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The Arid Mangrove Forest from Baja California Peninsula

VOLUME 1

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**THE ARID MANGROVE
FOREST FROM BAJA
CALIFORNIA PENINSULA**

VOLUME 1

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ENVIRONMENTAL RESEARCH ADVANCES

**THE ARID MANGROVE
FOREST FROM BAJA
CALIFORNIA PENINSULA**

VOLUME 1

**RAFAEL RIOSMENA RODRIGUEZ
ADRIAN F. GONZÁLEZ-ACOSTA
AND
RAQUEL MUÑIZ-SALAZAR
EDITORS**



New York

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PREFACE

The arid mangrove forest from the Baja California Peninsula represents a series of chapters devoted to the understanding of the biodiversity and fisheries value of this critical habitat. The book contents are devoted to making the reader aware of the great value of the mangrove in relation to their conservation and management. Mangrove forests in Mexico represent the northernmost populations along Pacific coast, and thus they are likely to be sourced populations for colonization at higher latitudes as the climate becomes more favorable. Today, these populations are relatively small and fragmented and prior research has indicated that they are poor in genetic diversity, but in the actual global warming scenario, it is very likely that forest cover will increase and genetic diversity will increase. Another relevant component of the book will be the biodiversity inventory still in progress for many biological groups and will be taken into account in the present volume. The present volume will review and analyze some physical, biological, and ecological factors that determine the success of these communities and their use in conservation or fisheries management. Marine Protected Areas include several of the mangroves studied here and the presented information is critical for the appropriate management. The book is organized in 10 chapters to evaluate studies based on the value of the forest in arid zones of the Baja California Peninsula and the surrounding areas worldwide. This book will be essential to understand the mangroves along Northwestern Mexico and will also be a critical consulting book for the next generation of managers and nature lovers who want to better understand our environment.

Chapter 1 - The flower and hypocotyls development of mangrove forest on the eastern Baja California Peninsula were analyzed, comparing four micro-topographic flooding conditions (coast, channel, permanent and temporary flood) and in relation to five environmental factors (relative humidity, rainfall, ambient temperature, solar radiation and wind speed). The main objective was to evaluate the effects of an arid environment on the reproductive effort of the mangrove species present. Between June 1 and November 5 of 2006, weekly visits were made to three sites in La Paz bay. During each visit, the authors registered the flower and hypocotyls development of three selected stems of one tree per species under the four micro-topographic conditions. The authors' results indicate that nearly 25% of all flowering plants passed to hypocotyls and the relative humidity and rainfall were the variables most associated with flower development. Also, significant differences in flower and hypocotyls development were found for all studied species in relation to micro-topographic conditions. The authors conclude that humidity triggers the reproductive effort of

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mangroves species in this arid region and that the microhabitat coast is the best condition to promote mangrove reproduction on the eastern Baja California Peninsula for restoration purposes. Impacts on other micro-topographic conditions may lead to loss of forest.

Chapter 2 - Mangrove forests in the Gulf of California, Mexico represent the northernmost populations along the Pacific coast. Today, these populations are relatively small, fragmented and poor in genetic diversity. The authors evaluated the population genetic structure and recent evolution of the red mangrove, *Rhizophora mangle* and the black mangrove, *Avicennia germinans*, along the northwestern coast of Mexico. Genetic diversity of both mangrove species showed a decreasing trend toward northern latitudes along the Pacific coast. The lowest levels of genetic diversity were found at the range limits around the Gulf of California and the outer Baja California peninsula. Lack of a strong spatial genetic structure in this area and recent northern gene flow in *A. germinans* suggest recent colonization of this species. On the other hand, lack of a signal of recent northern dispersal in *R. mangle*, despite the higher dispersal capability of this species, indicates a longer presence of populations, at least in the southern Gulf of California. The authors suggest that the longer history, together with higher genetic diversity of *R. mangle* at the range limits, likely provides a gene pool better able to colonize northwards under climate change than *A. germinans*.

Chapter 3 - Mangrove ecosystems are a critical habitat for many species. The productivity of mangroves is important due its contribution of organic detritus and secondary production. The objective was to identify indicators of secondary productivity of mangrove systems to establish a baseline for measuring changes in the mangrove system due potential natural or anthropogenic disturbances. The bivalve mollusks, the black ark *Anadara tuberculosa* and the palmate oyster *Saccostrea palmula* were chosen as indicators of secondary productivity. Samples were collected in three mangroves of the sand barrier El Mogote in the Ensenada de La Paz, from August 2007 to July 2009. Abundance and secondary production were estimated based on biomass increase. The estimated weight increase was performed by determining individual growth rates through an analysis of length frequency distributions. The average abundance was 1.27 and 510 ind. m^{-2} of black ark and palmate oyster respectively. Growth rates of $4.91 \text{ mm month}^{-1}$ ($3.67 \text{ g month}^{-1}$) for black ark and $3.80 \text{ mm month}^{-1}$ ($0.18 \text{ g month}^{-1}$) for the palmate oyster were estimated. Average secondary production of black ark was $4.51 \text{ g m}^{-2} \text{ month}$ and was higher during the spring, while for the mangrove oyster was $97.9 \text{ g m}^{-2} \text{ month}^{-1}$, and the maximum production was recorded during the summer. The authors found 78 macroinvertebrate species within the mangrove swamps surveyed in this study. There are just a few species of bivalves associated with mangroves that represent an economic interest for their productivity. The black arks and oysters are among the most important mangrove-associated species because of their commercial value and continual exploitation. The obtained results, such as a quality index of benthos, are suitable for measuring changes to the mangrove system in the Ensenada de La Paz, Baja California Sur, Mexico.

Chapter 4 - Some reproductive features of *Saccostrea palmula* (Carpenter, 1857) and *Anadara tuberculosa* (Sowerby, 1833) from the Santo Domingo (SD) and El Conchalito (ECO) estuaries were investigated to determine differences related to environmental variability and similarities arising from sharing the same location. Both species display five stages of gonadal development (undifferentiated, development, ripe, spawning and spent), which occur synchronously in females and males.

In both locations, *S. palmula* showed an annual reproductive cycle while *A. tuberculosa* showed a continuous reproductive cycle. Sex ratios significantly departed ($p < 0.05$) from 1:1 for both species in ECO, but not for *S. palmula* in SD. For *S. palmula*, a significant correlation ($p < 0.05$) between mean maturity index (MMI) and temperature was found at both sites, and between MMI and chlorophyll-a concentration in ECO. For *A. tuberculosa* no significant correlation ($p > 0.05$) between MMI and either temperature or chlorophyll-a concentration was observed. *S. palmula* behaves as an opportunistic species in SD, but as a conservative species in ECO, as does *A. tuberculosa*. This seems to be related to chlorophyll-a concentration at each site.

Chapter 5 - An updated systematic and zoogeographical checklist of fishes associated to mangrove estuaries of Baja California Sur (BCS) is provided in the present chapter. This checklist is based on records of literature and of verified voucher specimens from museums. Fish diversity from mangrove estuaries of BCS is composed by 228 species, 147 genera, 72 families, 23 orders, and two classes. Fish fauna of BCS is dominated by the class Actinopterygii (208 spp.) and the order Perciformes (125 spp.) with four families (Haemulidae, Sciaenidae, Gobiidae and Carangidae) representing more than 50% of the fish species into the order. Biogeographically, the ichthyofauna showed a mixture of species from derivation of temperate and tropical regions, with high affinities to the Cortez province (83.7%) followed by Mexican (73%) and Panamic (70.3%) provinces; however, more than 61.2% of the species extend their geographic ranges into the San Diegan province. Other species (6.6%) have circumtropical (13 spp.), amphiamerican (1 sp.) and amphipacific (1 sp.) affinities; and four species are from exotic origin (*Poecilia butleri*, *P. reticulata*, *Sparus aurata*, and *Tilapia cf. zillii*). Three taxa endemic to the Gulf of California were also recorded (*Aruma histrio*, *Barbulifer pantherinus* and *Gobiosoma chiquita*). Bahía de la Paz (62.7%) and Bahía Magdalena (55.7%) were the most extensive mangrove ecosystems in BCS, both contributing with the highest number of species. High fish diversity reported here, reflects the important role of the mangrove estuary ecosystems as primary habitat or nursery (feeding and spawning) for many marine species, contributing with the growth, recruitment and conservation of species with commercial or ecological value.

Chapter 6 - The authors provided a taxonomic list of fishes caught in the mangrove-lagoon ecosystem of the Isla San José, Baja California Sur, México, based on samplings performed during October 2004 to June 2009. Fifty-three species belonging to 40 genera and 29 families were recorded, of which the most diverse families were Haemulidae (six species), Lutjanidae and Gerreidae (four species each). Biogeographically, the fish fauna of San José estuary has a higher affinity with the Cortez province (77.4%), followed by the Mexican (75.5%) and Panamic (71.7%) provinces. Moreover, almost 70% (37 spp.) of the fish species reported here have commercial value. The King Angel fish, *Holacanthus passer*, is subject to special protection by the Mexican laws (NOM-059-SEMARNAT-2010). The use of the mangrove ecosystem of Isla San José by juveniles and adults of the species caught by artisanal fisheries and those subject to conservation programs, revealed the importance of this estuary as a critical habitat or nursery ground for several fish species in the Gulf of California.

Chapter 7 - Seasonal variation in the composition and abundance of a guild of seven gerreid species: *Diapterus brevirostris* (Sauvage, 1879), *Eucinostomus currani* Zahuranec, 1980, *Eucinostomus entomelas* Zahuranec. 1980, *Eucinostomus gracilis* (Gill, 1862), *Eucinostomus dowii* (Gill, 1863), *Eugerres lineatus* (Humboldt, 1821), and *Gerres cinereus* (Walbaum, 1792) was examined in order to assess the effect of the hydrological dynamics of

El Conchalito estuary, a mangrove swamp ecosystem located in the southern Gulf of California. Monthly samples were collected from June 1996 to August 1997 using a stationary flume net deployed across the tidal channel. Densities were recorded for each species, as well as the standard length and weight of each individual. Hydrological parameters (water temperature, salinity, tidal level and dissolved oxygen) were measured *in situ* during each sampling event. The most abundant species in terms of density was *D. brevirostris* (62.6 %), while *E. entomelas* was the most prevalent species in terms of biomass (34.6%). A significant correlation was found between temperature and species abundance (including *E. curranii*). Seasonal environmental variability in the ecosystem prompts changes in the composition and abundance of gerreids; the high incidence of juveniles during summer and winter highlights the importance of this ecosystem as a nursery for fish fauna.

Chapter 8 - Mangroves are capable of absorbing or reorganizing stress effects to maintain their function, processes, and structure. Nevertheless, in spite of their ecosystem resilience and environmental adaptation, they have to face climate change risk. At world level, the forecast is an increase in CO₂, CH₄, and N₂O concentrations, as well as in temperature, a higher impact in runways, drought, ocean acidification, sea level rise, and hurricanes. However, vulnerability of mangrove systems is not only according to biophysical interactions with environmental risks but also to links among socio-economical structures, cultural norms, and social values.

Particularly, an accelerated speed of sea level rise will probably have an impact in vegetation loss toward the sea, an inland or a lateral shift that will depend on the contribution of hydrology, sediments, and anthropogenic modifications to the landscape. The vulnerability degree of the mangrove ecosystem will depend on the particular regional conditions. In this chapter the authors show case studies on two lagoon systems of the eastern Gulf of California coast where the authors identify and assess the principal environmental and socioeconomic factors that cause mangrove vegetation vulnerability.

The authors' analyzed changes in mangrove cover in the two lagoon complexes Guásimas-LosAlgodones-Lobos (GLL), Sonora and Navachiste-San Ignacio-Macapule (NSM), Sinaloa, using LANDSAT (1990 and 2003) satellite images and SPOT 5 (2009). Forest cover maps were generated with ERDAS Imagine and Arc/Gis 9.1 programs. The authors used unsupervised (isodata algorit) and supervised (classes) classification methods; after that, they could detect changes and used the Puyravud Deforestation Analysis Rate (2003) during the 1990-2003 and 2003-2009 periods.

As a result, the authors detected coverage changes in the GLL lagoon complex, a decrease in specific areas from 1990-2003 where the lost rate was positive (from 0.5 to 2.1). From 2003-2009 the main values were negative, with the highest rate of -0.22 where there was an important increase in areas with little anthropogenic influence, possibly related to the increase of rainfall in that period. Changes during the first period were due to precipitation increase and cover loss and to natural cycles more than to socioeconomic risks in the second period.

In the case of the NSM lagoon complex, the authors observed mangrove cover reduction in 1990, 2003, and 2009. Negative rates were higher from 2003-2009 (-0.22) even when the weather was less arid with important cyclonic events including a major contribution of rainfall. The socioeconomic vulnerability analysis showed affectation by agricultural activities and contaminants draining to the system.

Chapter 9 - San Ignacio Lagoon is located within the Vizcaíno Biosphere Reserve. There has been limited study of its ichthyofauna to date; the objective of this chapter is to describe the composition and distribution, as well as the temporal and spatial abundance of the ichthyofauna associated with soft bottoms in this coastal lagoon. A total of 44 trawls were carried out over a 346,500 m² area. A total of 11 trawls per season were carried out, covering an 86,625 m² area (7,875 m² per trawl). A total of 1,361 organisms were collected, with a total biomass of 67,118 g. The ichthyofauna of this lagoon was composed of 21 families, 35 genera, and 44 species. The highest abundance of organisms and biomass was obtained during the autumn (34.6% and 30.4%, respectively), while the lowest abundance and biomass were obtained in the spring (16.5% and 13.5%, respectively). Of the 44 recorded species, eight were present during the overall study period (*S. annulatus*, *S. lispus*, *E. dowii*, *P. maculatofasciatus*, *Urobatis halleri*, *Hypsopsetta guttulata*, *Hippocampus ingens*, and *Paralichthys californicus*). The Biological Value Index showed that 25 species were dominant in this ecosystem. The species diversity was highest in winter (2.28 bits/ind.) and lowest during the autumn (0.41 bits/ind.). Water temperature was similar in spring and autumn (21.1°C and 20.9°C, respectively), and was higher in the summer (24.9°C). The zoogeographic affinity analysis showed that the provinces with the highest number of records were the Panamic (36%) and San Diego (31.8%) Provinces.

Chapter 10 - Mangrove systems are of great importance worldwide for their high primary and secondary productivity because they sustain fish abundance and diversity. The aim of this chapter was to compare the current structure and composition (2009-2010) of the fish community in three mangrove systems with different degrees of anthropogenic influence in the Bay of La Paz, Baja California Sur, with estimates made 30 years ago (1980-1981). Ecological attributes were estimated (number and weight density, species richness, Shannon-Wiener diversity, Pielou evenness), average taxonomic distinctness (ATD) and the variation in taxonomic distinctness (VarTD) from data collected monthly during each period. A total of 39,123 fish were recorded in 1980-1981 and 16,740 fish in 2009-2010. During both periods, the species that dominated the density in the three systems were *Eucinostomus dowii*, *Diapterus brevirostris*, and *Mugil curema*. The most dramatic changes in the fish structure community was between the periods in the system that has had more human intervention. In contrast, no significant differences between periods were found in the less impacted system, suggesting on the one hand that the fish community is a good indicator of healthy systems, and on the other that local effects of human intervention outweigh the climatic influence regionally.

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Chapter 1

**EFFECTS OF THE ARID ENVIRONMENT ON
MANGROVE SPECIES REPRODUCTION ON THE
EASTERN BAJA CALIFORNIA PENINSULA, MÉXICO**

***R. Domínguez-Cadena^{1,2}, R. Riosmena-Rodríguez^{1,*}
and J. L. León-de la Luz²***

¹Programa de Investigación en Botánica Marina, Departamento de Biología Marina.
Universidad Autónoma de Baja California Sur,
La Paz, Baja California Sur, México

²Programa de Planeación Ambiental y Conservación, Centro de Investigaciones
Biológicas del Noroeste, SC.,
La Paz, Baja California Sur, México

ABSTRACT

The flower and hypocotyls development of mangrove forest on the eastern Baja California Peninsula were analyzed, comparing four micro-topographic flooding conditions (coast, channel, permanent and temporary flood) and in relation to five environmental factors (relative humidity, rainfall, ambient temperature, solar radiation and wind speed). The main objective was to evaluate the effects of an arid environment on the reproductive effort of the mangrove species present. Between June 1 and November 5 of 2006, weekly visits were made to three sites in La Paz bay. During each visit, we registered the flower and hypocotyls development of three selected stems of one tree per species under the four micro-topographic conditions. Our results indicate that nearly 25% of all flowering plants passed to hypocotyls and the relative humidity and rainfall were the variables most associated with flower development. Also, significant differences in flower and hypocotyls development were found for all studied species in relation to micro-topographic conditions. We conclude that humidity triggers the reproductive effort of mangroves species in this arid region and that the microhabitat coast is the best condition to promote mangrove reproduction on the eastern Baja California Peninsula for restoration purposes. Impacts on other micro-topographic conditions may lead to loss of forest.

* Corresponding author: riosmena@uabcs.mx.

INTRODUCTION

Mangrove forests are distributed in tropical coastal areas of the world (MacNae, 1968; Chapman, 1976; Tomlinson, 1986), with some species having their geographical limits in the subtropical zone (Tomlinson 1986; Ramírez-García and Lot, 1994; Dawes et al. 1999; Pacheco-Ruiz et al. 2006; Zomlefer et al. 2006). Mangrove species in tropical areas are known to reproduce in the rainy season, as shown by Fernandez (1999) for the Brazilian coast and by Jiménez (1988) for Costa Rica mangroves. However, Tovilla (1998) and Tovilla and Orihuela (2002) have shown that a heavy-rainfall environment influences flower and hypocotyls development in *Rhizophora mangle*, producing reproductive structures all year with a peak in production during the summer season. In other species, such as *Avicennia marina*, reproduction is restricted to the humid period (Duke, 1990) and flower production varies on a latitudinal gradient (Clarke and Myerscough 1991).

In semi-arid forest on the Caribbean coast of Colombia country, Ángel (2001) found that *Avicennia germinans* and *Laguncularia racemosa* flowering during the dry season, but not *R. mangle*, suggesting that some species are adapted for local conditions of arid environments. On the Baja California Peninsula, limited fresh water availability (Lot-Helgueras et al. 1975; Rico-Gray and Palacios-Ríos 1996; Tyagi 2004) might play a role in mangrove reproduction. Dominguez-Cadena et al. (2011) have shown the effect of micro-topographic structure on mangrove species distribution, suggesting the influence of historical differential recruitment as a driver of present mangrove forest structure. On this basis, our goal is to evaluate the effects of the arid environment at different micro-topographic conditions on the reproduction of three mangrove species. Our hypotheses are: a) the appearance of flowers and hypocotyls and the development of each species will vary according to micro-topographic conditions as a result of the spatial influence of water availability/salinity stress and b)the appearance and development of flowers and hypocotyls are related to seasonal environmental changes as a result of rain regime.

METHODS

Study Site

The study area is located in La Paz Bay, 24° 10' 19" North and 110° 19' 26" West, on the Eastern Baja California Peninsula (Figure 1). We selected three sites based on their accessibility and the presence of all micro-topographic conditions: Balandra Lagoon (Site 1), Pichilingue (Site 2) and Zacatecas Estuary (Site 3). Each site was characterized by four micro-topographic conditions related to tide level inundation, named as: coast (CO), channel (CA), permanent (IP) and temporary flood (IT) (Figure 1). García (1981) reported dry arid climate type BWh, the hot season occurs from July to October and includes some days over 40 °C in September, as registered by CONAGUA (2006). The cold season occurs from December to February and is characterized by anticyclone winds, commonly in January, reaching temperatures below 10 °C in sunshine hours (Merrifield and Winant 1989; Álvarez-Borrego 2004).

Sampling Design

Samples were collected from three sites in La Paz Bay according four microtopographic conditions as described by Domínguez-Cadena et al. (2011) (Figure 1). At each site, we selected three plots of 5 m × 5 m, as described by Dawes et al. (1999). In each plot, we selected one tree per species, no more than 3 m high. In each tree we marked tree stems to following flowers and hypocotyls development. We selected three inflorescence groups for records data as: flowers development started with buds, followed by the appearance of sepals, hypocotyls and endocarp, until the flower falls from the mother tree. For each species, we determined the percentage of flowers and hypocotyls by observation every seven days from June 1st and November 5th of 2006. The results were obtained as follows: % of change = \sum numbers of fruits (each/7days) * 100/ \sum Numbers of Flowers (each 7 days). Data were not normal or homoscedastic, therefore a nonparametric Mann-Whitney U test was used. A multiple correlation test (r) was applied to determine if the appearance of flowers and hypocotyls correlated with ambient conditions named as; relative humidity (HR), rainfall (Pp), solar radiation (S), ambient temperature (T) and wind velocity (V). These climatic values (Figure 2) were obtained of the automatic climatology station of Centro de Investigaciones Biológicas del Noroeste (CIBNOR, S.C.) localized at the near coast of south La Paz Bay.

