain fisheries populations over the long term.

Methodologies to culture queen conch larvae to the juvenile stage for stock enhancement and for growout markets were established by several organizations between the 1970s and 1990s (Creswell 1994, Davis 2000). Typical culture methods start with collection of egg masses from the field. Larvae (veligers) are cultured for approximately 21 days and fed cultured phytoplankton. Juvenile conch are fed an artificial pellet diet and are grown in sand trays and shallow ponds. Final growout of conch is accomplished in field enclosures.

In this field and laboratory study veligers of *Strombus gigas* were fed natural foods from two adjacent tropical oligotrophic waters in the Bahamas (Great Bahama Bank and Exuma Sound) and cultured phytoplankton to determine how differences in food quantity and quality affect larval growth and survival to metamorphic competence. The results from this investigation present alternative larval food sources and systems for culturing veligers of *S. gigas*. The data can also be integrated with oceanographic processes to estimate dispersal potential and supply of larvae to settlement sites.

MATERIALS AND METHODS

Field and laboratory studies were conducted at the Caribbean Marine Research Center (CMRC) field station on Lee Stocking Island (LSI), Exuma Cays, Bahamas from July to August 1994 and from June to July 1995. One egg mass was collected each experimental year from an offshore reproductive site located at a depth of 18 m on the island shelf approximately 1 km east of LSI, in the Exuma Sound. The newly laid egg mass was incubated in the laboratory for 4 days in a flow-through system (Davis 1994a). On the day of hatching several strands of the egg mass were placed in individual 8 L plexiglass vessels. The following morning (day = 0) the newly hatched veligers were used for the field and laboratory studies to test for the effects of different food types on growth of conch veligers.

Field Studies

The *in-situ* flow-through mesocosm system was moored in 3-m deep water near Lee Stocking Island. The system was comprised of a 3.6 m diameter hexagon shaped platform that supported 6 mesocosms, solar array, batteries and pumping system (Davis et al. 1996). Each mesocosm was a 1.8 m long x 0.5 m diameter transparent fiberglass cylinder with a 45° conical bottom. The 200 L mesocosm was submerged except for the top 0.5 m to prevent sample loss and wash-over by waves. To avoid contamination, each mesocosm was covered with a transparent fiberglass top. A solar charged battery powered a 12-volt bilge pump, which provided a continuous flow of ambient water (1 L/min, 8 exchanges of water per day), to each mesocosm. The water was pumped from 1 m below the system. To retain the larvae, each cylinder was equipped with a porous standpipe. Seawater was filtered through 50 μm bags, which allowed phytoplankton to enter the mesocosms, but blocked potential predators and competitors. These bag filters were changed daily.

Newly hatched larvae were stocked into 2 mesocosms at a concentration of 20 veligers/L. Every four days density was reduced by counting veligers out of one mesocosm into a newly-filled mesocosm. Final density at the end of the experiment was 0.7 veligers/L. Larvae were grown until <50% of the veligers showed signs of metamorphic competence (Davis 1994b). These signs include dark green pigmentation on the propodium. The temperature fluctuated during the day according to tide and ranged from 28 to 30 °C. To determine growth and development rates of veligers, shell lengths of 15 veligers were made every other day using a dissecting microscope (40X) equipped with an ocular micrometer. Replicate water samples (800 ml) from the mesocosm were filtered every other day to determine Chl *a* concentration (ng/L). Extraction of Chl *a* and fluorometric readings were performed according to standard methods (Strickland and Parsons 1972).

Laboratory Studies

In the 1994 laboratory study, veligers were fed phytoplankton assemblages from three different sources:

i) natural foods from Great Bahama Bank water (Bank),
ii) natural foods from Exuma Sound water (Sound), and
iii) an algal mixture of cultured Caicos *Isochrysis* and *Chaetoceros gracilis*.

In 1995, veligers were only fed natural foods from Bank and Sound water. Shallow Bank water was collected on ebb tide and Sound water was collected from the shelf on the flood tide (Figure 1). The temperature for the Sound water was 27.5 ± 0.8 °C and for Bank water 29.0 ± 1.3 °C. The seawater was filtered through a 250 μm mesh sieve to exclude most of the potential predators, while conserving food cells in the culture filtrate.

The cultured algae, Caicos *Isochrysis* and *Chaetoceros gracilis*, fed to larvae in this study are typical foods used in conch larviculture (Davis 1994a). The algae were grown in 250 ml flasks using Guillard's (1975) methodology and fed to achieve a final density of 5000 to 10,000 cells/ml. To avoid introducing natural phytoplankton into cultured algal treatments, all water was 0.45 μm filtered prior to adding algae. When veligers were cultured in only filtered seawater they did not survive past day 8.