RESULTS

Flower and Hypocotyls Record

The Figure 3, show the flowering and hypocotyls period from June 1 to October 27, were annotated a total of 162 flowers in *A. germinans* and *L. racemosa* species, while 243 flowers in *R. mangle*. On the other hand *A. germinans* produced more flowers and hypocotyls in CO and IT conditions, *L. racemosa* in CO and CA conditions, and *R. mangle* in CO, CA and IP conditions. We found a total flowers reproductive time of 91 days in *A. germinans*, and 50 days for hypocotyls development. The Figure 3, show that the flowering period reached its peak in the second week of July, and most hypocotyls were observed by the end of June. Both flowers and hypocotyls finished in mid-September. In the case of *L. racemosa* trees, massive flowers were observed from the second week of June to the first week of August. The reproductive period lasted 98 days, while hypocotyls appeared from the second until the fourth week in August and finished in October after 77 days. In the case of *R. mangle* reproductive events, flowers were observed 70 days earlier in CO than in other conditions. The hypocotyls were present for 120 days in CO conditions, 50 days in CA conditions, and 57 days in IP conditions. In first week of August, all hypocotyls were finished.

Flower and Hypocotyls Development

We found more flower and hypocotyls development in CO and IP for *R. mangle*, in CA for *L. racemosa*, and in IT for *A. germinans*. In each condition we calculated the average

change from flowers to hypocotyls, yielding 19.2% in CO, 24.2% in CA, 13.8% in IP, and 1.9% in IT. By species we found that *A. germinans* had 24% fewer flowers but 48% more hypocotyls development in CO compared with IT. For *L. racemosa*, we registered 2% more flowering and 4% less hypocotyls development in CO than in CA conditions. In *R. mangle*, there was 14% more flowering in CO than in CA conditions, and hypocotyls development was 54% more in the CO condition than the CA condition and 32% more in the CO condition than the IP condition. In the CA condition, hypocotyls development was 8% less than in the IP conditions.

We compared the flowering change process to hypocotyls development by significant difference (U estimate $<$ U constant; $p < 0.05$) and test in *R. mangle* showed only in two conditions, CO-CA and CO-IP. On the other hand, *L. racemosa* and *A. germinans* did not show statistical differences (U estimate $>$ U constant; $p > 0.05$) for the four micro-topographic conditions.

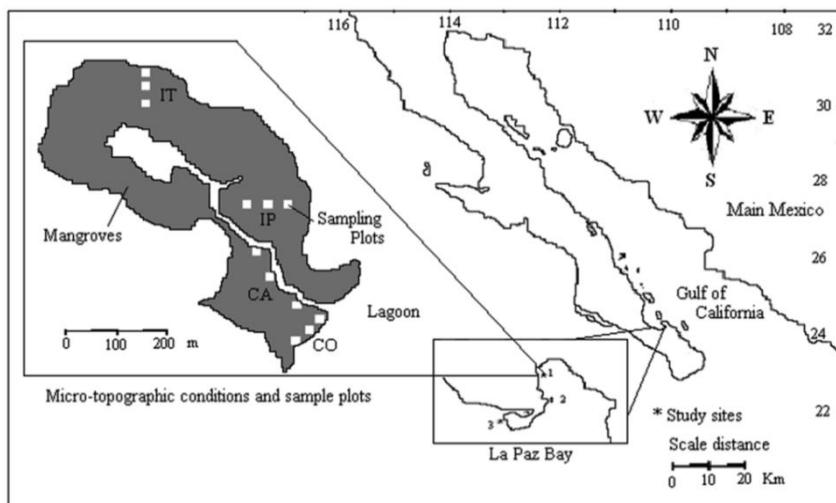


Figure 1. La Paz Bay locality where samples were taken from (1) Balandra, (2) Pichilingue, and (3) Zacatecas sites. The inside chart describes the microtopographic CO (coast condition), CA (channel condition), IP (permanent flood condition) and IT (temporary flood condition) for each sampling plots.

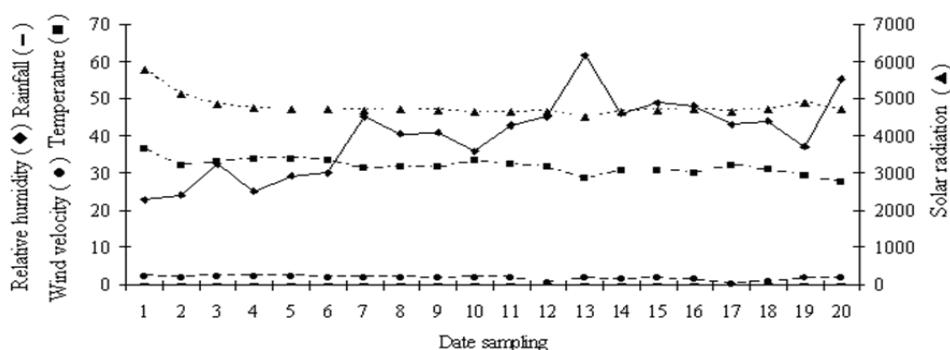


Figure 2. Weekly climatic data from the automatic climatology station of Centro de Investigaciones Biológicas del Noroeste (CIBNOR, S.C.) near the south coast of La Paz Bay.

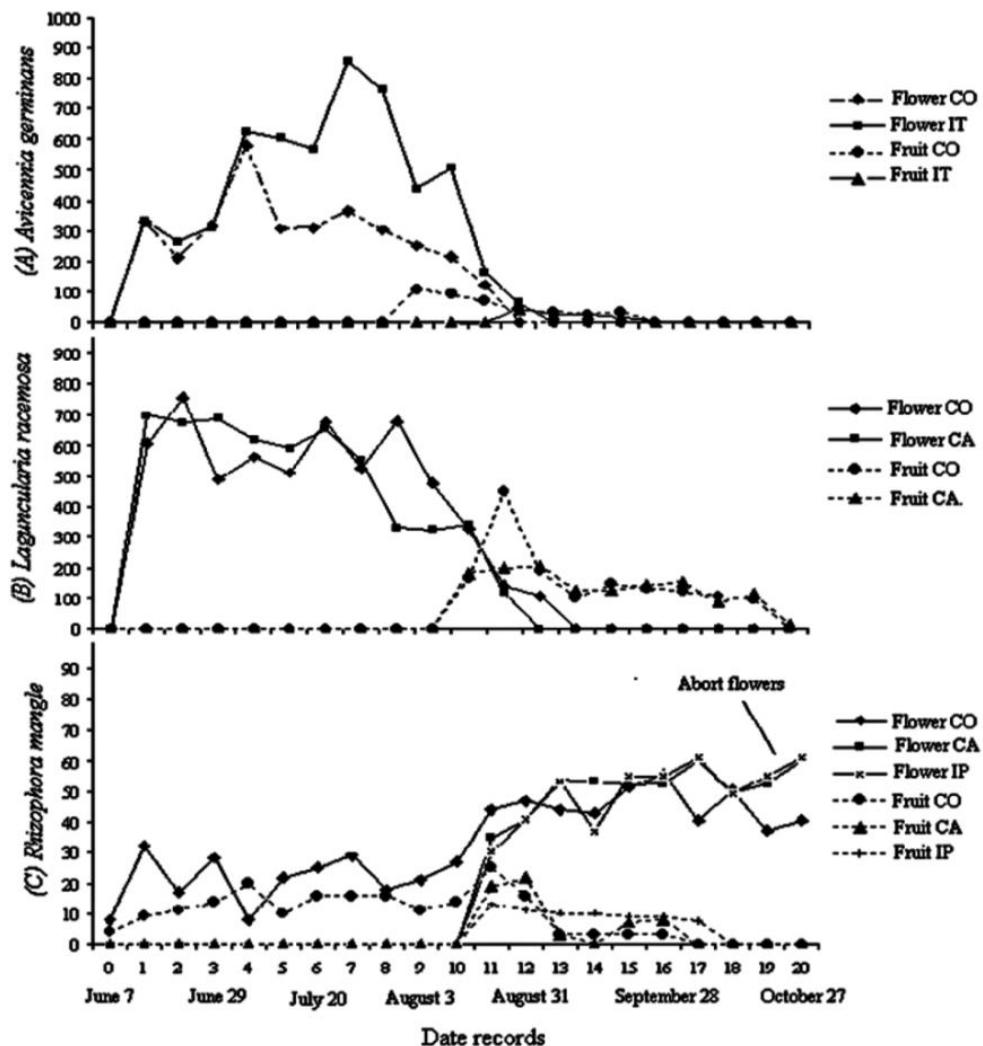


Figure 3. Flowers and hypocotyls registered for *A. germinans*, *L. racemosa*, and *R. mangle* species on 20 sampling test in CO, CA, IP and IT microtopographic conditions.

Flower and Hypocotyls Development Correlated (r) with Environmental Variables

We calculated the multiple correlations (r) of flowers and hypocotyls for *A. germinans*, *L. racemosa* and *R. mangle* species with selected ambient variables relative humidity, rainfall, solar radiation, and ambient temperature and wind velocity. The estimating equation (Y) obtained for *A. germinans* shows a significant level of correlation ($F = 24.06$; $p < 0.05$) for flowers in CO condition with HR and Pp. For flowers in IT condition, the model also shows a good correlation ($F = 23.19$; $p < 0.05$) with HR and Pp variables. For both equations, the variability exceeded 70%. For hypocotyls, the model did not show a significant value ($F = 1.03$; $p > 0.05$) with any one of the five environmental variables.

The estimating equation (Y) obtained for *L. racemosa*, shows a significant level ($F = 31.75$; $p < 0.05$) for correlation of flowers in CO condition with HR and Pp. For CA flowers, the model also shows a good correlation ($F = 34.47$; $p < 0.05$) with T, HR, V and Pp variables. For both flower equations, 76% of total variability was observed in CO and 87%, for CA. In the former condition, hypocotyls showed a significant level of correlation ($F = 5.51$; $p < 0.05$) with T, HR, and Pp. For hypocotyls, CO accounted for 27% and CA for 42% of variability.

The estimating equation (Y) obtained for *R. mangle* shows a significant level of correlation ($F = 16.33$; $p < 0.05$) between flowers and HR and Pp in CO condition. For CA flowers, the model also shows a good correlation ($F = 25.63$; $p < 0.05$) with HR and Pp variables. For flowers in IP condition, the model shows a good correlation ($F = 18.62$; $p < 0.05$) with HR and Pp variables. For three conditions, the CO accounted for 65%, CA for 75%, and IP for 67% of total variability in flowering. For hypocotyls in CO, the model was significant ($F = 8.31$; $p > 0.05$) with ambient temperature (T). For CA condition, all of the five environment variables were significant ($F = 1.10$; $p > 0.05$). However for IP, hypocotyls showed a significant level of correlation ($F = 6.40$; $p < 0.05$) with T, HR, and Pp. For hypocotyls, CO accounted for 27 % of variability, CA for 42%, and IP for 46%.

DISCUSSION

Our results show that the flowering episode of *A. germinans*, *L. racemosa*, and *R. mangle* species in the arid environment on the eastern Baja California Peninsula, México, occurs between June and July months. Those months have the highest incidence of water evaporation and drought of the year (CONAGUA 2006). August marks the beginning of the rainy season and is also the period when the tide level increases, as was reported by CICESE (2006). We observed that the flowering period starts with a rapid change of the recoverable hydric stress in the species studied. Jimenez (1988) and Angel (2001) found a clear relationship between the recovering of the hydric balance with flowering and hypocotyls development peaks. However, *R. mangle* in our study site exceeded the period of flowering production with a poor quantity of flowers in February, and this is comparable with the data for the CO condition for most of the year. In October, all flowers buds in *R. mangle* were interrupted by abortion as found by (Nagarajan et al., 2008). In the case of *A. germinans* and *L. racemosa*, flower production increased in June, appearing also in CA and IP micro topographic conditions.

Kotter et al. (2006) stated that the climate in equatorial regions of the world is divided into wet and dry periods and we assumed that our data support the idea that all of the studied species of mangroves (*A. germinans*, *L. racemosa* and *R. mangle*) developed in arid conditions have the same triggers for reproductive effort. This has been widely documented in tropical areas with abundant rain Jimenez (1988), Duke (1990), Fernández (1999), Angel (2001), Lopez-Portillo and Ezcurra (2002), and Sánchez-Nuñez and Mancera-Pineda (2011). *R. mangle* on the south eastern coast of the Baja California Peninsula presents a flowering time (although with very few flowers and hypocotyls) in the period when *A. germinans* and *L. racemosa* are not reproducing. This is similar to other reports from tropical regions by

Domínguez et al. (1998), Tovilla (1998), Fernández (1999), Ángel (2001) and Tovilla and Orihuela (2002).

The difference between tropical and arid regions is the occurrence of strong abortion of the propagules due to salt stress in late fall resulting from increased tide level. The parental cost is high for *A. germinans* and *R. mangle* but not for *L. racemosa*, because this species responds faster to humidity and flood-saturated ground as exposed by (López-Portillo and Ezcurra 2002). Tomlinson et al. (1979) and Rajendran and Sanjeevi (2004) have stated that changes in mangrove flowering periods are related to latitudinal distance but are not comparative with single local areas or with temporary events, thus, the arid environment of the eastern Baja California Peninsula probably acts as a synergic force of “humidity” as a biological switch for the appearance of flowers in all mangrove species, while hypocotyls development is linked to an intrinsic condition of each tree in poor-nutrient soil, such as in *R. stylosa* (Sharma et al. 2010) found. In our study sites, we found that flowering of all three mangrove species had a positive correlation ($p < 0.05$) with relative humidity (an effect that is expressed as arrival of tropical environmental conditions in June to August. We note that permanent fresh water of riverine areas is not found in this region as is observed for related species in tropical wet zones of the American continent (Álvarez-León 1992; Ángel 2001; Tovilla and Orihuela 2002; López-Portillo and Ezcurra 2002; Sánchez-Núñez and Mancera-Pineda 2011). In our study, we determined the quantity of flowers necessary to develop one hypocotyl, and the total flowers lost by each species in each micro-topographic condition. The flower-hypocotyls development found in this study does not contrast with other results from tropical regions. During long periods of rain and humidity in the tropical environment, the rate of hypocotyls development relative to the number of flowers produced is very low (Tovilla and Orihuela 2002; Sharma et al. 2010) compared with our results.

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Chapter 2

GENETIC STRUCTURE AND EVOLUTION OF *AVICENNIA GERMINANS* AND *RHIZOPHORA MANGLE* ALONG NORTHWESTERN MÉXICO

Eduardo Sandoval-Castro^{1,3,*}, Raquel Muñiz-Salazar^{1,†},
Rafael Riosmena-Rodríguez^{2,‡}, Richard S. Dodd^{3,§},
Juan M. López-Vivas², Sandra C. Moreno-Medina^{1,||},
and Nelva L. Victoria-Cota^{1,#}

¹Escuela de Ciencias de la Salud, Universidad Autónoma de Baja California, Ensenada, Baja California, México

²Programa de Investigación en Botánica Marina, Departamento de Biología, Universidad Autónoma de Baja California Sur, Mexico

³Department of Environmental Science, Policy and Management, University of California, Berkeley, California, US

ABSTRACT

Mangrove forests in the Gulf of California, Mexico represent the northernmost populations along the Pacific coast. Today, these populations are relatively small, fragmented and poor in genetic diversity. We evaluated the population genetic structure

^{*} Eduardo Sandoval-Castro, Escuela de Ciencias de la Salud, Universidad Autónoma de Baja California, Ensenada, Baja California, 22890. E-mail address: sandovalcastro@gmail.com.

[†] Corresponding author: Raquel Muñiz-Salazar. Escuela de Ciencias de la Salud, Universidad Autónoma de Baja California, Ensenada, Baja California, 22890, México. E-mail: ramusal@uabc.edu.mx.

[‡] Rafael Riosmena-Rodríguez: Programa de Investigación en Botánica Marina, Departamento de Biología Universidad Autónoma de Baja California Sur, Km. 5.5 carr al sur, La Paz B.C.S. 23080. E-mail: riosmena@uabcs.mx.

[§] Richard S. Dodd: Department of Environmental Science, Policy and Management, University of California, Berkeley, California, 94720, US. E-mail: dodd@berkeley.edu.

^{||} Sandra C. Moreno-Medina: Escuela de Ciencias de la Salud, Universidad Autónoma de Baja California, Ensenada, Baja California, 22890, México. E-mail: sandra.moreno.medina@gmail.com.

[#] Nelva L. Victoria-Cota: Escuela de Ciencias de la Salud, Universidad Autónoma de Baja California, Ensenada, Baja California, 22890, México. E-mail: nelva.victoria@uabc.edu.mx.

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and recent evolution of the red mangrove, *Rhizophora mangle* and the black mangrove, *Avicennia germinans*, along the northwestern coast of Mexico. Genetic diversity of both mangrove species showed a decreasing trend toward northern latitudes along the Pacific coast. The lowest levels of genetic diversity were found at the range limits around the Gulf of California and the outer Baja California peninsula. Lack of a strong spatial genetic structure in this area and recent northern gene flow in *A. germinans* suggest recent colonization of this species. On the other hand, lack of a signal of recent northern dispersal in *R. mangle*, despite the higher dispersal capability of this species, indicates a longer presence of populations, at least in the southern Gulf of California. We suggest that the longer history, together with higher genetic diversity of *R. mangle* at the range limits, likely provides a gene pool better able to colonize northwards under climate change than *A. germinans*.

INTRODUCTION

Mexican mangroves cover 770,057 ha (Acosta-Velázquez et al., 2009) of which 66% is located along the Atlantic coast (11% in the Gulf of Mexico and 55% on the Yucatan peninsula). Along the Pacific coast, the largest mangrove forests are located in “Marismas Nacionales”, followed by the mangrove forests along the coast of southern Mexico in Oaxaca and Chiapas. Mangroves around the Gulf of California are less extensive and they represent the northern natural limit of mangrove species along the Pacific coast (Pacheco-Ruiz et al., 2006).

The mangrove forests in Mexico are composed of species from four genera (*Rhizophora*, *Avicennia*, *Laguncularia* and *Conocarpus*). *Rhizophora mangle* and *Avicennia germinans* are two most prevalent species in northwestern Mexico and reaches its northern limits along the peninsula of Baja California and the Gulf of California (Pacheco-Ruiz et al., 2006).

Mangroves are highly productive tropical ecosystems that support numerous food chains in the coastal zone and neighboring ecosystems (Tomlinson, 1994). For tropical species, future climate change could provide suitable habitat at higher latitudes providing the opportunity for them to expand their range limits (Jacobs et al., 2004; Milá et al., 2007; Perry and Mendelsohn, 2009; Yamano et al., 2011).

Therefore, understanding the response of mangrove species to climate change is of utmost importance for the management of coastal resources. The global distribution of mangrove marsh is mainly influenced by temperature, restricting species to warm tropical and subtropical latitudes (Duke et al., 1998). Extreme cold events have been hypothesized to explain range transformations and severe regional extinctions and latitudinal limits of mangrove distribution (Graham, 2006; Pil et al., 2011; Rull, 1998).

Population expansions can leave detectable signatures in the distribution of genetic diversity (Pil et al., 2011) that help us to trace their evolution through time. Therefore, a retrospective view of demographic changes in populations can provide insights into their colonization potential under future climate change.

Mangroves at range margins have shown lower genetic diversity and higher genetic structure relative to core populations.

These conditions might be attributed to very low effective population size, pollinator scarcity and higher environmental pressures (Arnaud-Haond et al., 2006; Giang et al., 2003; Maguire et al., 2000b).

Despite their low diversity, peripheral populations are potentially important for conservation because they may preserve rare alleles and gene combinations that are important for adaptation to extreme environmental conditions (Hampe and Petit, 2005; Lesica and Allendorf, 1995). As mangrove forests are facing severe habitat destruction, their conservation is critical.

An improved understanding of their genetic structure is necessary because decreases in population size and connectivity in mangrove forests can directly affect both their genetic diversity as well as that of other species that inhabit mangrove communities (Dudash and Fenster, 2000; Landergott et al., 2001; Srikwan and Woodriff, 2000; Young and Clarke, 2000).

Propagule dispersal plays an important role in the colonization of new environments and a particularly strong influence in species structuring, distribution, genetic composition and connectivity (Hanson et al., 2008; Sousa et al., 2007). As a consequence of the importance of genetic variability in evolutionary processes, many conservation plans are currently designed to maintain and maximize it.