The newly hatched veligers were stocked at 12 veligers/L in 800 ml transparent polypropylene containers. On day 10 the concentration was reduced to 5 veligers/L. There were five replicates per treatment for the 1994 study and four replicates per treatment for the 1995 study. To maintain densities additional veligers were cultured under the same treatment conditions to replace animals that died. To keep track of mortality, the number of veligers that died was recorded on each measuring date. When all the veligers died, the replicate was discarded. The treatments were run until veligers showed signs of metamorphic competence, halted development, or died.

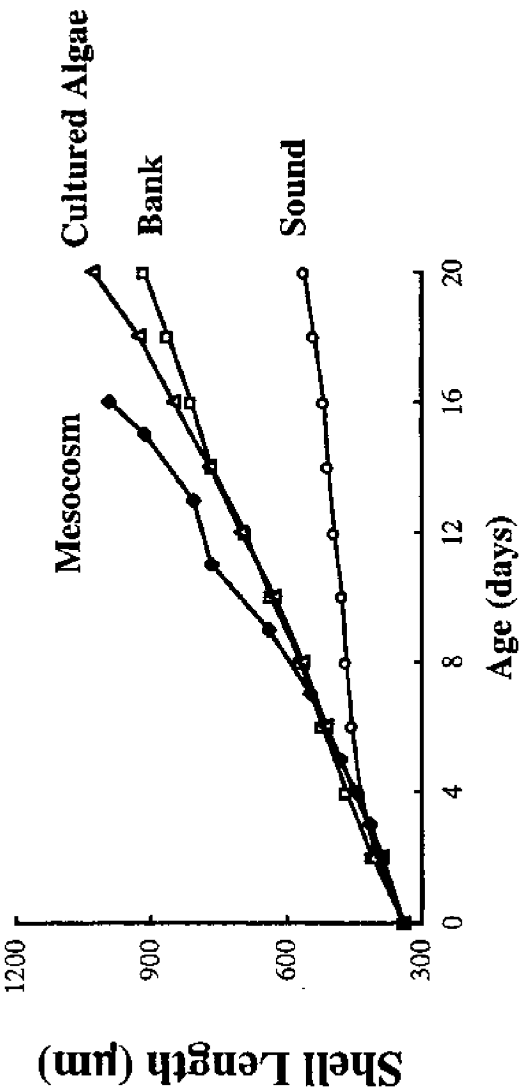


Figure 1. Growth curves for veligers of *S. gigas* fed natural foods (Bank, Sound, and mesocosm waters) and cultured algae (*Isochrysis* and *Chaetoceros*).

Veligers were cultured in an environmentally controlled incubator (28-29 °C, 38 ‰, 12 hr light: 12 hr dark). Water in the culture vessels was static and changed daily by pouring the veligers and water through a submerged sieve (180 μm mesh). A wash bottle with 0.45 μm filtered seawater was used to move the veligers from the sieve into the treatment water and culture vessel.

To determine growth rates of veligers in each treatment, shell length of all veligers in each replicate was measured from the apex to the siphonal canal every other day using a dissecting microscope (40X) equipped with an ocular micrometer.

Replicate water samples (800 ml) from each treatment were collected every other day to determine Chl *a* levels. To identify and count natural food cells, 15 L from each water mass (Bank and Sound) was concentrated to 1 L using the tangential filtration system (Davis 1998). Water was collected four times from the Bank and three times from the Sound over a one week period (20-26 June 1995) during the experiment. Five samples were taken from the concentrate and cells were counted and observed using a 4 chamber hemacytometer and a compound microscope (400 - 1000 X).

RESULTS

In 1994, when veligers were fed Bank water or high concentrations of cultured algae (7,939 cells/ml ± 1,211) they grew at similar rates from day 6-18, but on day 20 shell length (SL) was statistically smaller for veligers fed Bank water ($p < 0.05$) (Figure 1). On day 20, 35% of the veligers fed Bank water were metamorphically competent and shell length (SL) was 915 ± 62 μm. For veligers fed cultured algae, 80% were competent and SL was 1030 ± 28 μm. None of the veligers fed Sound water became competent by day 20. Development was arrested and they only attained 560 ± 21 μm SL (Figure 1). Veligers in the mesocosm system grew faster than their laboratory counterparts. Metamorphosis for veligers fed 50 μm filtered phytoplankton was first seen on day 13. By day 16, 95% of the veligers were competent or had completed metamorphosis with an average size of 933 ± 51 μm SL.

Although veligers fed cultured algae grew at rates (35 μm/day) equal to or slightly faster than veligers fed Bank water (29 μm/day), the Chl *a* concentrations were remarkably different. Average concentration was 4,700 ng/L for cultured algae and 176 ng/L for the Bank water, a 25-fold difference. The Chl *a* concentration in the mesocosm was 160 ng/L that was not statistically different than Bank water ($p > 0.05$). However, growth rates were faster for veligers grown in the mesocosm (41 μm/day) than on Bank water (29 μm/day). Chl *a* concentrations were statistically higher for Bank water than Sound water (107 ng/L) ($p < 0.05$). However, this Chl *a* difference (69 ng/L) was not large, and in 1995 studies ambient Chl *a* concentrations for Bank (147 ng/L) and Sound

(109 ng/L) water were not significantly different ($p > 0.05$). Even though Chl *a* concentrations were similar or slightly higher for Bank water, the average growth rates for veligers were considerably faster in Bank water than Sound water in both 1994 (29 µm/day vs 11 µm/day) and 1995 (26 $\mu m^{/day}$ vs 7 µm/day) (Figure 2).

The digestive glands of veligers fed Sound water contained a minimal amount of food and were clear to pale in color. These veligers developed 6 velar lobes; however, these lobes were stunted, deformed and wider than lobes of veligers fed Bank water. Lowest survival (62%) occurred for veligers fed ambient Sound water. Veligers cultured with Bank water, cultured algae or mesocosm water had golden brown guts full of algae, were very active, and developed 6 elongated lobes. Overall survival for these cultures was 90%, ~100%, and 73%, respectively.

Cell counts and composition of natural foods may partially explain growth differences for veligers fed natural foods from Bank and Sound water. The average total cell count for Bank water was 725 cells/ml and was statistically higher (3 times) than Sound water ($p < 0.05$), which was 234 cells/ml. The Bank and Sound water had similar dominant species and overall composition. However, the higher cell count in Bank water was attributed to the cell count in the class Bacillariophyceae and presence of cells in the class Euglenophyceae. In both water sources the dominant class was Cyanophyceae, followed by Haptophyceae or green flagellates, then colorless microflagellates. Even though the dominant classes were similar in the two water sources, the size and number of taxa differed. The Bank water contained four size categories of cells: microplankton (50 - 500 µm), nannoplankton (5 - 50 µm), ultraplankton (2 - 5 µm) and picoplankton (< 2 µm), whereas the Sound water had only the 3 smaller size categories. The Bank water also had 3.3 times more taxa and more cells in most size categories than Sound water.

During the week of 20-26 June 1995, there were fluctuations in variety and number of cells found in the water masses, especially in Bank water. Cell counts in Sound water ranged from 117 - 359 cells/ml per day and Bank water ranged from 467 - 1107 cell/ml per day. Even though fluctuations in cell count occurred on a daily basis, the growth curves for veligers fed natural foods from Bank and Sound water were similar during two years (Figure 2).