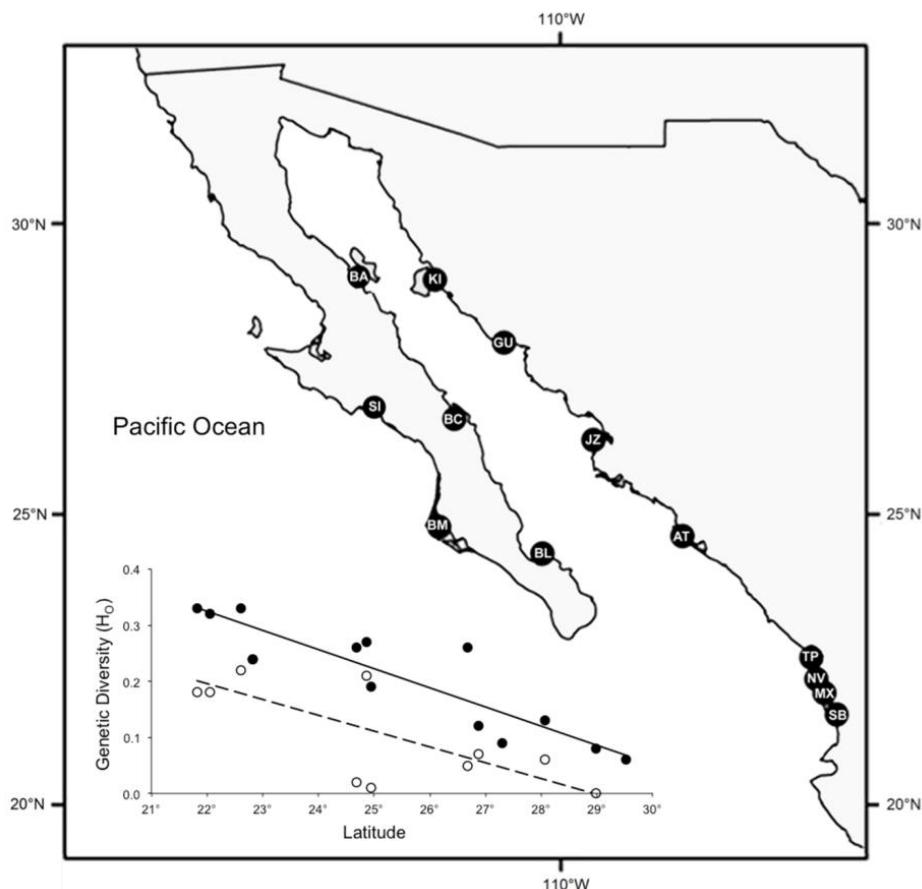


Figure 1. Sampling locations are illustrated on the map. Graph below shows decreasing trend in genetic diversity toward northern populations of mangrove species, *A. germinans* (white circles) and *R. mangle* (black circles).

MATERIALS AND METHODS

Plant Material and DNA Isolation

Leaf tissue from 295 individuals of *A. germinans* and 399 of *R. mangle* was collected from mangrove forests along the northwestern coast of Mexico. These data are part of a more extensive study and have been reported in Sandoval-Castro et al. (2014) but here we focus only at northwestern coast of Mexico. Sampling was performed from Southern Marismas Nacionales (San Blas Nayarit) to the northernmost mangrove populations in the Gulf of California and Peninsula of Baja California (Figure 1). Samples were taken from specimens separated by at least 30 m when possible to prevent consanguinity and to maximize the probability of collecting diverse genotypes.

The samples were dehydrated and stored in silica gel until DNA extraction. Total genomic DNA was isolated from approximately 200 mg dry weight of leaf using a modified CTAB/PVP method (Sandoval-Castro et al., 2012).

Microsatellite Analysis

Individuals were genotyped at seven loci (AgT4, AgT7, AgT8, AgT9, AgD6, AgD13 and CA_002) previously designed for *A. germinans* (Ceron-Souza et al., 2006; Nettel et al., 2005) and six loci (Rm7, Rm11, Rm19, Rm21, Rm38, and Rm46) previously designed for *R. mangle* (Rosero-Galindo et al., 2002). These repeat motifs were isolated from genomic DNA and are expected, a priori, to be neutral with respect to natural selection. The forward primers were fluorescent-labeled with FAM, VIC, PET and NED (Applied Biosystems Inc). All amplifications were performed on a MyCycler BIORAD thermal cycler in 20 µL PCR reactions containing 1x Buffer (10mM Tris HCl, 50 mM KCl, pH 8.3, SIGMA), 2.0 mM MgCl₂, 200 µM of each dNTP, 0.15 µM of each primer, 1 unit *Taq* DNA polymerase (SIGMA) and 20 ng of genomic DNA. With the exception of some small variations on annealing temperatures, all microsatellite loci were amplified with a similar thermocycler profile. The profile consisted of an initial denaturation step at 95 °C for 5 min, followed by 35 amplification cycles as follows: 95 °C for 30 s, annealing temperature (50 °C for all the loci in *R. mangle* and for the loci AgT4, AgT7, AgT8 and AgT9, 55 °C for AgD13 and CA_002 and 59 °C for AgD6 in *A. germinans*), for 30 s and 45 s at 72 °C, ending with an extension cycle at 72 °C for 30 min. To ensure reproducibility and consistency in PCR amplification, approximately 5% of samples were re-amplified. In addition, a negative control was run for each set of PCR reactions and genotyped to check for contamination. Amplified products were run on an ABI 310 automated DNA sequencer, and microsatellite alleles were visualized and scored in the program GeneMarker 1.97 (Softgenetics).

Data Analysis

Genetic Diversity

To determine spatial patterns in genetic diversity that might inform us about latitudinal trends consistent with recent colonization northwards and population genetic structure, we estimated the number of alleles (A), unbiased expected heterozygosity (HE) and observed heterozygosity (HO) for each locus across all localities using GDA 1.1 (Lewis and Zaykin, 2001). Allelic richness per locus (AR) and per population were calculated using FSTAT 2.9.3 (Goudet, 2001). Global tests for deviation from Hardy-Weinberg equilibrium were performed using a Markov chain algorithm and linkage between all pairs of loci was estimated using GENEPOP 4.0 (Raymond and Rousset, 1995; Rousset, 2008) with significance levels determined using the Markov chain method. For all Markov chain tests, the default parameters in GENEPOP were used with 100 batches of 1000 iterations each. Null alleles, large allele dropout and stutter peaks were explored using Micro-Checker 2.2.3 (Oosterhout et al., 2004).

Clustering Analyses

We applied the Bayesian clustering algorithm implemented in STRUCTURE v2.2.3 (Pritchard et al., 2000) as an exploratory analysis to infer population genetic structure, assigning individuals (probabilistically) without a priori knowledge of population boundaries. STRUCTURE uses individual multilocus genotype data to cluster individuals into K groups while minimizing Hardy-Weinberg disequilibrium and gametic phase disequilibrium between loci within groups (Pritchard et al., 2000). STRUCTURE runs were based on 500,000 iterations after a burn-in of length 500,000 and assumed correlated allele frequencies and an admixture model with an estimated proportion α of admixed individuals. To check for Markov chain Monte Carlo (MCMC) convergence, we performed 10 replicates for each K value and checked the consistency of results. The most likely number of clusters (K) was considered to be the K value with the highest $\text{Pr}(X|K)$ (Falush et al., 2003; Pritchard et al., 2000). Also, the optimal K value was calculated after the ΔK method described by Evanno et al. (2005).

Genetic Structure

Genetic differentiation among populations was evaluated at each locus and over all loci by calculating the measures of relative genetic differentiation among populations defined under the infinite allele model (IAM; F_{ST}) (Weir and Cockerham, 1984) and the stepwise mutation model (SMM; R_{ST}) (Michalakis and Excoffier, 1996). The presence of phylogeographic structure was assessed by using SPAGeDi 1.4 (Hardy and Vekemans, 2002) through permutations of allele sizes among alleles within a single locus (pR_{ST}) (10,000 permutations).

This analysis compares the observed R_{ST} value (before randomization) with the distribution of pR_{ST} values obtained for all possible configurations of allele size permutations. If observed R_{ST} is within the upper 5% of the distribution of pR_{ST} the contribution of mutations to population differentiation is non-negligible compared with genetic drift and migration (Hardy et al., 2003). Population genetic structure was also examined using hierarchical analysis of molecular variance (AMOVA) among the clusters determined by

STRUCTURE v.2.3.3. At this level, genetic differentiation was quantified with F-statistics (Weir and Cockerham, 1984) using ARLEQUIN v.3.5. (Excoffier and Lischer, 2010). The distribution of genetic variation was assessed at four hierarchical levels: among groups, among populations within groups, among individuals within populations and within individuals (Excoffier et al., 1992). Statistical significance of the variance was tested by 10,000 non-parametric permutations.

Migration Rates

On the basis of STRUCTURE analysis, we classified localities into populations. The magnitude and direction of gene flow were estimated among these populations. Estimates of evolutionary patterns of gene flow were obtained according to the maximum likelihood approach implemented in MIGRATE v.3.2.7 (Beerli and Felsenstein, 1999; Beerli and Palczewski, 2010). MIGRATE uses a coalescence approach to estimate migration rates (Nm) among populations, assuming a constant per-locus mutation rate. This approach is judged to estimate gene flow more accurately than other FST methods, especially when multiple loci are employed (Beerli and Felsenstein, 1999). We used 10 short-chain searches and three long-chain searches over the number of assayed microsatellite loci to obtain the magnitude and direction of gene flow according to Beerli and Felsenstein (1999). For each locus, the program was run for 10 consecutive exploratory chains with lengths of 5×10^6 genealogy visits to adjust the driving values for both the run and the 3 long chains; the last chain was used to generate the presented results. Each of the long chains visited 5×10^8 genealogies, sampling 5000 after an initial burn-in of 10,000 steps. The program assumes discrete populations and generations, mutation-drift equilibrium, non-selective effects and the stepwise mutation model for microsatellite markers (Otha and Kimura, 1973).

Furthermore, we estimate recent levels of gene flow between populations by using the program BAYESASS (Wilson and Rannala, 2003), which uses transient levels of linkage disequilibrium produced by recent migrants or their immediate descendants to infer levels of migration into populations.

The program uses MCMC sampling in a Bayesian statistical framework to estimate gene flow in the recent past. The run involved 3×10^6 MCMC iterations, discarding the first 10^6 . The program estimates the mean value for migration rate, and a 95% confidence interval for the estimate.

Isolation by Distance

To test the significance of Isolation By Distance (IBD), the correlation between genetic and geographic distance matrices was tested using a Mantel test with 2000 permutations (Jensen et al., 2005). The geographical distances between samples were based on coastline distances and the genetic distance was expressed as $F_{ST}/(1-F_{ST})$ following Rousset (1997). IBD analysis was only performed on the mainland coast populations (KI, GU, JZ, AT, TP, NV, MX, and SB) to avoid complex scenario of dispersal to and around the peninsula of Baja California.

RESULTS

Genetic Diversity

Microchecker analyses showed no evidence of null alleles nor allelic drop out that might interfere with posterior analysis on genetic diversity. Genotypic linkage disequilibrium was not detected among any of the pairwise loci comparisons across all populations. This suggests no evidence for selective sweeps and supports the expectation that the markers are neutral with respect to natural selection. We detected 20 and 44 alleles in *R. mangle* and *A. germinans*, respectively. Furthermore, *A. germinans* showed higher number of alleles. However, *R. mangle* showed higher genetic diversity considering the other genetic diversity indexes (allelic richness and heterozygosity, see Table 1) because most of the alleles in *A. germinans* were present at very low frequency. ANCOVA analysis applied to the corrected variation (residuals) of genetic diversity (H_O) versus latitude as a covariate supported this lower representation of alleles in *A. germinans* and evidenced a higher genetic diversity of *R. mangle*. Genetic diversity in both mangrove species showed a decreasing trend northward (Figure 1); lowest values were detected along the northern coasts of the Gulf of California, where mangrove species reach their natural range limits. The Pearson correlation coefficients between observed heterozygosity and latitude was $r = -0.75$ for *A. germinans* and -0.91 for *R. mangle* ($P < 0.05$).

Table 1. Genetic diversity of *A. germinans* and *R. mangle* in terms of total number of alleles (A), Allelic richness (AR), expected (HE) and observed heterozygosity (HO)

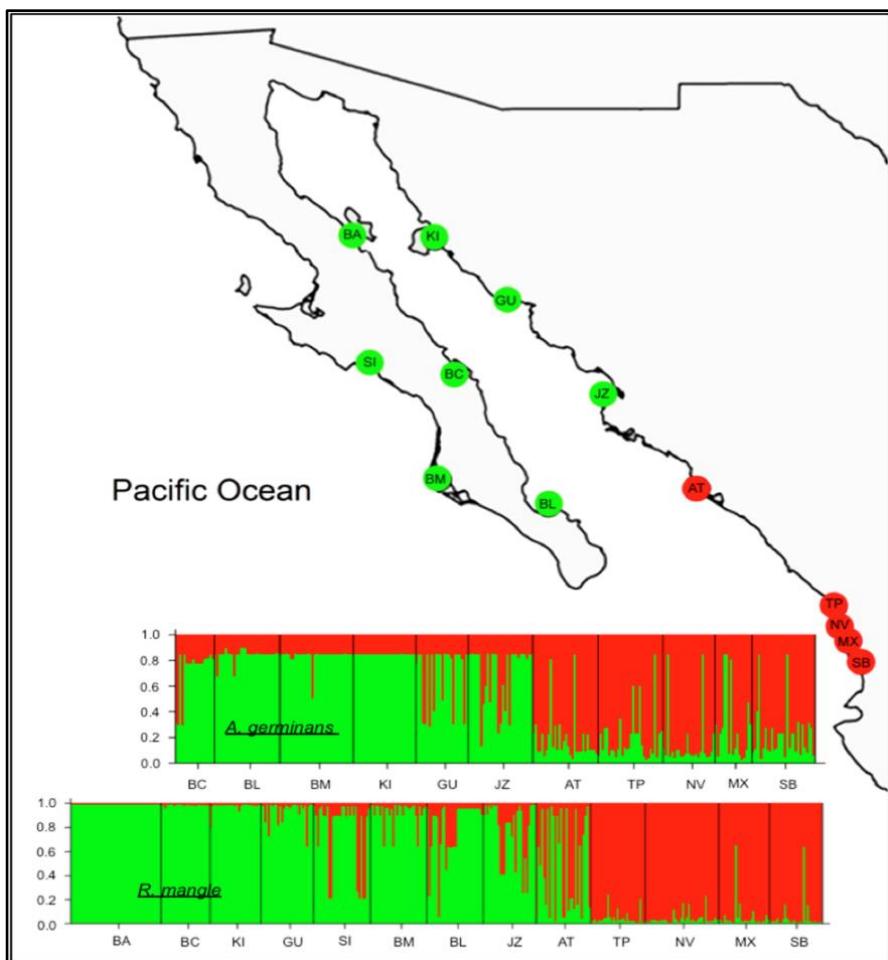
Sampling site	Code	N	A	AR	HE	H_O	N	A	AR	HE	H_O
		<i>A. germinans</i>						<i>R. mangle</i>			
Bahía de los Ángeles	BA	---	---	---	---	---	48	8	1.25	0.09	0.06
Laguna San Ignacio	SI	---	---	---	---	---	30	9	1.43	0.13	0.09
Bahía Kino	KI	29	7	1.00	0.00	0.00	27	9	1.33	0.10	0.08
Bahía Concepción	BC	18	10	1.29	0.08	0.07	26	9	1.43	0.12	0.12
Bahía Magdalena	BM	34	9	1.08	0.01	0.01	30	10	1.48	0.18	0.19
Balandra	BL	30	9	1.15	0.02	0.02	30	11	1.60	0.18	0.26
Guaymas	GU	24	14	1.41	0.07	0.06	28	13	1.76	0.15	0.13
El Jitzámuri	JZ	30	15	1.37	0.05	0.05	28	12	1.87	0.26	0.26
Bahía de Altata	AT	30	19	1.94	0.21	0.21	29	15	2.12	0.26	0.27
Teacapán	TP	30	22	2.12	0.24	0.24	29	14	2.12	0.32	0.24
Novillero	NV	24	24	2.21	0.25	0.22	39	14	2.05	0.32	0.33
Mexcaltitán	MX	17	18	2.03	0.22	0.18	26	13	2.00	0.31	0.32
San Blas	SB	29	19	1.96	0.23	0.18	29	14	2.07	0.33	0.33
<i>Mean</i>		27	15.09	1.60	0.13	0.11	31	11.62	1.73	0.21	0.21

(N) Number of sampled individuals, (---) no sample.

Both mangrove species showed lower levels of genetic diversity around the Gulf of California, where mangrove forests are less extensive and more fragmented.

Clustering Analyses

Applying the Bayesian analysis in STRUCTURE and the approach of Evanno et al. (2005) the most likely number of genetic clusters was two for both *A. germinans* and *R. mangle*, respectively. Sample locations (AT, TP, NV, MX, SB) from “Marismas Nacionales” formed one of the groups and the rest located in the peninsula and Gulf of California the other (BA, SI, BM, BC, BL, KI, GU, JZ; see Figure 2). Sampling locations in the entrance of the Gulf of California showed admixture and shared genotypes among themselves.



In the graph, the Y-axis represents the percentage of assignment of individuals into the groups and x-axis the sampling sites.

Figure 2. STRUCTURE analysis clustered mangrove populations in two groups, which are depicted in colors on the map. Green color represents populations from the Gulf of California and red color from Marismas Nacionales.

Genetic Structure

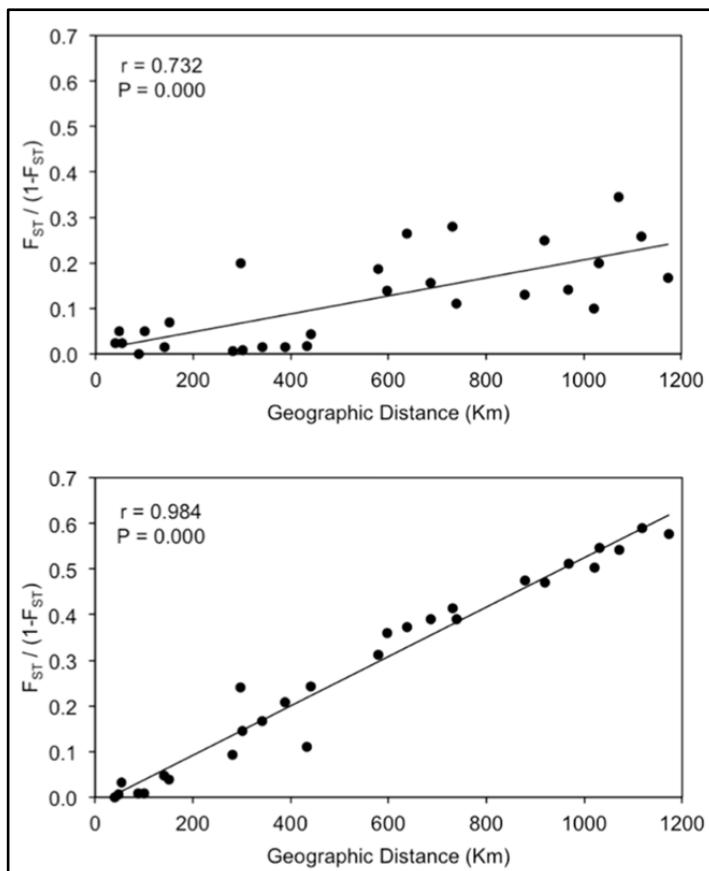
SPAGeDi analyses showed contrasting results. *R. mangle* showed that differentiation among all populations taking into account allele sizes ($R_{ST} = 0.53$) was significantly larger than differentiation based on allele identities ($F_{ST} = 0.35$).

On the other hand, in *A. germinans* the lowest values of genetic differentiation were observed when allele sizes were considered ($R_{ST} = 0.08$, $F_{ST} = 0.17$; $\alpha = 0.05$, $P < 0.000$).

These results suggest more recent evolutionary processes of migration and genetic drift presents in the populations of *A. germinans* at northwestern coast of Mexico.

Isolation by Distance

Both mangrove species showed that Isolation by distance was significant over all sampling locations along the northwestern Pacific coast of Mexico (Figure 3).



The Y-axis is $F_{ST}/(1-F_{ST})$ following Rousset (1997) and the X-axis the geographic distance (km).

Figure 3. Isolation by distance is showed in the graphs, above *A. germinans* and below *R. mangle*. Correlations and probabilities were estimated from a Mantel test with 2000 repeats of bootstrap resampling.

Migration Rates

In view of the genetic structure observed along the northwestern coast, we focused on estimating directional gene flow between the Gulf of California (GC) and “Marismas Nacionales” (MN; Figure 4).

Evolutionary-scale rates of migration (MIGRATE) showed a higher magnitude than contemporary-scale rates (BAYESASS). Bayesian estimates based on MCMC simulations performed using BAYESASS suggest that for *A. germinans* contemporary gene flow is predominantly northward.

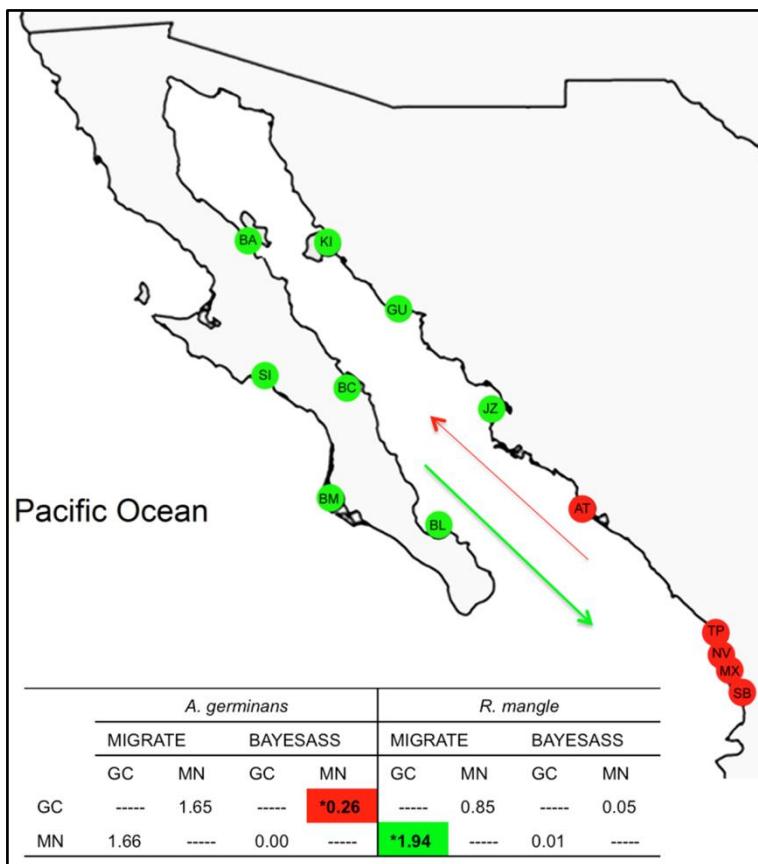
On other hand, for *R. mangle* gene flow on an evolutionary-scale (MIGRATE) was higher southward from GC to MN. These inferred migration rates suggest that *R. mangle* populations in the Gulf of California are older than those of *A. germinans*. This is consistent with the present more northerly distribution of *R. mangle* along the Peninsula of Baja California today.