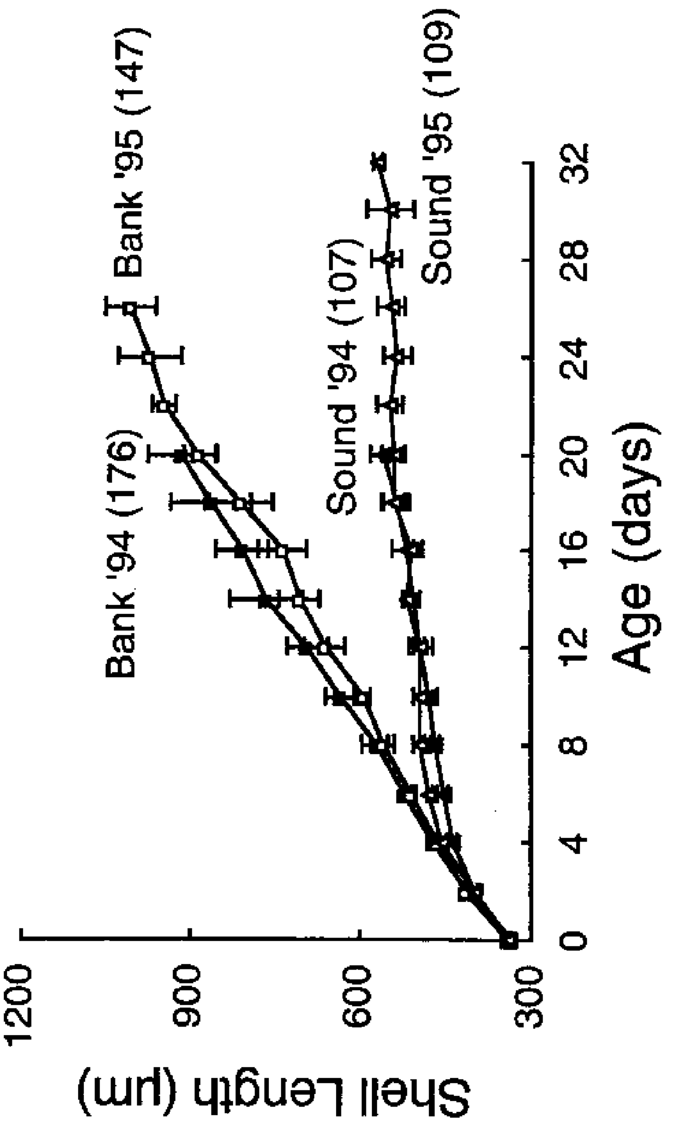


Figure 2. Comparison of growth curves for veligers of *S. gigas* fed natural foods from Bank and Sound water sources in August 1994 and June-July 1995. Data points represent mean ± sd (n = 5 replicates in 1994 and 4 replicates in 1995). Numbers in parentheses are the average Chl *a* concentrations.

DISCUSSION

Food quality and quantity are important conditions that influence length of larval life (Scheltema 1986). These are significant environmental factors from both an aquaculture and ecological viewpoint. Ideal food conditions in aquaculture promote high growth and survival rates. Traditionally, a wide range of laboratory cultured algae have been used to raise *S. gigas* larvae in research and commercial operations (Aldana-Aranda and Suarez 1998). The results from this study show that veligers of *S. gigas* can be cultured on certain types of phytoplankton assemblages collected in the waters where veligers disperse naturally. Veliger growth and time to metamorphic competence were good parameters to evaluate the nutritional quality of water.

In the laboratory, growth rates for veligers fed natural foods from Bank water were as rapid as for veligers fed cultured algae, a finding also reported by Boidron-Metairon (1992). However, to achieve this similar growth rate, veligers fed cultured algae required 11 times the number of cells and 25 times the Chl *a* level of natural foods. The mesocosm system proved to be a successful culturing unit to raise veligers to metamorphosis. Growth rates in the mesocosm were faster (16 days to metamorphosis) than in other treatments (cultured algae: 20 days, Bank: 26 days), due to continuous flow of natural foods. These results suggest that natural foods contain nutritionally superior cells compared to cells cultured in the laboratory. These findings not only suggest that natural foods can be used to raise veligers, but also underline the necessity of using natural foods for nutritional studies when field predictions will be extrapolated from the data.

Chl *a* concentrations were used as a relative index of the food availability and the quality of each water mass. However, the use of this water quality parameter proved to be misleading, corroborating the finding of Olson et al. (1987). In 1995 the ambient Chl *a* levels for Bank (147 ng/L) and Sound (109 ng/L) water were not statistically different, but growth and survival rates were considerably higher in Bank water. Veligers grown in Sound water proved to be a severely food limited and did not achieved metamorphosis. Cell composition and density may be better parameters to evaluate food quality. In this study high growth and survival rates for veligers fed Bank water may be attributed to at least three factors: larger cells, more diverse taxa (3.3 times more), and higher cell count than Sound water.

Growth rates for veligers fed natural foods from Bank water and from Sound water were similar during the peak months of two spawning seasons (June to August 1994 and 1995). The nutritional stability of these waters may be due to lack of long term or seasonal blooms which is typical of tropical, low latitude regions where phytoplankton density is low due to the lack of a constant source of nutrients (Raymont 1980, Harris 1986, Howarth 1988). It is possible that overall growth and time to metamorphic competence maybe predictable from year to year in the peak months of the spawning season when veligers feed from the same water mass. To fully understand the relationship between growth of veligers and seasonal and short-term changes in food abundance and composition, nutritional experiments need to be conducted over the entire culturing and spawning season.

From an ecological viewpoint, food conditions in the oligotrophic waters of the Bahamas can be limiting or promote high growth and survival rates for veligers of S. gigas. Veligers suffered slow growth rates, arrested development and had low survival when fed natural foods from the Sound, whereas they grew at high rates to metamorphic competence when fed natural foods from shallow, nearshore Bank water. In the field it is likely that veligers encounter different feeding environments due to oceanographic circulation patterns and larval behavior. These differences will influence dispersal potential and settlement patterns for this important pan-Caribbean species.