DISCUSSION

The northern-most populations of mangroves along the Pacific coast, in the Gulf of California and along the outer coast of the Baja California peninsula, are likely to be source populations for colonization at higher latitudes as climate becomes more favorable for mangrove establishment. Today, these populations are relatively small and fragmented and prior research has indicated that they are poor in genetic diversity (Nettel and Dodd, 2007; Nettel et al., 2008; Sandoval-Castro et al., 2012). Here we set out to investigate whether the low diversity in this region was a result of recent founder events, or fragmentation and genetic drift of once more extensive mangroves due to climatic changes in the recent past.

The alternative origins of the contemporary mangrove populations in north-western Mexico are likely to result in contrasting adaptive potential in this region of high aridity, low sedimentation and elevated salinity. Although genetic drift in fragmented populations of an earlier more continuous presence of mangroves may lead to suboptimal residual genotypes, adaptive genes for these stressful environments are still likely to be present. On the other hand, recent colonizations from lower latitude populations are more likely to include maladapted genes from more benign environmental conditions.

Paleobotanical records suggests that mangroves have been present in the Americas since the Eocene, *R. mangle* arrived earlier (\approx 33 million years ago), and *A. germinans* 10 million years later, during the Miocene (Rull, 1998). Although apparently both mangrove species colonized the American Pacific coast from Atlantic coast through the open Central American Isthmus (Graham, 1995, 2006; Rico-Gray, 1993; Rull, 1998), Takayama et al. (2009) suggest also a colonization across Pacific in *R. mangle*, giving the opportunity to accumulate more genetic variation than *A. germinans* given that his earlier colonization and direct migration from Indopacific populations. This explains the higher genetic diversity of *R. mangle* in terms of allelic richness and heterozygosity.



Arrows show significant gene flow, red color shows recent and green the historical gene flow.

Figure 4. Estimates of the historical gene flow and migration directions estimated in MIGRATE; and estimate of recent migration rate and directions estimated in BAYESASS for the northwestern Mexican mangrove populations of *A. germinans* and *R. mangle*.

We compared molecular diversity of the two major taxa of the ecosystem, expecting that recent mangrove establishment would show a similar spatial pattern of genetic diversity for both *A. germinans* and *R. mangle*. Our results showed the lowest levels of genetic diversity around the Gulf of California, mainly at populations located near to the northern limit of mangrove distribution. It is well documented that populations near to their limits of distribution are characterized by low levels of genetic diversity since they are away from ancestral populations (Triest, 2008) and also because the environmental conditions are not the most appropriate to the recruitment and optimum development of the species (Krauss et al., 2008).

This study shows that Kino Bay is apparently the most affected population; in that locality we did not observe genetic variation among 29 individuals analyzed in *A. germinans*. So, our results with seven microsatellite loci suggest that individuals genetically identical integrate Kino Bay, which could be not favorable to the long-term persistence and stability of *A. germinans* in the system.

Mangrove vegetation located around the Baja California Peninsula are geographic and naturally isolated by the lack of appropriate habitats, thus we would hope to find high levels

of both inbreeding and genetic structure. However, our results did not show significant inbreeding. Therefore, the low genetic diversity could be partially explained by sporadic founder events from a small number of individuals, which do not fully represent the ancestral gene pool.

In general, our findings indicate a notable poor genetic diversity compared with the reported for *A. germinans* along the Pacific coast of Costa Rica and southern Mexico (Nettel et al., 2008), and for both *A. germinans* and *R. mangle* in Pacific coasts (Arbeláez-Cortes et al., 2007; Cerón-Souza et al., 2010; Salas-Leiva et al., 2009).

The mangrove vegetation along the Gulf of California is more discontinuous (principally at peninsula of Baja California) than the observed in Marismas Nacionales, suggesting that processes as fragmentation and the effective dispersion of propagules have noticeable effects on the genetic variation of this specie along their ranges of distribution. This same pattern has been reported in previous studies for *R. mangle* on northern range limits in Mexico (Sandoval-Castro et al., 2012), in Brazilian (Pil et al., 2011) and Colombian coasts (Arbeláez-Cortes et al., 2007) with microsatellites as well as for *A. germinans* (AFLP) along American coasts (Dodd et al., 2002; Nettel and Dodd, 2007), and *A. marina* (Arnaud-Haond et al., 2006; Giang et al., 2003; Maguire et al., 2000a; Maguire et al., 2000b) with microsatellite loci around the Indo-West Pacific (IWP).

MIGRATE analysis shows bi-directional migration for *A. germinans* and stronger southward migration for *R. mangle*. This presumably reflects earlier phases when mangroves were present in the Gulf, perhaps during the early Holocene. The southward movement could be because of the prevailing California Current. BAYESASS shows a different pattern with *A. germinans* predominantly dispersing northwards consistent with recent colonization in the Gulf, but not for *R. mangle*. These results suggests that *R. mangle* might probably been present in the Gulf of California for a long time, perhaps surviving somewhere in the Gulf of California to the harsh conditions during the last glacial maximum. This hypothesis is supported by evidence of historical southward migration (See green arrow in Figure 4) and also by the higher levels of genetic diversity observed in *R. mangle* at the Gulf of California and by the northernmost range limits, compared with *A. germinans*. On the other hand, we most believe that *A. germinans* was severely affected and inclusive extinct from the Gulf of California by harsh conditions during glaciations. We hypothesize that current populations in the Gulf of California were recent colonized after the last glacial maximum (around 18,000 years ago) from “Marismas Nacionales”. This hypothesis is supported by the BAYESASS analysis, which showed a northward migration from “Marismas Nacionales” to the Gulf of California (See red arrow in Figure 4) and also by the observed linear gradients in genetic diversity and significant levels of IBD, which is consistent with an expansion scenario for mangrove forests in the northwestern coast of Mexico, as suggested by López-Medellín et al. (2011) in Bahía Magdalena. Our findings highlight the importance of “Marismas Nacionales” along time as an important source of propagules during the mangrove colonization into the Gulf of California.

Today “Marismas Nacionales” is one of the more extensive areas of mangrove forests and with the highest levels of genetic diversity.

In conclusion, our data provide compelling evidence that Gulf of California and outer Baja California were recently colonized by *A. germinans* through northward propagule dispersal, perhaps from relictual stands around present-day Marismas Nacionales. On the other hand, *R. mangle* showed no signal of recent northern dispersal despite the greater

dispersal capability of this species, suggesting the presence of populations at least in the southern Gulf of California for some time. The northernmost populations of *R. mangle* in the Gulf are probably recent colonizations.

Therefore *R. mangle* may serve as a better adapted gene pool for latitudinal advances under climate change than *A. germinans*. Populations with a longer history at a site are likely to be better adapted to local conditions than founders from populations under more benign conditions. Under this scenario, *R. mangle* would serve as a better adapted gene pool for latitudinal advances under climate change, even though *A. germinans* showed evidence of more successful northward migration in recent time.

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Chapter 3

SECONDARY PRODUCTION OF BIVALVE POPULATIONS IN THE MANGROVES SWAMPS

***Esteban F. Felix-Pico^{1,*}, Mauricio Ramirez-Rodriguez¹
and Jorge A. Lopez-Rocha²***

¹Centro Interdisciplinario de Ciencias Marinas del Instituto Politecnico Nacional,
La Paz, Baja California Sur, Mexico

²Unidad Multidisciplinaria de Docencia e Investigacion, Facultad de Ciencias,
Universidad Nacional Autonoma de Mexico, Sisal, Yucatan, C.P. Mexico

ABSTRACT

Mangrove ecosystems are a critical habitat for many species. The productivity of mangroves is important due its contribution of organic detritus and secondary production. The objective was to identify indicators of secondary productivity of mangrove systems to establish a baseline for measuring changes in the mangrove system due potential natural or anthropogenic disturbances. The bivalve mollusks, the black ark *Anadara tuberculosa* and the palmate oyster *Saccostrea palmula* were chosen as indicators of secondary productivity. Samples were collected in three mangroves of the sand barrier El Mogote in the Ensenada de La Paz, from August 2007 to July 2009. Abundance and secondary production were estimated based on biomass increase. The estimated weight increase was performed by determining individual growth rates through an analysis of length frequency distributions. The average abundance was 1.27 and 510 ind. m⁻² of black ark and palmate oyster respectively. Growth rates of 4.91 mm·month⁻¹ (3.67 g month⁻¹) for black ark and 3.80 mm month⁻¹ (0.18 g month⁻¹) for the palmate oyster were estimated. Average secondary production of black ark was 4.51 g m⁻² month and was higher during the spring, while for the mangrove oyster was 97.9 g m⁻² month⁻¹, and the maximum production was recorded during the summer. We found 78 macroinvertebrate species within the mangrove swamps surveyed in this study. There are just a few species of bivalves associated with mangroves that represent an economic interest for their productivity. The black arks and oysters are among the most important mangrove-associated species because of their commercial value and continual

* Email: efelix@ipn.mx.

exploitation. The obtained results, such as a quality index of benthos, are suitable for measuring changes to the mangrove system in the Ensenada of La Paz, Baja California Sur, Mexico.

Keywords: Mollusks, *Anadara*, *Saccostrea*, mangrove, secondary productivity, Baja California Sur

INTRODUCTION

Mangrove ecosystems are critical habitats for a large variety of mollusks, crustaceans, and fish of economic importance and ecological interest. The mangroves biological richness makes these zones one of the most productive ecosystems. Mangroves are also important because they provide food and shelter for a large population of juvenile organisms of many species (Mann, 2000; Aburto-Oropeza et al., 2008).

Mangrove productivity is fundamental because of its contribution of organic detritus and secondary productivity. Mangrove systems are considered one of the most productive ecosystems on earth; they are four times as productive as the continental shelf and 40 times as productive as oceanic regions (Berger et al., 1989).

Mollusks inhabiting mangrove ecosystems are mostly gastropods and bivalves, which are commonly found attached to the roots of mangrove trees; examples include *Littorina* and *Nerita* snails and the oysters *Crassostrea corteziensis* (Hertlein, 1951) and *C. columbiensis* (Hanley, 1846). In muddy substrates, the bivalves *Chione* spp., *Tagelus* spp., *Anadara* spp., and *Mytella* spp. are common (Lacerda et al., 2002)]. In the mangrove systems fringing La Paz Bay, two of the most common bivalve species are the black ark (*Anadara tuberculosa* Sowerby, 1833) and the palmate oyster (*Saccostrea palmula* Carpenter, 1857).

The arcid bivalve mollusk *Anadara tuberculosa* (G.B. Sowerby I, 1833) is an important commercial and subsistence resource in many estuaries along the Pacific coast from Mexico to Peru. Average landings over the last 10 years are 800 t of fresh whole weight, from Bahia Magdalena, Baja California Sur, Mexico. *A. tuberculosa* is recognized as a good biological candidate for culture because of its high fecundity and good growth rate, reaching commercial size (50- mm shell length in natural conditions) at age two years (Baqueiro-Cardenas, 1980; Cruz & Palacios, 1983; MacIntosh, 1994).

The black ark is a species of great economic importance and is harvested throughout the Pacific littoral (MacKenzie, 2001). This species is associated with the red mangrove (*Rhizophora mangle* Linnaeus) in the intertidal zone, where it lives buried in the mud. It is distributed in the Eastern Pacific, from the coasts of the Mexican state of Baja California Sur (B.C.S.) south to Peru (Keen, 1971). In Mexico it is mainly harvested in Bahía Magdalena, B.C.S., (Figure 1) (Hernandez-Valenzuela, 1996; Felix-Pico et al., 2009).

On the other hand, the mangrove oyster is distributed from San Ignacio Lagoon in the Mexican state of Baja California, south to Panama. It is also found in the Galapagos Islands of Ecuador and Cocos Island, Costa Rica (Keen, 1971; Cruz & Jimenez, 1994; Coan & Valentich-Scott, 2012) and, due to its small size, has little commercial value.

Mangrove areas have been seriously reduced in recent decades due to human activities (Valiela et al., 2001), a situation that makes the formulation of mangrove sustainable management schemes increasingly important. For this reason, the objective of this

investigation was to define indicators of secondary productivity in mangrove systems in Ensenada de La Paz, B.C.S., Mexico, to set out a baseline that will allow measurement of anomalies due to potential natural or anthropogenic perturbations (Figure 2).

BIOLOGY AND ECOLOGY

Species and Distribution

Scientific name: *Anadara tuberculosa* (G.B. Sowerby I, 1833)

Common names: Almeja Pata de Mula Negra (Mexico), Piangua (Costa Rica), Black Ark (FAO).

Synonyms: *Anadara septicostata* (Reeve, 1844); *Arca septicostata* Reeve, 1844; *Arca tuberculosa* G. B. Sowerby I, 1833 (original combination).

Mexican species code: 24,34 (Ramirez-Rodriguez, 2013).

FAO species code: 3160407112.

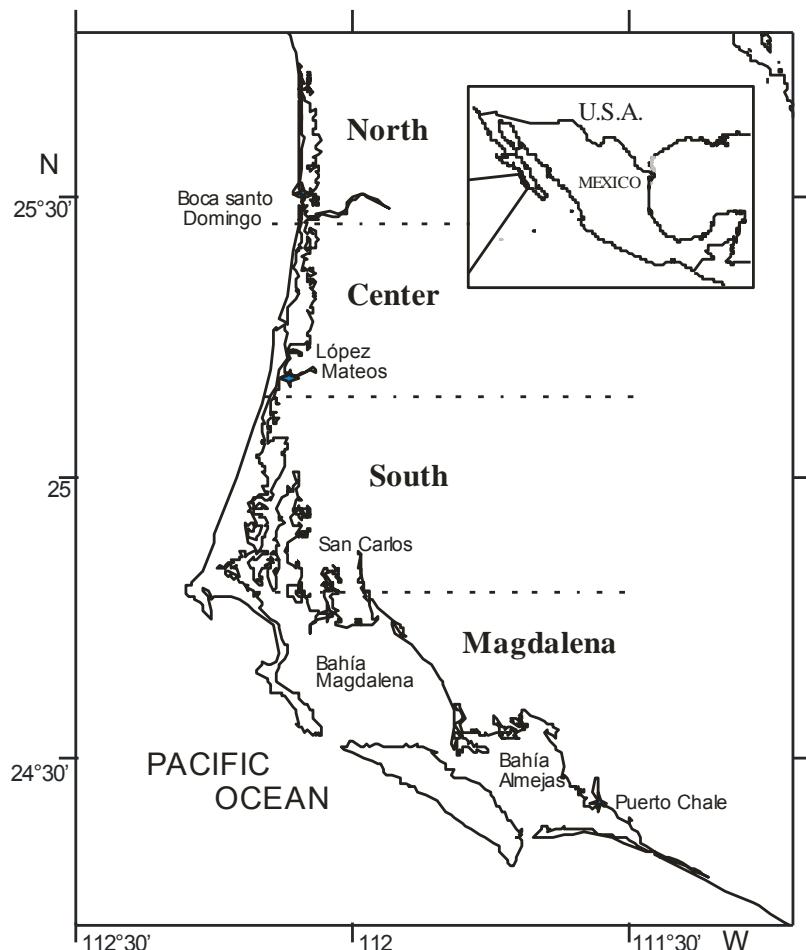


Figure 1. Map of black ark fishing areas in Bahía Magdalena, Baja California Sur, Mexico.

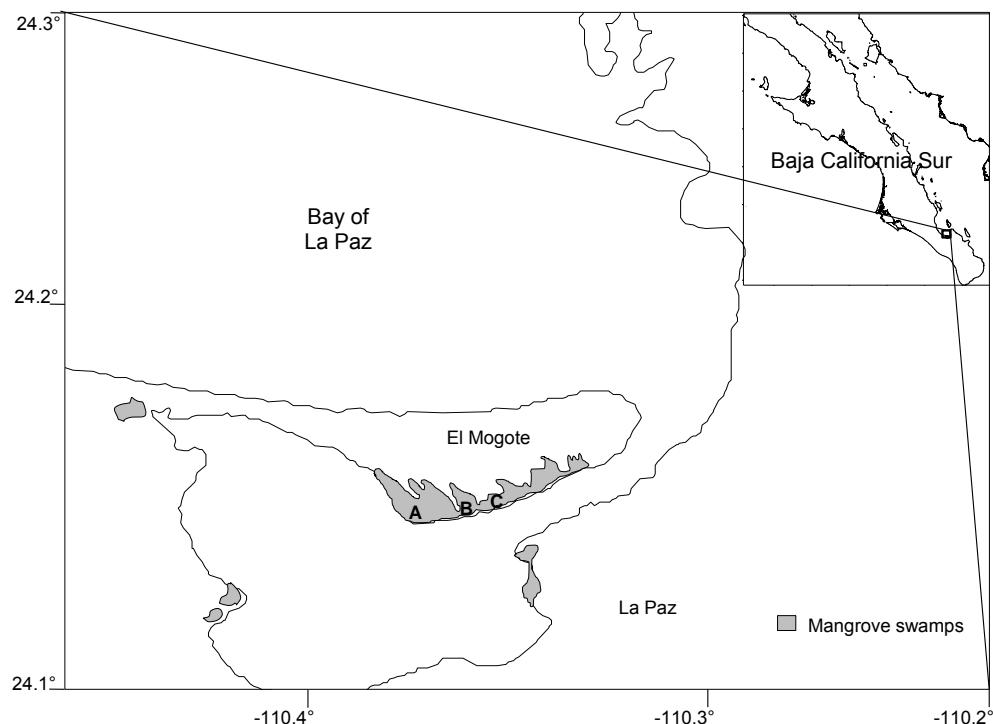


Figure 2. Study area in El Mogote, La Paz Bay, B.C.S. The locations of the three estuaries (A, B, C) where secondary productivity was estimated are shown.

Diagnostic features: Shell ovate-subquadrate, heavy, inflated; inequilateral, much longer posteriorly; anterior end broadly rounded; posterior end obliquely subtruncate; beaks inflated, prosogyrate; dorsal margin squared anteriorly and posteriorly; sculpture of 32-40 rounded, radial ribs, with weak to heavy tubercles, interspaces narrow; periostracum dark brown, heavy, scabrose, with setae in interspaces (except in large specimens); cardinal area narrow, entirely covered with ligament, with 2-4 chevron grooves; hinge plate wide, with 24-27 anterior and 25-30 posterior. Length to 97 mm (Coan & Valentich-Scott, 2012).

Literature: [Baqueiro-Cardenas (1980), Baqueiro-Cardenas et al. (1982), Betancourth & Cantera (1978), Campos et al. (1991), Coan et al. (2000: 133), Cruz-Soto (1982, 1984), Espinosa et al. (2008, 2011), Felix Pico et al. (2004, 2007, 2009, 2011), Flores & Licandeo (2008); García-Dominguez et al. (2008), Grant & Gale (1931: 141), Hertlein & Strong (1943: 157-158), Hertz & Hertz (1992), Keen (1971: 44), MacKenzie (2001), Maury (1922: 188-189), Muñeton et al. (2004), Olsson (1961: 87-88), Poutiers (1995:115), Reinhart (1943: 63), Rost (1955: 195), Silva & Bonilla (2001), Squires et al. (1975), Stevenson (1972: 201); Cited by Coan & Valentich-Scott (2012)].

Distribution: Although reported in Mission Bay, California, U.S.A. (33.7°N), populations are not permanently established north of Laguna San Ignacio on the Pacific coast of Baja California Sur (26.8°N). It occurs in the Gulf of California at La Paz, Baja California Sur (San Jose island, 24.54°N), and Bahia Kino, Sonora (28.8°N), Mexico and south to Punta Telegrafo, Paita, Peru (5.1°S) [Cited by Coan & Valentich-Scott (2012)]. It is found from the intertidal zone to 20 m, in the mud of bays and mangrove swamps.

Scientific name: *Saccostrea palmula* (Carpenter, 1857)

Common names: Ostión de Manglar o Palmado (México), Palmate Oyster (FAO).

Synonyms: *Ostrea conchaphila palmula* Carpenter, 1857; *O. panamensis* Carpenter, 1864; *O. amara* Carpenter, 1864; *O. mexicana* G.B. Sowerby II 1871; *O. cumingiana* Dunker, auctt., non Dunker, 1847.

Mexican species code: 557 (Ramirez-Rodriguez, 2013).

FAO species code: 3160700305.

Diagnostic features: Shell irregularly shaped; narrowly to moderately attached by left valve, deeply cupped; right valve concave, flat, or somewhat convex, fitting inside left valve; shell margin strongly undulating; sculpture of a few conspicuous sharp plications, especially in left valve; resilifer long, narrow; strong chomata on either side of resilifer; exterior color cream to purplish-brown, some with radial rays; interior color white with green or purple patches, some with purple margin. Height to 75 mm (Keen, 1971; Coan & Valentich-Scott, 2012).

Distribution: Laguna San Ignacio, Baja California Sur (26.8°N), in the Gulf of California as far north as near its head at Bahía la Choya, Sonora (31.4°N), Mexico, and south to Isla del Coco, Costa Rica, Bayovar, Piura, Peru (5.9°S) and the Islas Galápagos, Ecuador; on rocks or mangroves. Recorded in the Pleistocene and Pliocene of California and Baja California (Coan & Valentich-Scott, 2012).

Literature: [Felix-Pico et al. (2007a, b), Gemmell et al. (1987: 26), Harry (1985: 150), Hertlein & Strong (1946a: 55-56), Inaba et al. (2004: 22), Keen (1971: 84), Moore (1987: 32), Olsson (1961: 173), Ortiz et al. (2001), Poutiers (1995: 160), Romo & Garcia (2004); Cited by Coan & Valentich-Scott, (2012)].

Ecology

Intertidal and subtidal species of the macroinvertebrates registered in mangrove swamps of bays, coves, mattings, and on the islands of the Baja California peninsula are very diverse. Mangrove swamps diversify the space and offer colonization possibilities to a great number of macroinvertebrate species.

Many ecologists consider that the best role of the mangrove swamp community is ecological, since it is a feeding and reproduction area of many invertebrates. The species are still being inventoried; and most of the phyla are represented by species that are not described, since some are cryptic groups. In the compilations performed by diverse authors in the Gulf of California a total of 213 species is listed: 72 crustaceans, 62 bivalves, 36 gastropods, 14 polychaetes, 5 anemones, 5 sponges, 5 echinoderms, 3 polyplacophorans, 4 chordates, 3 bryozoans and one sipunculan. However, there is a great number of macroinvertebrate species that appear occasionally in mangrove swamps although they belong typically to another type of habitat (Whitmore et al., 2005).

We only found 78 macroinvertebrate species strictly within the mangrove swamps we studied. There are a few species that represent a great economic interest because of their productivity. The bivalve filter-feeders *Anadara tuberculosa* and *Saccostrea palmula* associated with mangrove swamps are their most important components because of their

commercial value and constant exploitation, but unfortunately they have been studied little (Felix-Pico et al., 2011).

HISTORY OF FISHERY

Fishery

At Bahía Magdalena and San Ignacio Lagoon, Baja California Sur, Mexico, the average landings over the last 15 years are 587 t of fresh whole weight. The mangrove black ark lives buried in muddy sediments of mangrove swamps, between tree roots, mainly *Rhizophora mangle* and *Laguncularia racemosa*. This species has high fecundity and good growth rate, reaching commercial size (50 mm shell length in natural conditions) at age two years (Felix-Pico et al., 2009).

We analyzed landing data of *A. tuberculosa* catch statistics recorded from 1991 to 2006 in order to identify trends. Catch data reveal considerable fluctuations but no clear trend towards a decline. No defined seasonal exploitation pattern was observed, but on average the months with the highest catches coincide with the highest reproductive activity. Data show a decrease in black ark density, which might be accounted for by the fishery (Figure 3).

In Mexico, black ark fisheries take place in coastal lagoons along the Gulf of California and Southern Baja California's western coast. In this zone, the major fishing areas are located in the Santo Domingo, Bahía Magdalena and Bahía Almejas estuaries (Baqueiro-Cárdenas, 1980). From 1991 to 1995 (Hernandez-Valenzuela, 1996) estimated catch from 28 boats, which reported an average of 673 metric tons (mt) per year (total black ark weight). MacKenzie, (2001) estimated 85 fishermen operating across the whole State of Baja California Sur (B.C.S.) in 1999. According to official information from the B.C.S. Federal Fisheries Office, the black ark fisheries in Bahía Magdalena are carried out by 40 fishermen.

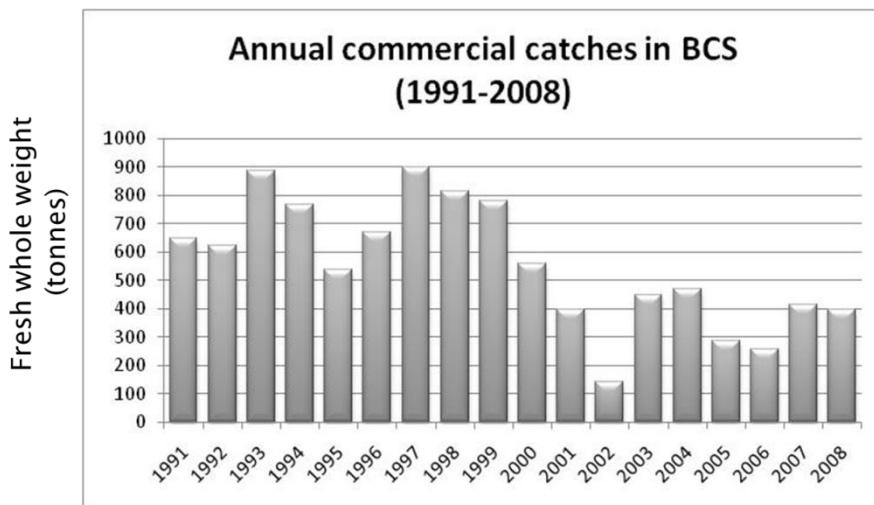


Figure 3. Annual catch of black ark in the Bahía Magdalena region, Baja California Sur, Mexico.

For proper management of the black ark fisheries, minimum sizes have been established to protect the ripening and spawning periods which take place throughout the year, when individuals reach between 23 and 44 mm in shell length (Squires et al., 1975; Baqueiro-Cardenas et al., 1982; Cruz, 1984; Ampie & Cruz, 1989; Borda & Cruz, 2004; Perez Medina, 2005).

Based on the size range recorded from catches, MacKenzie (2001) considers that the exploitation of *A. tuberculosa* is not very heavy in Mexico, since specimens reach 75 mm total length (TL). For comparison, in Colombia and Ecuador they reach 60 mm TL (Borda & Cruz, 2004, Santos & Moreno, 1999) while in Nicaragua the corresponding figure is 65 mm (Ampie & Cruz, 1989). The minimum size of black ark caught ranges between 47 and 60 mm, depending on the country, and although this rule is not strictly applicable, it reflects a low exploitation of juveniles with a slight effect on recruitment. For the Bahia Magdalena-Almejas fishery, Baqueiro-Cardenas et al., (1982) recommended a minimum size of 60 mm TL and a prohibition of commercial exploitation from March 15 to May 12 aimed at protecting the reproductive period; however no such prohibition has become officially mandatory.

There is no commercial harvesting of mangrove oysters, only subsistence and recreational catches.

GROWTH

Black Ark

The parameters derived from the von Bertalanffy growth equation confirm that individuals reach the minimum catch size of 50 mm in a little more than one year. Catch sizes ranged between 40 and 80 mm. It is necessary to consider that the estimation method requires that the population size structure be represented in base samples, but no specimens smaller than 40 mm were found in this study, so K is likely to be overestimated.

Black ark density ranged from 0.5 to 8 arks m⁻² in 2002. According to Baqueiro-Cardenas et al., (1982), the mean black ark density in Bahia Magdalena in the early 1980s ranged between 1 to 18 arks m⁻².

The parameters of growth estimated for each stock were: Los Praditos $L_{\infty} = 93.26$ mm, $K = 0.99$ per year and $t_0 = -0.217$, and Santa Elenita: $L_{\infty} = 88.48$ mm, $K = 0.90$ per year and $t_0 = -0.601$. However, small black arks of less than 40 mm were not well represented. The growth index values ($\phi' = 0.09$ and -0.06 respectively) indicate changes in growth patterns between zones.

The parameters estimated with VBGE were: $L_{\infty} = 81.19$ mm, $K = 1.85$ per year and $t_0 = -0.0002$. However, small black arks of less than 40 mm were not well represented (Figure 4).

Using the same method as in the present work, Borda & Cruz (2004a) estimated values of $L_{\infty} = 86$ mm and $K = 1.17$ for *A. tuberculosa* in Colombia. The growth index values ($\phi' = 0.09$ and -0.06 respectively) indicate changes in growth patterns between zones. It is necessary to consider that the estimation method requires that the population size structure be represented in base samples, but no specimens smaller than 40 mm were found in this study, so K is likely to be overestimated.

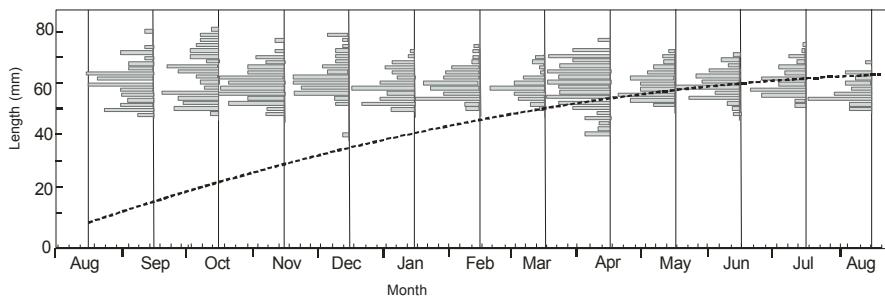


Figure 4.

Monthly length-frequency distributions averaged across a year and a curve describing the growth of black arks in the Bahía Magdalena region, Mexico.

Experimental Aquaculture with Black Ark

There is potential for aquaculture in Mexico, since it is possible to produce black ark spat. The juvenile spat were reared in the laboratory of the Centro de Investigaciones Biológicas del Noroeste (CIBNOR) for 3 months. The spat had a mean shell length of 6 mm. Procedures routinely employed for culture of oysters in Baja California Sur were used for the black ark. Ten thousand seeds were placed into floating trays and deployed in the mangrove channel of the Estero El Conchalito, La Paz, B.C.S. Trays consisted of 50 x 60 cm mesh (2 mm) envelopes (bags) placed inside 57 x 57 x 7 cm polyethylene cages (6-mm mesh). Shell length of 50 to 80 individuals from each batch were measured over four months. Four stocking densities were tested using 60, 120, 300, and 1000 per cage with three replicates used for each batch. At the end of 3 years, the shell length increased between 42 mm (60 ind. density) to only 28 mm (1000 ind. density). Growth results showed a significant growth response to stocking densities. In general, larger sizes were produced when density diminished. The estimated growth parameter K had a low value of 0.32 at highest density and highest value 0.55 at low density. Mortality was determined periodically by counting dead ark shells. The mortality rate varies during the culture period with lower stocking having better survival. Mortality was estimated at about 10% after one year and at high densities more than 15% after one year (Felix-Pico & Holguin-Quiñones, 2002).

Mangrove Oyster

The palmate oyster *Saccostrea palmula* is an import commercial and subsistence resource for people living in small villages along the Pacific coast. This oyster population is attached to mangrove roots or to rocks, especially on reefs exposed to waves, in the intertidal zone to depths of 7 m. Palmate oyster reached commercial size of 50 mm in shell length at one year of age in natural conditions. The oyster growth was observed in the coastal waters of La Paz Bay. To determine the growth 30 oysters were collected monthly. Shell length and weight were recorded. The results showed that the oyster grew from a mean size of 26.2 mm in October 1998 to 37.5 mm in October 1999 (0.9 mm per month) (Figure 5). Specific growth rates of the oyster during this study were found to average 0.031 mm per day between size-frequency of 14-60 mm of shell length. Lower rates were found in the order of 0.002 to 0.004 mm per day (Felix-Pico et al., 2004).

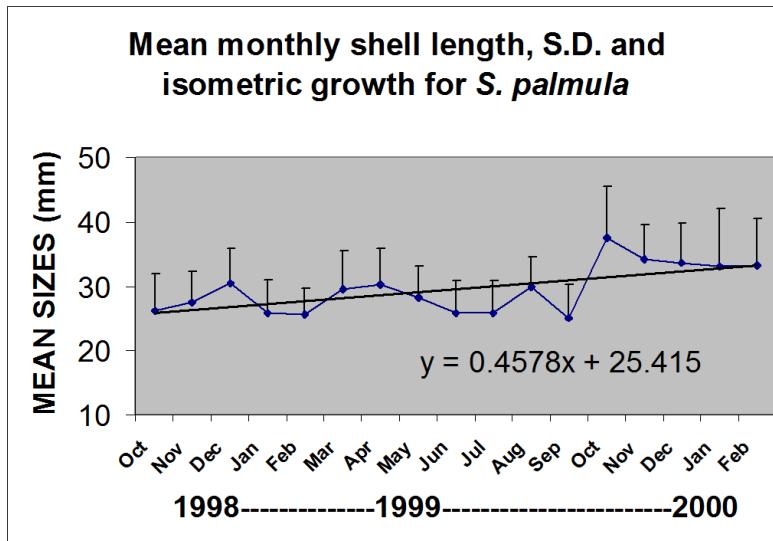


Figure 5. Monthly length-frequency distributions averaged across a year and a curve describing the growth of palmate oysters in the Bahía de La Paz region, Mexico.

SECONDARY PRODUCTION

To estimate secondary productivity in El Mogote estuary, the two most abundant bivalve species were selected: the black ark and the palmate oyster. Within the study area, three estuaries were chosen with a high coverage of mangroves and where occurred. The location of these three estuaries is shown in figure 2 (Felix-Pico et al., 2009).

Within each estuary, sampling was carried out monthly from August 2007 through July 2008, and once every three months from October 2008 through June 2009. All black arks observed during ca. 30-min searches through the estuaries were collected. The area covered by these searches was estimated to the nearest square meter by multiplying the transect length by the 2-m wide search area. Abundance per square meter was estimated and specimens collected were measured (shell length, height, and width) and weighed (with and without shell).

The estimation of secondary productivity in mangrove bivalves was based on biomass increase. Living tissue production for each species was estimated as:

$$P = N B,$$

where P is productivity, N is the density during the sampling period, and B is the increase in average mass over the same period of time.

Mass increment (B) was estimated by determining the rate of individual growth for each species using size-frequency distribution analysis. Measurements of total shell length were grouped into monthly length-frequency distributions (LFD) with 2 mm-wide intervals. The Battacharya method (Bhattacharya, 1967) was used to identify modal LFD groups, which were assumed to represent cohorts or age groups (Sparre & Venema, 1995). Average and standard deviation of length were calculated for each modal group identified.

Modal groups were tracked over time to measure length increase due to the individuals' body growth. These methods were implemented using the FISAT II (Gayanilo et al., 2004) analysis software. Based on this information, growth rates were calculated in $\text{mm}\cdot\text{month}^{-1}$. To estimate individual mass increment in $\text{g}\cdot\text{month}^{-1}$, the relationship between shell-free mass (W) and shell length (L) was described as follows:

$$L = a \cdot W^b,$$

where a and b are model parameters.

In addition, recruitment patterns for each species were estimated using the methods described by Moreau and Cuende (1991) and Pauly (1984). Differences in secondary productivity between estuaries were tested by means of the Kruskal-Wallis rank-sum test, since the data did not follow a normal distribution.

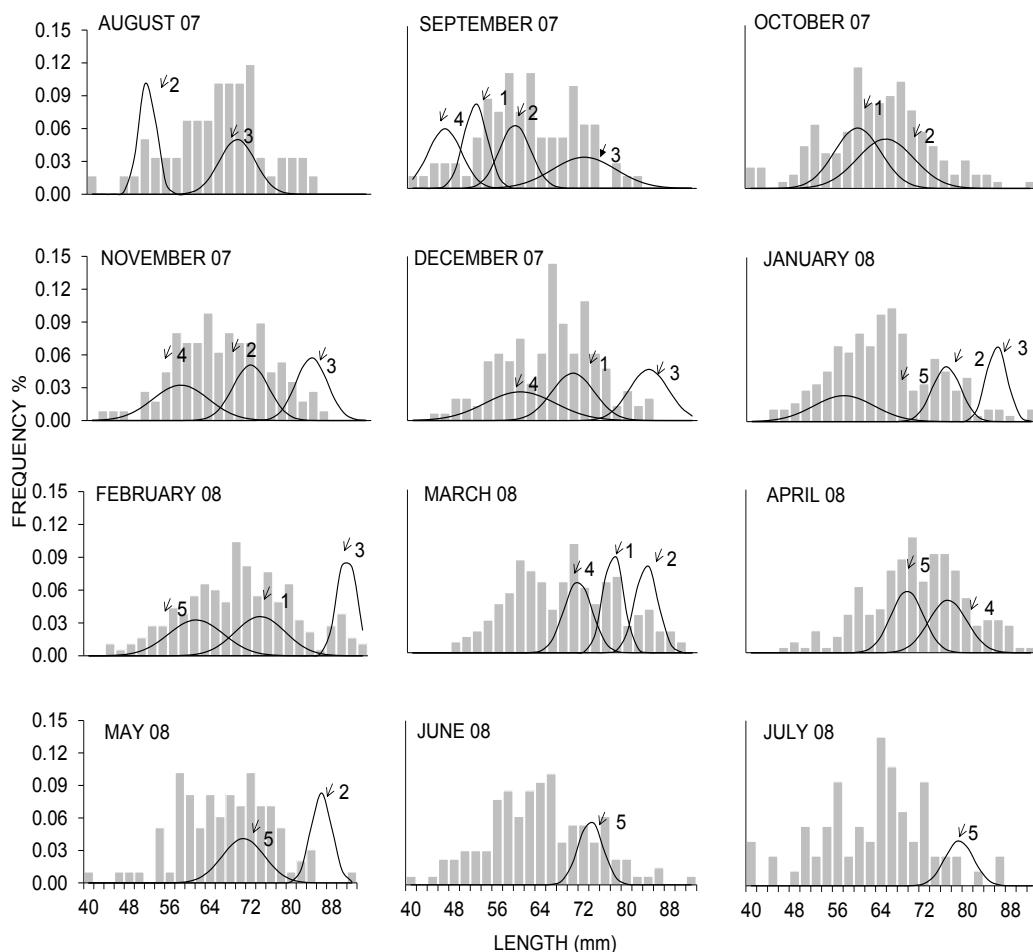


Figure 6. Length frequency distributions of the black ark, *Anadara tuberculosa*, in El Mogote, La Paz Bay, B. C. S. The numbers indicate the modal groups identified and their monthly progression.

Black Ark

A total of 2,412 arks with an average total length of 65.0 mm (range 32.8-97.5 mm) were collected. Total mass ranged from 7.5 to 275.0 g with an average of 86.7g, and the shell-free mass ranged from 2.1 to 50.8 g with an average of 16.0 g. Estimated dry weight ranged between 0.1 and 7.6 g, with an average of 3.4 g.

Average density by estuary was 1.76 ± 0.74 standard deviation (s.d.), 1.28 ± 0.49 (s.d.), and 0.60 ± 0.46 (s.d.) individuals per square meter (ind. m^{-2}) for estuaries A, B, and C, respectively, with an overall average of 1.27 ± 0.75 (s.d.) ind. m^{-2} . Minimum and maximum values were 0.15 ind. m^{-2} (estuary C, October 2007) and 3.45 ind. m^{-2} (estuary A, April 2009). Between August 2007 and July 2008, several modal groups were identified (Figure 6); based on this information, length increase over time was estimated (Figure 7). The estimated growth rate ranged from $3.81 \text{ mm month}^{-1}$ to $5.51 \text{ mm month}^{-1}$, with an average of 4.91 ± 0.64 (s.d.).

The relationship between shell-free mass (W) and shell length (L) was represented by the equation: $W = 0.00006 L^{2.97}$. From this, mass gain rates were calculated, ranging from $2.64 \text{ g month}^{-1}$ to $5.47 \text{ g month}^{-1}$, with an average of $3.67 \text{ g month}^{-1} \pm 1.06$ (s.d.). Biomass production due to body growth was, on average, $4.51 \text{ g m}^{-2} \text{ month}^{-1}$, ± 2.62 (s.d.), but there were significant ($p < 0.05$) differences between estuaries. The highest productivity was recorded in estuary A ($6.17 \text{ g m}^{-2} \text{ month}^{-1}$, ± 2.68 s.d.), followed by estuary B ($4.55 \text{ g m}^{-2} \text{ month}^{-1}$, ± 1.75 s.d.) and, then, by estuary C ($2.14 \text{ g m}^{-2} \text{ month}^{-1}$, ± 1.64 s.d.). Maximum productivity occurred during the spring in estuaries A and C (Figure 8), while peak productivity in estuary B was recorded during the winter (Figure 8b). In 2007-2008 and 2008-2009, the same temporal pattern of productivity was observed in estuaries A and B, with particularly high productivity in 2008-2009. In estuary C, during 2008-2009, the highest productivity was recorded in the winter, showing a shift from 2007-2008, when the highest productivity occurred in the spring (Figure 8b).