ACKNOWLEDGEMENTS

This research was supported by a grant from the National Undersea Research Program, NOAA (U.S. Department of Commerce) to the Caribbean Marine Research Center. Dr. Allan Stoner provided support and guidance throughout the studies. Special thanks to Gary Hodgkins who built the mesocosm system. Tammy Hall, Linda Hambrick, Richard Jones, Nikhil Mehta, Sheila O'Connell, and Ed Wishinski assisted in the laboratory and field studies. Harbor Branch Oceanographic Institution contribution # 1410.

LITERATURE CITED

Aldana-Aranda, D. and V.P. Suarez. 1998. Overview of diets used in larviculture of three Caribbean conchs: queen conch *Strombus gigas*, milk conch *Strombus costatus* and fighting conch *Strombus pugilis*. *Aquaculture* **167**:163-178.

Appeldoorn, R.S. 1994. Queen conch management and research: Status, needs and priorities. Pages 301-319 in: Appeldoorn, R.S., Rodriguez, B. (Eds.), *Queen Conch Biology, Fisheries and Mariculture*. Fundacion Cientifica Los Roques, Caracas, Venezuela.

Boidron-Metairon, I.F. 1992. A new approach to comparative studies of *Strombus gigas* larvae at the developmental and nutritional levels. *Proc. Gulf Carib. Fish. Inst.* **41**:459-467.

Chakallal, R., Cochrane, K. 1997. The queen conch fishery in the Caribbean: An approach to responsible fisheries management. *Proc. Gulf Carib. Fish. Inst.* **49**:531-554.

Creswell, R.L. 1994. An historical overview of queen conch mariculture. Pages

223-230 in: Appeldoorn, R.S., Rodriguez, B. (Eds.), *Queen Conch Biology, Fisheries and Mariculture*. Fundacion Cientifica Los Roques, Caracas, Venezuela.

Davis, M. 2000. Queen conch (*Strombus gigas*) culture techniques for research, stock enhancement and growout markets. Pages 127-159 in: Fingerman, M., Nagabhushanam, R. (Eds.), *Recent Advances in Marine Biotechnology, Vol 4, Aquaculture*, Science Publishers, Inc. New Hampshire USA.

Davis, M. 1998. *The effects of natural foods, temperature and salinity on the length of larval life for the tropical gastropod Strombus gigas*. Ph.D. Dissertation. Florida Institute of Technology, Melbourne, Florida. 136 pp.

Davis, M. 1994a. Mariculture techniques for queen conch (*Strombus gigas* Linne): Egg mass to juvenile stage. Pages 231-252 in: Appeldoorn, R.S., Rodriguez, B., (Eds.), *Queen conch biology, fisheries, and mariculture*. Caracas, Venezuela: Fundacion Cientifica Los Roques.

Davis, M. 1994b. Short-term competence in larvae of queen conch *Strombus gigas*: Shifts in behavior, morphology and metamorphic response. *Mar. Ecol. Prog. Ser.* **104**:101-108.

Davis, M., G.A. Hodgkins, and A.W. Stoner. 1996. A mesocosm system for ecological research with marine invertebrate larvae. *Mar. Ecol. Prog. Ser.* **130**:97-104.

Guillard, R.L. 1975. Culture of phytoplankton for feeding marine invertebrates. Pages 29-60 in: Smith, W. L., Chanley, M.H. (Eds.), *Culture of Marine Invertebrate Animals*. New York, USA: Plenum Press.

Harris, G.P. 1986. *Phytoplankton Ecology: Structure, Function and Fluctuation*. Great Britain, Cambridge: University Press, 225-226.

Howarth, R.W. 1988. Nutrient limitation of net primary production in marine ecosystems. *Ann. Rev. Ecol.* **19**:89-110.

Olson, R.R., I., Bosch, and J.S. Pearse. 1987. The hypothesis of Antarctic larval starvation examined for the asteroid Odontaster validus. *Limnol. Oceanogr.* **32**:686-690.

Randall, J.E. 1964. Contributions to the biology of the "queen conch" *Strombus gigas*. *Bull. Mar. Sci. Gulf Carib.* **14**:246-295.

Raymont, J. E. G. 1980. *Plankton and Productivity in the Oceans*. Pergammon Press. 489 pp.

Scheltema, R.S. 1986. On dispersal and planktonic larvae of benthic invertebrates: an eclectic overview and summary of problems. *Bull. Mar. Sci.* **39**:290-322.

Strickland, J.D.H. and T.R. Parsons. 1972. *A practical Handbook of seawater analysis, 2nd edition*. Ottawa: Fisheries Research Board of Canada.