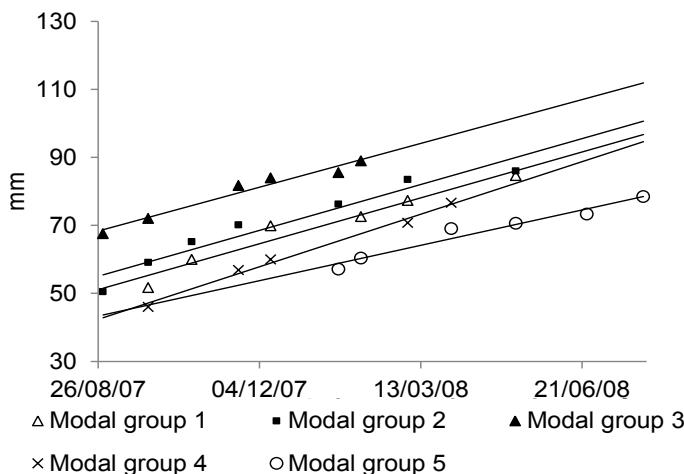


Figure 7. Average size increase for each of the modal groups identified in the length frequency distributions of the black ark *Anadara tuberculosa* in El Mogote, La Paz Bay, B. C. S.

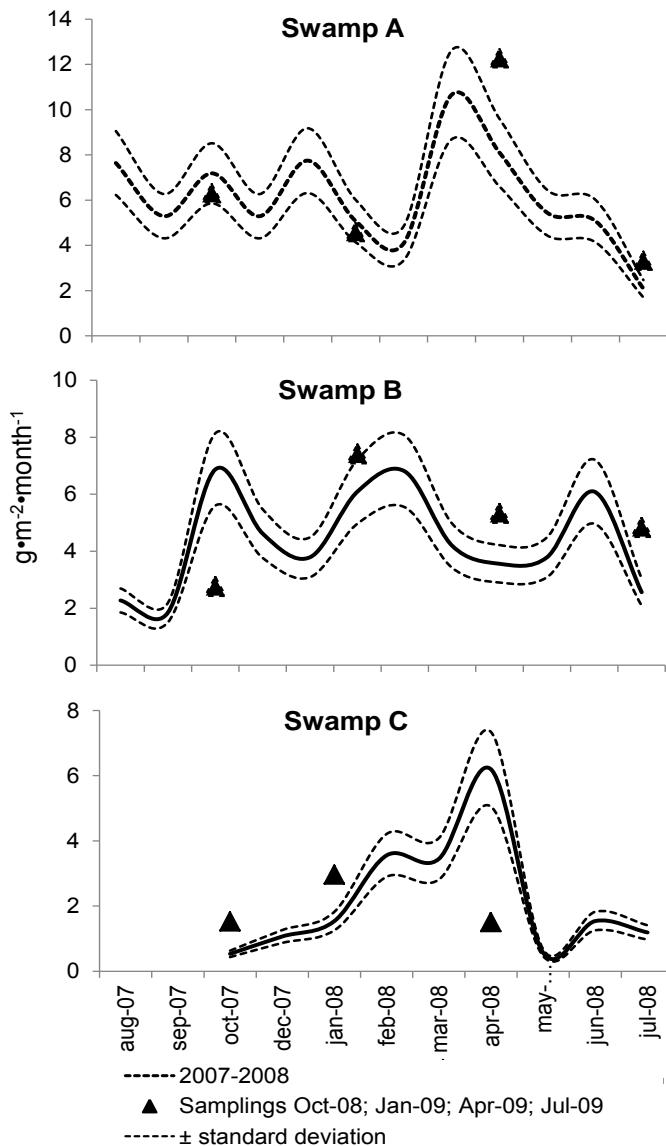


Figure 8. Temporal variation (2007-2009) in secondary productivity due to body growth of the black ark *Anadara tuberculosa* in El Mogote, La Paz Bay, B. C. S.

The estimated population densities of the two species were highly variable, which is likely related to local environmental conditions and survival and recruitment rates. As a response to environmental conditions, these clams show different preferences for and tolerances to temperature, salinity, substrate composition, and tidal levels. The largely different densities of these two species (1.27 ind. m^{-2} for black ark and 510 ind. m^{-2} for mangrove oyster) was due to the three-dimensional use of space by mangrove oysters (from one to 1.5 m) attached to mangrove roots.

The estimated densities of black ark are low compared to those found in Bahia Magdalena, B.C.S., Mexico, with an average of 12 ind. m^{-2} (Baqueiro-Cardenas, 1980), and for *Anadara notabilis* (Röding, 1798) in Venezuela, with 14 ind. m^{-2} (Prieto et al., 1985).

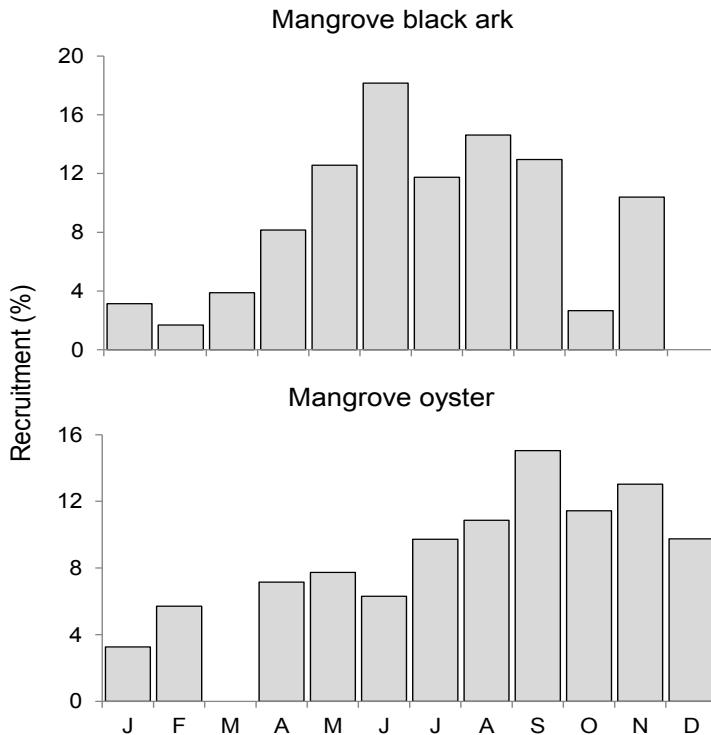


Figure 9. Temporal variation in the recruitment of the black ark *Anadara tuberculosa* and palmate oyster *Saccostrea palmula* in El Mogote, La Paz Bay, B. C. S.

However, densities of black ark found in El Mogote estuary are within the range found in Colombia and Costa Rica (de 0.5 a 4.23 ind. m⁻²,) (Squires et al., 1975; Campos et al., 1990; Borda & Cruz, 2004b). The high density of black ark recorded in Bahia Magdalena, B. C. S., makes this the most important region for black ark harvest in Mexico, in contrast to El Mogote, where no fishery for this species has developed.

The between-estuary variation in secondary productivity is directly proportional to the density values found. The lowest secondary productivity for black ark, recorded in the summer (Figure 9), was related to a higher recruitment in those months (June-September, Figure 9a), when there is a larger proportion of small individuals in the population, a pattern that is similar to, although not as clear as, that of the mangrove oyster (Figure 9b).

Palmate Oyster

As for oysters, we noticed that these were particularly abundant at the mouth of estuary A; hence, it was decided that both the inner (A1) and mouth (A2) zones of the estuary would be sampled. Within each estuary, sampling was carried out monthly from August 2007 through July 2008, and once every three months from October 2008 through June 2009 (Felix-Pico et al., 2009). At each sampling site, four mangrove roots were cut and all specimens present were counted. Abundance per square meter was estimated and specimens

collected were measured (shell length, height, and width) and weighed (with and without shell).

A total of 1,068 oysters were collected, with an average total length of 31.1 mm (range 11.0 to 66.2 mm). Total mass ranged from 1.8 to 59.7 g, with an average of 13.3 g. The shell-free mass ranged from 0.1 to 7.0 g, with an average of 1.3 g, and the estimated dry weight ranged from 0.09 to 0.85 g, averaging 0.37 g (Felix-Pico et al., 2009).

Estimated population density was $510 \text{ ind. m}^{-2} \pm 333$ (s.d.), with a minimum of 80 ind. m^{-2} in estuary A1 in January 2009 and a maximum of $1,720 \text{ ind. m}^{-2}$ in estuary A1 in June 2007. The average density by estuary was 413 ± 409 (s.d.), 573 ± 360 (s.d.), 395 ± 136 (s.d.), and 661 ± 312 (s.d.) ind. m^{-2} for estuaries A1, A2, B, and C, respectively.

From August 2007 to July 2008 several modal groups were identified (Figure 10), from which growth rates were estimated. Growth rates varied between 2.8 and 5.1 mm month⁻¹, with an average of 3.8 mm month⁻¹ ± 1.2 (s.d., Figure 11).

The relationship between shell-free mass (W) and shell length (L) was described by the equation: $W = 0.012 L^{1.31}$. From it, the estimated rate of mass gain due to body growth varied from 0.12 g month⁻¹ to 0.25 g month⁻¹, with an average of 0.18 g month⁻¹ ± 0.05 (s.d.).

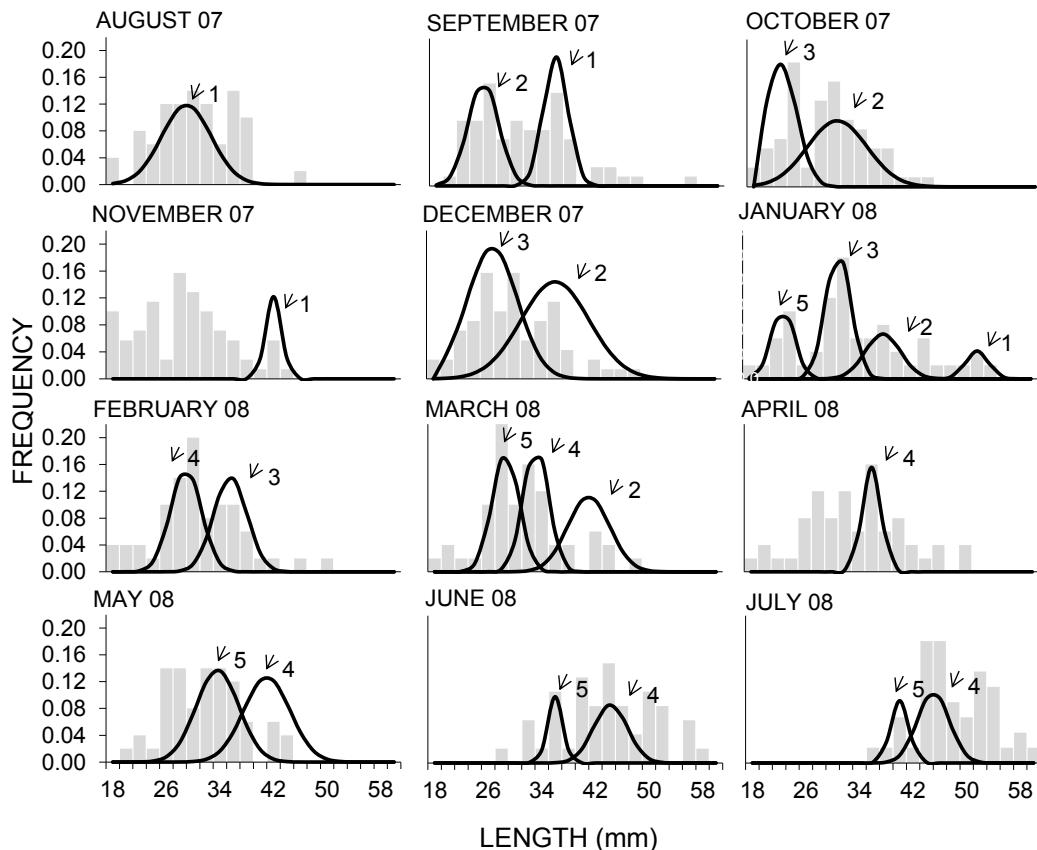


Figure 10. Length frequency distributions of the palmate oyster *Saccostrea palmula* in El Mogote, La Paz Bay, B. C. S. The numbers indicate the modal groups identified and their monthly progression.

The increase in biomass due to body growth was, on average, $97.9 \text{ g m}^{-2} \text{ month}^{-1}$, ± 63.9 (s.d.). The peak productivity was recorded in estuary C ($127.0 \text{ g m}^{-2} \text{ month}^{-1}$, ± 59.9 s.d.), followed by estuary A2 ($110.0 \text{ g m}^{-2} \text{ month}^{-1}$, ± 69.3 s.d.), inlet A1 ($79.4 \text{ g m}^{-2} \text{ month}^{-1}$, ± 78.5 s.d.) and, then, estuary B ($76.0 \text{ g m}^{-2} \text{ month}^{-1}$, ± 26.3 s.d.). Productivity in estuary C was significantly higher than in estuaries A1 and B ($p < 0.05$). The maximum productivity was observed in the summer (October 2008 to July 2009) in estuaries A1, A2, and B (Figure 11), while in estuary C productivity remained constant. Variations in biomass production in the 2007-2008 and 2008-2009 periods showed similar temporal productivity patterns in all four estuaries (Figure 11).

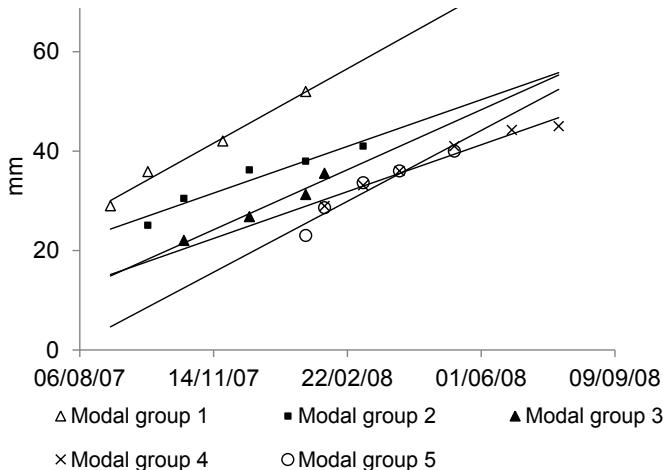


Figure 11. Average size increase for each of the modal groups identified in the length frequency distributions of the palmate oyster, *Saccostrea palmula*, in El Mogote, La Paz Bay, B. C. S.

Recruitment occurred year-round in both species. The main season for black ark was from June through September (Figure 9a), while for mangrove oyster recruitment occurred mainly from September through December (Figure 9b).

The densities of mangrove oyster found in this work were higher than those reported for other bivalves in a range of environments, and are comparable only to those found for *Crassostrea rhizophorae* (Guilding, 1828) (818 ind. m^{-2}) (Prieto et al., 2008) and *Tivela mactroides* (Born, 1778) (485 ind. m^{-2}) in Venezuela (Tata & Prieto, 1991).

Length-frequency distributions for both species were multi-modal, which is consistent with the continuous or prolonged recruitment and reproduction patterns reported for these species (Perez Medina, 2005; Romo-Piñera, 2005). Within each length-frequency distribution, five modal groups were identified in which monthly increases in mean length were observed. This growth estimate was deemed appropriate, as the increase in average length was very similar in the various modal groups, indicating a consistent growth pattern.

It was not possible to compare secondary productivity of these bivalves in the same zone since there are no previous studies on this subject; however, mean secondary productivity estimated for the black ark in this study ($4.51 \text{ g m}^{-2} \text{ month}^{-1}$) lies within the range reported for other bivalves such as *Modiolus squamosus* (Beauperthuy, 1967), *Tivela mactroides* (Born, 1778), *Chione cancellata* (Linnaeus, 1758), *Arca zebra* (Swainson, 1833), and *Argopecten*

purpuratus (Lamarck, 1819), in which productivity ranged between 0.8 and 16.3 g m⁻² month⁻¹ (Tata & Prieto, 1991; Prieto et al., 1998; Prieto et al., 2001; Tarazona et al., 2007).

The estimated productivity of mangrove oyster (97.9 g m⁻² month⁻¹) is of the same order of magnitude as that reported for *Crassostrea rhizophorae* (196.3 g m⁻² month⁻¹) (Prieto et al., 2008).

Figures 8 and 12 show the comparison of monthly estimates of secondary productivity for the first year of the study (2007-2008), and quarterly estimates for the following year (2008-2009). Similar variation patterns are observed throughout, which suggests a period of stability both in terms of environmental conditions and possible effects of human activities such as pollution, habitat degradation, and harvest. Conditions in the estuaries remained hypersaline (>36 PSU) and warm (>21 °C) throughout the two sampling cycles. Additionally, no significant weather changes were observed during the sampling periods.

EXPLOITATION AND MANAGEMENT

In order to explore the potential strategy of rotating fishing areas for the black ark *A. tuberculosa*, an experiment to estimate density recovery after intensive fishing was performed in two mangrove swamps, “El Conchalito” and “El Mogote,” at Ensenada de La Paz, Mexico. On each swamp, one fishing and one repopulation area were delimited. Black ark density (ind. m⁻²) in each area was estimated during October-November 2010. From February 2011 through April 2013, seven samples were done in each zone to record ark densities and size of individuals.

The average density at “El Conchalito” was 1.05 ind. m⁻² (± 0.21 s.d.) with a minimum of 0.70 ind. m⁻² and a maximum of 1.30 ind. m⁻². The effect of fishing was clearly observed during the first and second bimonthly period with densities of 0.93 ind. m⁻². Recovery to near the initial density level (1.3 ind. m⁻²) took seven months. At “El Mogote,” average density was 1.52 ind. m⁻² (± 0.71 s.d.), with a minimum of 0.76 ind. m⁻² and a maximum of 2.52 ind. m⁻². The effect of fishing was also evident with values of 0.8 ind. m⁻² in the following ten months; recovery of initial density levels (around 2.0 ind. m⁻²) took eighteen months. At the repopulation areas, densities increased from 0.83 to 1.32 ind. m⁻² in “El Conchalito” and, ranged from 1.0 to 3.0 ind. m⁻² in “El Mogote.” Length-frequency analyses of black ark showed a decrease in the number of large individuals and the recruitment of smaller ones during the winter months. However, the cause of this is not well understood as black ark reproduction and recruitment behavior is not well known.

The effect of environmental changes on the black ark biology remains unknown. Perez Medina, (2005) found a positive correlation between the reproductive cycle and photoperiod, but failed to find a relationship with temperature or chlorophyll concentration; by contrast, Borda & Cruz (2004c) report that precipitation and surface sea temperature associated with the El Niño phenomenon are related to the rise in catches in an out-of-phase fashion of 17 and 20 months, respectively. Our findings show the highest annual harvest when El Niño occurred (1992-93, 1997-98), but the information available is insufficient to conduct detailed analyses.

OUTLOOK

Given the significance of mangroves as critical habitat for numerous mollusk, crustacean, fish, bird, and mammal species (Lara-Dominguez & Yañez-Arancibia, 1999), the conservation and appropriate management of mangrove ecosystems is of utmost importance.

The findings of this investigation constitute a necessary element for establishing a baseline to evaluate the consequences of the various natural and anthropogenic pressures in the mangrove systems of El Mogote of La Paz Bay, B.C.S., Mexico.

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Chapter 4

REPRODUCTIVE ASPECTS OF MANGROVE BIVALVE MOLLUSKS IN BAJA CALIFORNIA SUR

***Abril Romo-Piñera^{1,2,*}, Federico Andrés García-Domínguez²
and Esteban Fernando Félix-Pico²***

¹Universidad Autónoma de Baja California Sur, Departamento

Académico de Biología Marina, La Paz, Baja California Sur, México

²Centro Interdisciplinario de Ciencias Marinas, Departamento de Pesquerías y Biología
Marina, La Paz, Baja California Sur, México

ABSTRACT

Some reproductive features of *Saccostrea palmula* (Carpenter, 1857) and *Anadara tuberculosa* (Sowerby, 1833) from the Santo Domingo (SD) and El Conchalito (ECO) estuaries were investigated to determine differences related to environmental variability and similarities arising from sharing the same location. Both species display five stages of gonadal development (undifferentiated, development, ripe, spawning and spent), which occur synchronously in females and males.

In both locations, *S. palmula* showed an annual reproductive cycle while *A. tuberculosa* showed a continuous reproductive cycle. Sex ratios significantly departed ($p < 0.05$) from 1:1 for both species in ECO, but not for *S. palmula* in SD. For *S. palmula*, a significant correlation ($p < 0.05$) between mean maturity index (MMI) and temperature was found at both sites, and between MMI and chlorophyll-a concentration in ECO. For *A. tuberculosa* no significant correlation ($p > 0.05$) between MMI and either temperature or chlorophyll-a concentration was observed. *S. palmula* behaves as an opportunistic species in SD, but as a conservative species in ECO, as does *A. tuberculosa*. This seems to be related to chlorophyll-a concentration at each site.

* Corresponding author: Romo-Piñera Abril. Universidad Autónoma de Baja California Sur, Departamento Académico de Biología Marina, Carretera al Sur km. 5.5, Col. Mezquitito, La Paz, Baja California Sur, 23080, México. Centro Interdisciplinario de Ciencias Marinas, Departamento de Pesquerías y Biología Marina, Ave. IPN s/n, Col. Playa Palo de Santa Rita, A.P. 592, La Paz, Baja California Sur, 23096, México. E-mail: akromo@uabcs.mx.

INTRODUCTION

Mangrove systems contribute to the flow of energy between terrestrial and marine systems and are one of the most important ecosystems in coastal zones (Aburto-Oropeza et al. 2008). They provide an environment suitable for the development of a number of species, including several with economic and nutritional value, such as bivalve molluscs. In Baja California Sur, Mexico, bivalve species inhabiting these important systems, including the black ark, *Anadara tuberculosa* (Félix-Pico et al. 2011), and the mangrove oyster, *Saccostrea palmula* (Baqueiro-Cárdenas et al. 1982). The black ark, *A. tuberculosa*, ranges from the San Ignacio Lagoon in Baja California Sur, Mexico to Punta Pirua, Peru (Coen and Valentich-Scott 2012). It inhabits mangrove swamps (Coen and Valentich-Scott 2012), where it lives buried in the mud, from the intertidal zone down to a 5 m depth (Poutiers 1995). The black ark fishery constitutes an important economic activity in Bahía Magdalena (Félix-Pico et al. 2009), where an average of 673 tons is harvested each year (Hernández-Valenzuela 1996). The mangrove oyster, *S. palmula*, ranges from the San Ignacio Lagoon in Baja California Sur (including the Gulf of California) to Bayovar, Peru, it is also found in Cocos Island, Costa Rica, and the Galapagos Islands, Ecuador (Coen and Valentich-Scott 2012).

This species lives firmly attached to the roots of the mangrove *Rhizophora mangle* (Félix-Pico et al. 2011) and is used by man both as food and for handcraft manufacture (Baqueiro-Cárdenas et al. 1982, Holguín-Quiñones and González-Pedraza 1994). Reproductive cycles in bivalves are closely related to endogenous factors, such as genotype and neuroendocrine cycles (Barber and Blake 1991, Shumway 1996, Thompson et al. 1996), as well as to environmental factors (Meneghetti et al. 2004) such as temperature, food availability (Barber and Blake 1991), salinity and tidal cycles (Andrews 1979, Sastry 1979, Mackie 1984). As a result, bivalve species display a number of reproductive patterns, which may vary even within the same species in response to year-to-year or site-to-site environmental variations; besides, different species living in the same locality may share a similar reproductive pattern. Studies on the reproductive patterns of bivalve species inhabiting mangrove systems along the east and west coasts of Baja California Peninsula are scarce.

In this chapter, we describe the reproductive patterns of *A. tuberculosa* and *S. palmula* living in El Conchalito (ECO) and Santo Domingo (SD) estuaries, both located at the southern portion of the Baja California Peninsula.

MATERIALS AND METHODS

Study Area

Sampling was conducted in the SD estuary located at Bahía Magdalena-Almejas lagoon complex, and in the ECO estuary located in Ensenada de La Paz (Figure 1). Specimens of *S. palmula* were collected at ECO from October 1998 to October 1999 and at SD from December 1997 to December 1998. Specimens of *A. tuberculosa* were only collected at ECO from October 1998 to October 1999. Approximately 20 specimens of each species were collected monthly at each site. Water temperature was recorded at each site using a HORIBA U-10 water quality analyzer. Additionally, a 2L water sample was filtered through a 200µm

sieve to remove large particles. Then, three 1L samples were filtered through 47 mm diameter Whatman GF/F filters. Chlorophyll-*a* concentration was determined in these samples, as an estimate of food availability. Chlorophyll-*a* was extracted with acetone and its concentration was determined using a modified fluorometric method (Parsons 1984). In the laboratory, the following parameters were recorded for each specimen: shell length (± 0.1 mm), total weight and shell-free weight (± 0.1 g).

Sex Ratio

Since *A. tuberculosa* and *S. palmula* do not display sexual dimorphism, the sex of each specimen was determined based on a microscopic examination of the gonad.

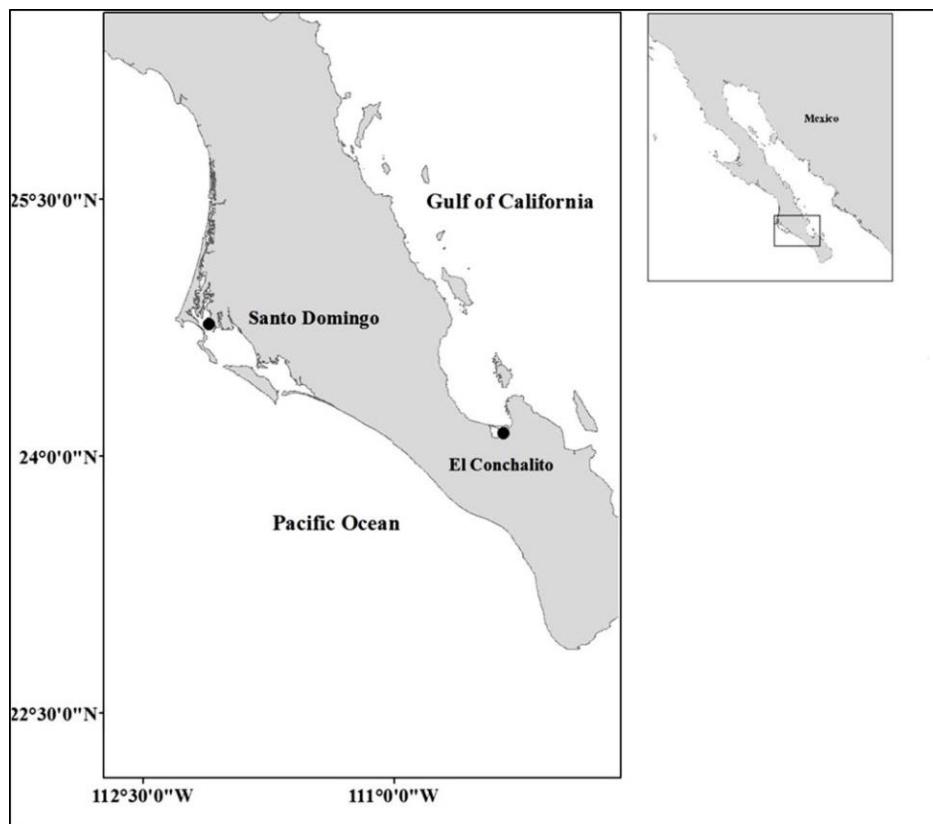


Figure 1. Sampling was conducted in the Santo Domingo estuary located at Bahía Magdalena–Almejas lagoon complex, and in the in El Conchalito estuary located in Ensenada de La Paz.

Sex ratios were calculated for each month and for the full data set to determine whether these departed from a 1:1 ratio. Variations in sex ratio were analyzed using a Yates' χ^2 test (Zar 1996).

Histological Analysis and Mean Maturity Index

S. palmula and *A. tuberculosa* have diffuse gonads i.e., the gonad is embedded in connective tissue between the acini of the digestive gland, forming the visceral mass. A cross-section of the visceral mass (including the gonad) was obtained from each specimen and fixed in 10% formalin. The conventional histological technique was used (Humanson 1979), which consists of dehydrating the tissue sample through a sequence of alcohols of increasing concentration, clarifying it with Hemo-De®, and finally embedding it in Paraplast-Xtra®. Tissue sections of 7 µm were placed on slides, stained with hematoxylin-eosin and examined under the microscope.

To characterize the gonad development stages of *S. palmula* and *A. tuberculosa*, we used hystomorphological criteria previously used by other authors for the same species (Cuevas-Guevara and Martínez-Guerrero 1979) or genus (Power 2002), respectively. Both criteria consider five stages: undifferentiated, development, ripe, spawning and spent. For each species, and based on the histological analysis of gonadal tissue, the Mean Maturity Index (MMI) (Grant and Tyler 1983) was calculated.

To this end, the number of specimens at each gonad development stage in a given month was multiplied by the value assigned to each stage (spent = 1; development = 2; ripe = 3; spawning = 4); the product was then divided by the total number of organisms collected in that month (Seed 1976, Kautsky 1982). MMI varies from 1—when the sampled population is in post-spawning stage— to 4 —when the population is spawning.

Reproductive Cycle

To describe the reproductive cycle of these species, the relative frequencies of the different gonad developmental stages in each month were calculated and plotted for each species. To understand the relationship between the environmental variables temperature and Chlorophyll-*a* concentration and the reproductive cycle of these species, a Spearman's rank correlation analysis was performed, using MMI as a quantitative expression of the reproductive cycle.

The STATISTICA v6.0, StatSoft software was used for all statistical analyses. A significance level $\alpha = 0.05$ was used in all statistical tests.

RESULTS

Sex Ratio

A total of 174 specimens of *S. palmula* were collected at SD estuary. Specimen length ranged from 28.8 to 69.9 mm ($45.9 \text{ mm} \pm 8.3 \text{ mm}$). Forty six females (26.4%), 58 males (33.3%) and 70 (40.2%) undifferentiated specimens were recorded.

The overall sex ratio was not significantly different from 1:1 ($\chi^2 = 1.3, p > 0.05$), but the monthly ratios were significantly different ($p < 0.05$), except in August (Table 1).

Table 1. Percentage of females and males of *Saccostrea palmula* in the Santo Domingo estuary

Month	Females (%)	Males (%)	χ^2	Sex ratio
December 97	0	100	100	0:1
January 98	0	100	100	0:1
February 98	0	0	0	0
March 98	0	100	100	0:1
April 98	100	0	100	-
May 98	28.57	71.42	18.37	0.4:1
June 98	60.00	40.00	4.01	1.5:1
July 98	41.17	58.82	3.12	0.7:1
August 98	52.63	47.36	0.28	1.1:1
September 98	71.42	28.57	18.37	2.5:1
October 98	18.18	81.81	40.50	0.2:1
November 98	0	100	100	0:1
December 98	0	100	100	0:1
TOTAL	44.23	55.76	1.34	0.7:1

Numbers in bold are significantly different from the 1:1 sex ratio (chi-square, $p \leq 0.05$).

The proportion of males was higher than females in most months except April, June, August and September 1998, when the ratio was inverted (Table 1). In February 1998, all the specimens collected were undifferentiated.

A total of 255 specimens of *S. palmula* were collected at ECO estuary. Specimen length ranged between 14 and 60.2 mm (mean = 28.4 mm \pm 6.2 mm). Thirty four females (14.9%), 86 males (37.9%) and 107 (47.1%) undifferentiated specimens were recorded. The overall ($\chi^2 = 18.8$, $p < 0.05$) and monthly ($p < 0.05$) sex ratios departed significantly from 1:1 with a larger proportion of males relative to females, except for October, 1998 when a larger proportion of females was registered (Table 2).

A total of 194 specimens of *A. tuberculosa* were collected at ECO estuary. Shell length ranged between 37.1 and 78.0 mm (mean = 56.51 mm \pm 8.8 mm). One hundred and eight specimens were females (55.6%), 69 males (35.5%) and 17 (8.7%) undifferentiated. The overall sex ratio departed significantly from 1:1 ($\chi^2 = 4.8$, $p < 0.05$); monthly sex ratios for October to December 1998 and February to July 1999 were also significantly ($p < 0.05$) different from 1:1, with a larger proportion of females than males, except for July (Table 2).

Histological Analysis, Reproductive Cycle and Gonad Index

The five stages of gonadal development previously described for *S. palmula* and *A. tuberculosa* were observed. Based on the histological examination, a general criterion was used to describe the gonad stages in both species (Table 3).

Table 2. Percentage of females and males, of *Saccostrea palmula* and *Anadara tuberculosa* in El Conchalito estuary

Month	<i>Saccostrea palmula</i>				<i>Anadara tuberculosa</i>			
	Females %	Males %	χ^2	Sex Ratio	Females %	Males %	χ^2	Sex Ratio
October 98	61.11	38.88	4.94	1.5:1	75.00	25.00	25.01	3:1
November 98	22.22	77.77	30.87	0.28:1	66.66	33.33	11.12	2:1
December 98	0	100	100	0:1	68.42	31.57	13.58	2.1:1
January 99	0	100	100	0:1	46.66	53.33	0.45	0.8:1
February 99	0	100	100	0:1	66.66	33.33	11.12	2:1
March 99	16.66	83.33	44.45	0.2:1	71.42	28.57	18.37	2.5:1
April 99	20.00	80.00	36.01	0.2:1	81.25	18.75	39.07	4.3:1
May 99	0	100.00	100	0:1	60.00	40.00	4.01	1.5:1
June 99	0	100.00	100	0:1	70.00	30.00	16.01	2.3:1
July 99	30.00	70.00	16.01	0.4:1	36.36	63.63	7.44	0.5:1
August 99	31.25	68.75	14.07	0.4:1	43.75	56.25	1.57	0.7:1
September 99	35.29	64.70	8.66	0.5:1	41.66	58.33	2.78	0.7:1
October 99	31.25	68.75	14.07	0.4:1	50	50.00	0.01	1:1
TOTAL	28.33	71.66	18.78	0.3:1	61.01	38.98	4.86	1.5:1

Numbers in bold depart significantly from a 1:1 sex ratio (chi-square, $p < 0.05$).

Table 3. Stages of gonad development in *Saccostrea palmula* and *Anadara tuberculosa*

Stages	Female	Male
Undifferentiated	Connective tissue fills the spaces between empty and collapsed acini. As no sex cells are observed, it is not possible to determine the specimen's sex.	
Development	Growing oocytes are observed within follicles, attached to the walls. Gaps between follicular walls are occupied by connective tissue	Both spermatogonia and spermatocytes can be found within the seminal tubules. The area occupied by connective tissue begins to decrease.
Ripe	Follicles are filled with mature oocytes. There is little space between follicle walls. Some follicles with oocytes attached to the follicular walls can be observed, indicating that gametogenesis progresses.	Seminal tubules are filled with spermatozooids. There is little space between the seminal tubules. Connective tissue is scarce.
Spawning	Empty spaces can be observed inside follicles and between their walls. Fewer mature oocytes can be found. Torn follicular walls can be found—clear indication of spawning.	There is an evident decrease in the amount of spermatozooids inside seminal tubules. Empty spaces within seminal tubules – left by recently expelled sperm – are evident.
Spent	Follicles are smaller and their walls are torn. Phagocytes and residual oocytes can be observed. Connective tissue is present again.	The torn walls of seminiferous tubules and some residual spermatozooids are observed. The size of seminiferous tubules decreases.

In both species, relative frequencies of gonad development stages throughout the cycle were obtained pooling females and males together, because no differences in the temporality pattern between sexes were observed. Undifferentiated individuals of *S. palmula* at SD were recorded from December 1997 through May 1998 and from September to December 1998. All the specimens collected in February were in this stage. The development stage was observed in December 1997 and 1998, January, and from April to September 1998. The highest percentage (80%) of specimens in the development stage was observed in June 1998. Ripe specimens were observed from June to September 1998; the highest frequency (30%) was observed in July 1998.

The spawning stage occurred in January 1998 and from June to October 1998. The highest percentage (50%) of specimens in the spawning stage was recorded in August and September 1998. Post-spawning specimens were registered in March and from August to December 1998; the highest frequency (40%) of post-spawning specimens occurred in October 1998 (Figure 2A).

The reproductive cycle of *S. palmula* in ECO included undifferentiated specimens throughout the entire cycle, except for September 1999; the highest frequency (> 50%) of undifferentiated specimens was observed from December 1998 to June 1999. The development stage occurred from February to October 1999; the highest percentage (58%) of specimens in the development stage was found in July 1999. The ripe stage was only observed in August, September and October 1999; the highest frequency of ripe specimens occurred in August (42%) and September (47%) 1999. The spawning stage was observed from October to December 1998 and from August to October 1999. The highest percentage of spawning specimens was found in October, 1998 (78%) and October, 1999 (66.6%). The post-spawning stage was noted from October to December 1998 and in January and February 1999 (Figure 2B). The reproductive cycle of *A. tuberculosa* in ECO showed a very low frequency (< 15%) of undifferentiated specimens in December 1998, February, June and September 1999; the highest frequency was found in May (41%) and July (26%). The development stage was observed in December 1997, February, June and September, always in low frequency (< 15%), and in May (41%) and July (26%). The development stage was present in October, December 1998, January and April 1999 always in low frequency (< 15%). From May to August and in October 1999, the frequency of specimens in the development stage increased (> 29%). The ripe stage was present from October 1998 to March, August and September 1999; the highest frequency (69%) of ripe specimens was observed in September. The spawning stage was observed throughout the cycle except in June 1999. The highest frequency (85%) of spawning specimens was observed in March 1999. The post-spawning stage was present from November 1998 to July 1999, peaking in November (73%), February (40%) and April (50%) (Figure 2C).

According to MMI values, the specimens of *S. palmula* collected at SD went through a stage of low gonadal activity (≤ 0.50) from December 1997 to March 1998, followed by the onset of gonad development, ranged from 0.80 in April to 0.87 in May. Subsequently, the spawning stage started in June showing a MMI value of 2.3 and lasted in October with 1.3, when MMI decreased. A new cycle rise with little gonadal activity between November (0.23) and December 1998 (0.26) (Figure 3A).

The specimens of *S. palmula* collected in ECO showed low MMI values in January (0.13) and May (0.42) when most of the population had little gonadal activity; this began to increase

in June (0.77), with a higher frequency of specimens in the development stage and the appearance of ripe and spawning specimens starting in August (2.21).

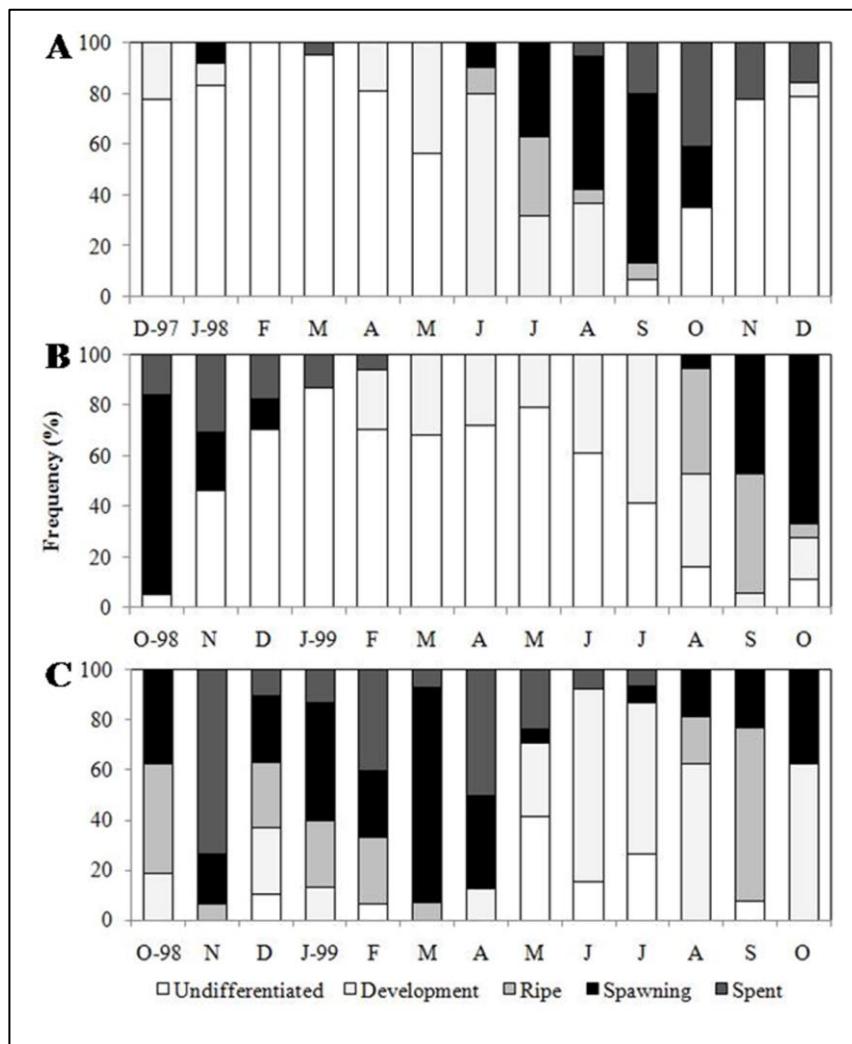


Figure 2. Monthly percent frequencies of the different gonad stages of *Saccostrea palmula* throughout the study period in Estero Santo Domingo (A), Estero El Conchalito (B) and of *Anadara tuberculosa* in Estero El Conchalito (C), B.C.S. Mexico.

MMI peaked (3.41) in September, coinciding with the highest frequency of ripe and spawning specimens.

In October 1998 and 1999 lower MMI values were found (3.15 and 3.16, respectively), denoting the presence of post-spawning and undifferentiated specimens (Figure 3B).

The MMI values for *A. tuberculosa* in ECO showed that the lowest values (< 2) were recorded in November 1998, May, June and July 1999, when most specimens were in the post-spawning or development stages. Highest values (> 2.5) were registered in October 1998, January, March and April 1999, and from August to October 1999, denoting the high frequency of spawning and some ripe specimens (Figure 3C).

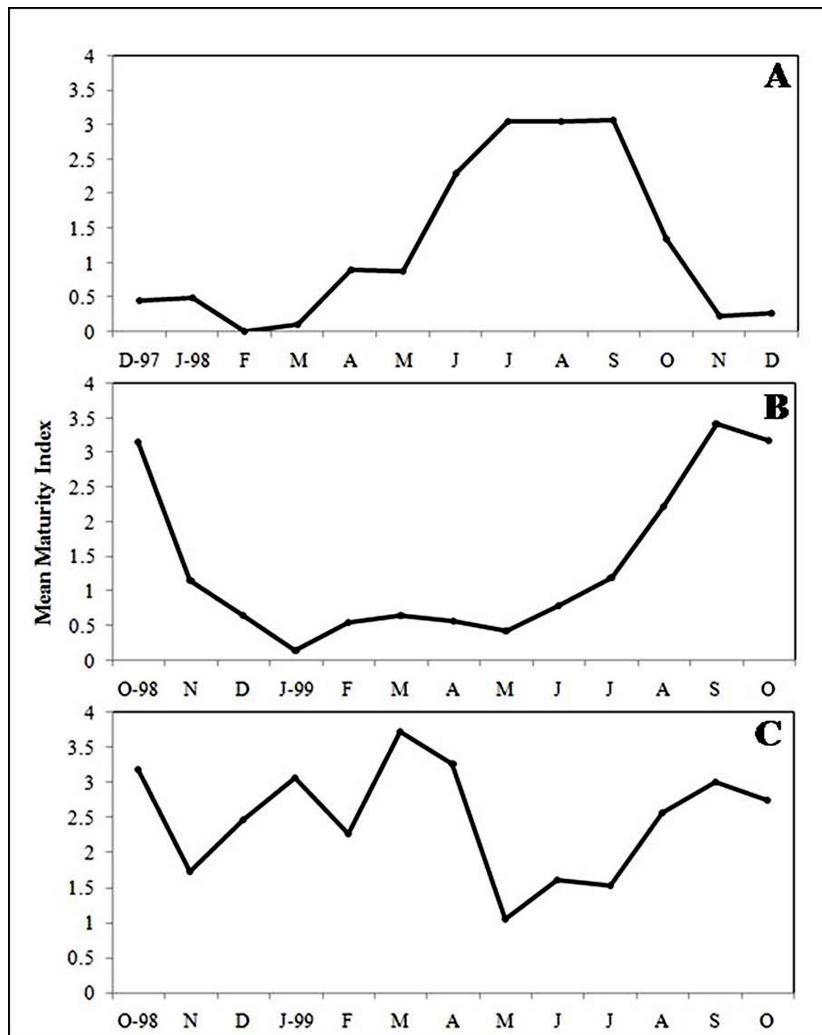


Figure 3. The mean maturity index for gonads (male and female) from *Saccostrea palmula* collected at Estero Santo Domingo (A), Estero El Conchalito (B) and from *Anadara tuberculosa* collected at Estero El Conchalito (C), B.C.S. Mexico.

Water temperature in SD ranged from 18.7°C in December 1998 to 29.4 °C in August 1998 (Figure 4A). Chlorophyll-*a* concentration ranged from 0.50 mg/m³ in June to 9.4 mg/m³ in May 1997 (Figure 4A). MMI values for *S. palmula* were positively correlated with both environmental variables: temperature ($R = 0.74, p = 0.003$) and chlorophyll-*a* concentration ($R = 0.64, p = 0.017$). In ECO the water temperature ranged from 16.3 °C in January to 28.1 °C in June (Figure 4B).

Chlorophyll-*a* ranged between 0.30 mg/m³ to 0.81 mg/m³ in May and June 1997, respectively (Figure 4B). A significant positive correlation between MMI for *S. palmula* and temperature ($R = 0.76, p = 0.002$) was found, but not between MMI and chlorophyll-*a* concentration ($R = 0.06, p = 0.83$). No positive correlation between temperature ($R = 0.002, p = 0.99$) and chlorophyll-*a* ($R = -0.16, p = 0.58$) and MMI for *A. tuberculosa* was found either.

DISCUSSION

Sex Ratio

Most bivalve species are dioecious and usually have similar numbers of males and females (Gosling 2003). In this study, we registered that *S. palmula* displayed a 1:1 sex ratio in SD which has been reported in a population from Puntarenas, Costa Rica (Cabrera-Peña et al. 2001).

However, in ECO, a significantly higher proportion of males (0.3F:1.0M) was observed. In this regard, Cabrera-Peña et al. (2001) mentions that males are more abundant in the size classes between 14.0 and 40.5mm, while females are more abundant between 40.5 and 67.0 mm.

In this study, the average size of the ECO population was smaller (28.4 ± 6.2 mm) compared to the SD population (45.8 ± 8.3 mm). Taking this difference in average size into account, this difference in sex ratio may be due to the absence of small specimens in our sample from ECO. In this regard, Arellano-Martínez et al. (2006) considers that the sex ratio is a species-specific characteristic that is not influenced by environmental conditions.

For the *A. tuberculosa* studied from ECO, we found a 1.5F:1.0M sex ratio, which differs from the 1:1 ratio reported by Pérez-Medina (2005) for SD and by Cruz (1984) for a population from Puntarenas, Costa Rica. Cole (1954) mentioned that variations in sex ratio are a strategy to increase the reproductive success in low-density populations. However, *A. tuberculosa* population examined here by does not show evidence of over exploitation (Félix-Pico et al. 2009). On the other hand, Yoyole (1975) mentions that *Anadara senilis* males are most abundant in the size classes between 5 and 17 mm, while females are most abundant in size classes larger than 20 mm. In this regard, the size of specimens examined in this study ranged between 37.1 mm and 78 mm (56.51 ± 8.8 mm), which may account for the larger proportion of females observed in our sample.

Reproductive Cycle and Gonad Index

The histological analysis showed that, in both localities, the gonad development of *S. palmula* males and females synchronously. This synchrony has also been reported for other populations of *S. palmula* from the State of Nayarit, Mexico (Cuevas-Guevara and Martínez-Guerrero 1979) and Puntarenas, Costa Rica (Cabrera-Peña et al. 2001).

A. tuberculosa males and females also displayed a synchronous gonadal development, and previous studies in SD had reported a similar behaviour (Pérez-Medina 2005, García-Domínguez et al. 2008). In this respect, some authors suggest that this synchrony is related to the periodicity of spawning (Giese and Pearse 1974, Barber and Blake 1991, Román et al. 2001).

It is known that differences in the reproductive cycle may occur within the same species and even within the same population, depending on latitude and the local environmental conditions (Andrews 1979, Sastry 1979, Barber and Blake 1991). In SD, *S. palmula* went through the spawning stage from June to October and showed a brief release of gametes in January. In ECO, spawning occurred from October to December 1998 and from August to

October 1999. On the other hand, Cuevas-Guevara and Martínez-Guerrero (1979) observed the spawning stage from June to November in Nayarit, whereas Cabrera-Peña (2001) reported gamete release throughout the year in Puntarenas, Costa Rica, with two spawning peaks in November and January. In this respect, Giese and Pearse (1974) mention that in the northern hemisphere, the reproductive cycle of bivalves tends to be longer at the southern end of their distribution range.

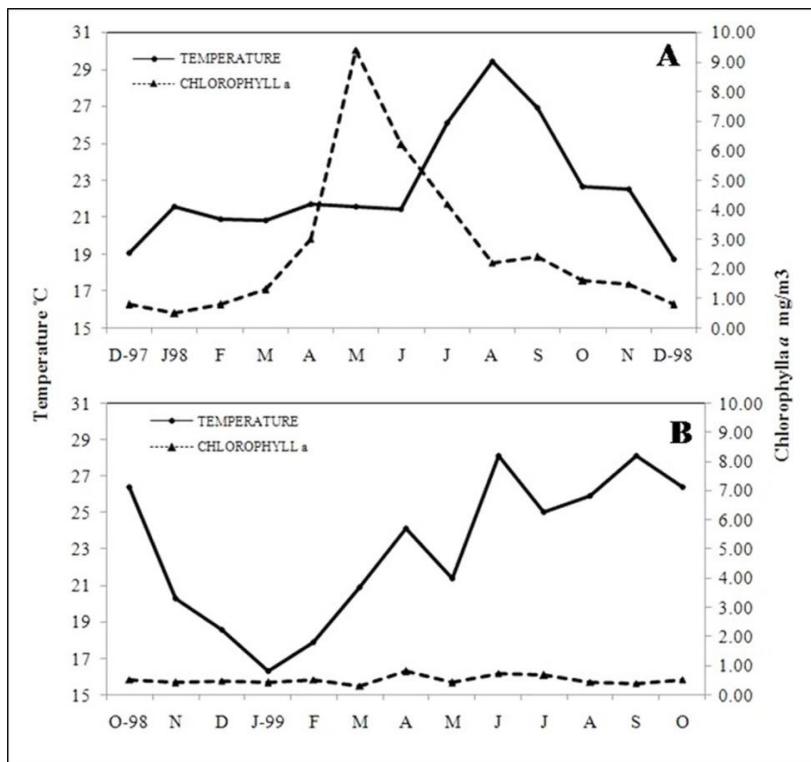


Figure 4. Temporal variation of the bottom seawater temperature and *chlorophylle-a* at Estero Santo Domingo (A), Estero El Conchalito (B), B.C.S. Mexico.

This is consistent with the hypothesis put forward by Orton (Orton 1920), who states that the reproductive stage of marine species is related to their latitudinal distribution as, for the same species, its longest breeding season occurs at the southern end of its distribution, and the shortest one at its northernmost end of its distribution range.

By contrast, *A. tuberculosa* in ECO spawned throughout the study period, except for June, and the highest percentages of spawning specimens were recorded in March 1998 (85.7%) and January 1998 (46.6%). On the other hand, Baqueiro-Cárdenas et al. (1982) observed spawning specimens in April and August 1978 and in January 1979 in La Paz Bay, Mexico. In Bahía Magdalena, Mexico, (García-Domínguez et al. 2008) observed two spawning periods in 1998, one from January to March and the other one from August to December. In the same locality, *A. tuberculosa* displayed the largest release of gametes from May to June and in November-December in the year 2000 (Pérez-Medina 2005). Reproductive cycles of marine bivalves can be annual, semi-annual or continuous, allowing reproductive periods to match environmental conditions suitable for offspring development

and growth (Sastry 1979). In this sense, the results from our study suggest that *S. palmula* shows an annual reproductive cycle in SD and ECO estuaries, while *A. tuberculosa* has a continuous reproductive cycle in SD.

Water temperature has traditionally been considered as a key environmental factor controlling the reproductive cycles of marine bivalves (Newell et al. 1982). For several bivalve species, rises in water temperature have been reported to trigger gametogenesis and spawning (Ceballos-Vázquez et al. 2000), (Machreki-Ajmi et al. 2012). On the other hand, records of spawning at different temperatures for some species have led some authors to suggest that it is a sudden rise in temperature, rather than its absolute value, what triggers spawning (Martínez-Castro and Vázquez 2012). In this study, the simultaneous increase observed in temperature and MMI for *S. palmula*, along with the results from the correlation analyses support this hypothesis. However, in other species such as *Pinctada mazatlanica* (García-Domínguez et al. 1996) and *Ostrea edulis* (Joyce et al. 2013), no straight forward association between water temperature and gonad development has been identified, as was the case for *A. tuberculosa* in this and previous studies (Pérez-Medina 2005, García-Domínguez et al. 2008). In addition to temperature, another factor that regulates the reproductive cycle in bivalves is food availability (Sastry 1979, Gabbott 1983, Barber and Blake 1991, Eversol 2001, Gosling 2003) as spawning can be timed to coincide with conditions that are nutritionally favourable for larval development (Jaramillo et al. 1993).

MMI for *S. palmula* in SD showed a significant correlation with the increase in chlorophyll-*a* concentration and the percentage of ripe organisms. This correlation has also been reported for other species such as *Ensis macha* (Avellanet et al. 1996) and *Chlamys amandi* (Jaramillo et al. 1993).

In contrast, no clear trend in this respect has been observed in species such as *Pteria sterna* (Hernández-Olalde et al. 2007), *Nodipecten subnodosus* (Arellano-Martínez et al. 2004) and *Megapitaria squalida* (Romo-Piñera 2010), and neither has it been observed in *S. palmula* or *A. tuberculosa* populations in ECO. This might suggest that both species follow a conservative reproductive strategy, i.e. the energy required for gonad development is obtained from the energy stored in their tissues (Bayne 1976). However, *S. palmula* in SD behaves as an opportunistic species, using the high concentration of food available in the medium for gonad development (Bayne 1976). In this regard, some authors suggest that there may be local or inter-annual differences between species, and even within the same species, in the nutrient storage cycle and reproduction, in response to environmental conditions (Bayne 1976, Kang et al. 2000).

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Chapter 5

COMPOSITION AND ZOOGEOGRAPHY OF FISHES IN MANGROVE ECOSYSTEMS OF BAJA CALIFORNIA SUR, MÉXICO

**Adrián F. González-Acosta¹, Gorgonio Ruiz-Campos²
and Eduardo F. Balart³**

¹Centro Interdisciplinario de Ciencias Marinas-IPN,
Playa Palo de Santa Rita, La Paz, México

²Colección Ictiológica, Facultad de Ciencias, Universidad Autónoma de Baja California,
Ensenada, Baja California, México

³Colección Ictiológica & Laboratorio de Necton y Ecología de Arrecifes, Centro
Investigaciones Biológicas del Noroeste,
Colonia Playa Palo de Santa Rita. La Paz, Baja California Sur, Mexico

ABSTRACT

An updated systematic and zoogeographical checklist of fishes associated to mangrove estuaries of Baja California Sur (BCS) is provided in the present chapter. This checklist is based on records of literature and of verified voucher specimens from museums. Fish diversity from mangrove estuaries of BCS is composed by 228 species, 147 genera, 72 families, 23 orders, and two classes. Fish fauna of BCS is dominated by the class Actinopterygii (208 spp.) and the order Perciformes (125 spp.) with four families (Haemulidae, Sciaenidae, Gobiidae and Carangidae) representing more than 50% of the fish species into the order. Biogeographically, the ichthyofauna showed a mixture of species from derivation of temperate and tropical regions, with high affinities to the Cortez province (83.7%) followed by Mexican (73%) and Panamic (70.3%) provinces; however, more than 61.2% of the species extend their geographic ranges into the San Diegan province. Other species (6.6%) have circumtropical (13 spp.), amphiamerican (1 sp.) and amphipacific (1 sp.) affinities; and four species are from exotic origin (*Poecilia butleri*, *P. reticulata*, *Sparus aurata*, and *Tilapia cf. zillii*). Three taxa endemic to the Gulf of California were also recorded (*Aruma histrio*, *Barbulifer*

* E-mail address: aacosta@ipn.mx.

pantherinus and *Gobiosoma chiquita*). Bahía de la Paz (62.7%) and Bahía Magdalena (55.7%) were the most extensive mangrove ecosystems in BCS, both contributing with the highest number of species. High fish diversity reported here, reflects the important role of the mangrove estuary ecosystems as primary habitat or nursery (feeding and spawning) for many marine species, contributing with the growth, recruitment and conservation of species with commercial or ecological value.

Keywords: Fish diversity, Baja California Peninsula, mangrove estuaries, taxonomy

INTRODUCTION

Mangrove estuaries in the Baja California Peninsula [PBC] are hypersaline coastal lagoons locally named “esteros” (Brusca et al., 2005), which play an important role as primary habitat or nursery grounds (for feeding and spawn) for a great variety of young and adult fish species that regularly enter to these biotopes, alternating seasonal changes on its density and biomass (González-Acosta et al. 2001, 2005), some of which are commercially important or represent critical links in the food chains (Oberrecht, 2014).

The most extensive mangrove vegetation areas in BCP occur mainly throughout both coasts of Baja California Sur (BCS), comprising the Laguna San Ignacio system, Bahía Magdalena and adjacent waters, the shoreline along the Pacific coast, Bahía de La Paz and the Gulf side (Whitmore et al., 2005); being Bahía de La Paz, the most studied area (e.g., Maeda-Martínez et al., 1982; Leija-Tristán et al., 1992; González-Acosta et al. 1999, 2004, 2005; López-Rasgado et al., 2012; Moreno-Sánchez et al., 2012).

At least 160 fish species associated to mangroves in BCS has been previously reported for these ecosystems (c.f. Brusca et al., 2005; Whitmore et al., 2005); however, considering the fact that the specific richness in these areas could increase substantially as the fish fauna of the mangrove ecosystems of BCS is more profusely studied and new species are reported or formally described (e.g., Whitmore et al., 2005; Hastings et al., 2010). For this reason, the main objectives of this study were the elaboration of an updated systematic checklist of the fish species occurring in mangroves estuaries of BCS and provide worthy information about its zoogeography and taxonomical status.

MATERIAL AND METHODS

The systematic checklist of fishes inhabiting the mangrove ecosystems of Baja California Sur was performed on the critical review of related literature as well as the verification of records with voucher specimens from biological museums: Bahía Concepción (Osburn & Nichols, 1916; Follett, 1960; Rodríguez-Romero et al., 1994; Ruiz-Campos et al., 2003) and Bahía de La Paz (Maeda-Martínez, 1981; Maeda-Martínez et al., 1982, Leija-Tristán et al. 1992; González-Acosta et al., 1999, 2001, 2005; López-Rasgado et al., 2012; Moreno-Sánchez et al., 2012); likewise, Bahía Magdalena, Laguna Guerrero Negro, Laguna Ojo de Liebre and Laguna of San Ignacio (Castro-Aguirre et al., 1999; Whitmore et al., 2005; Chávez, 2006; Froese and Pauly, 2014). Our systematic list follows Page et al. (2013) for

nomenclature and classification, whereas the zoogeographical affinities of the species were based on Briggs (1974) and Castro-Aguirre et al. (2005, 2006).

A comparative analysis of the species richness recorded in the mangrove estuaries from the both coast of BCS was performed. The taxonomy and distributional status of the fish species included in the present checklist were based on different sources of information (e.g., Castro-Aguirre and Espinosa Pérez, 1996; Castro-Aguirre et al., 1999; Espinosa Pérez et al., 2004; Page et al., 2013; Eschmeyer, 2014; Froese and Pauly, 2014).

RESULTS

The systematic checklist of fishes associated to mangrove estuarine ecosystems in Baja California Sur (BCS) is composed by two classes, 23 orders, 72 families, 147 genera and 228 species (Appendix 1). The class Actinopterygii was the most representative group with 91.2% (208 spp.) of the total fish diversity reported for the study region (including 132 genera, 61 families and 18 orders), contrasting with the class Chondrichthyes that grouped 20 species, 15 genera, 11 families and five orders. The most diverse order of cartilaginous fish is Myliobatiformes, which was represented by five families, six genera and eight species, most of them included in the family Urotrygonidae (two genera and four species); whereas the order Perciformes was the teleostean group with the highest fish diversity: 26 families, 76 genera and 125 species (Figure 1).

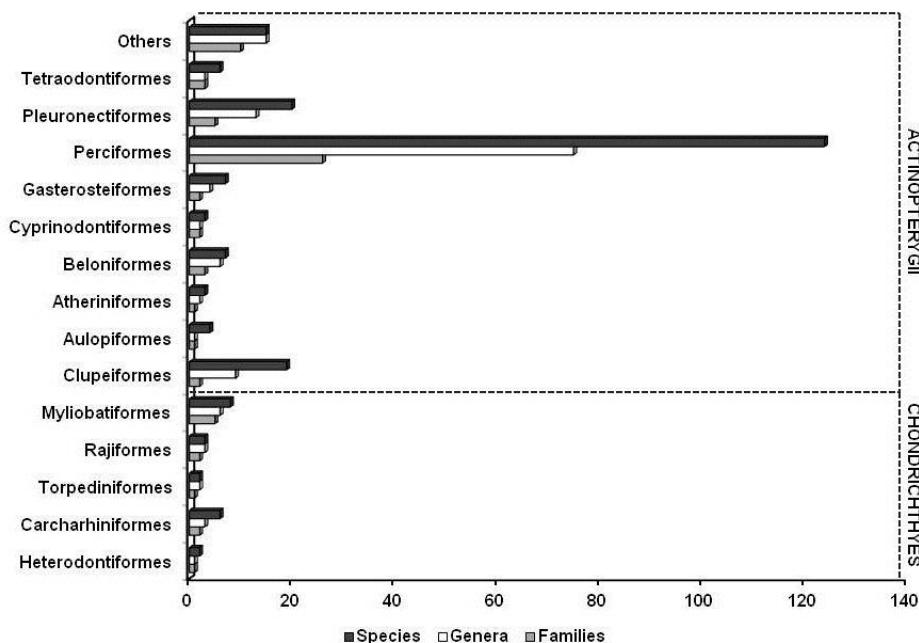


Figure 1. Distribution of the number of families, genera and species by order of the fishes recorded in mangrove estuaries of Baja California Sur.

Eight families of the order Perciformes contain 38.1% of the total species reported here, being Haemulidae (nine genera and 22 species) the most diverse family (Figure 2), followed by Sciaenidae (eight genera and 17 species), Gobiidae (11 genera and 12 species) and Carangidae (eight genera and 15 species) families.

Comparatively, the fish diversity reported in this chapter for mangrove ecosystems of BCS is notably higher than previously reported in fish surveys for these biotopes in both coasts of the Baja California peninsula (Table 1). Biogeographically, the fish fauna associated to mangrove estuaries of BCS showed a distribution confined to the Cortez province (83.7%) from the Californian region [CR] followed by the Mexican (73%) and Panamic (70.3%) provinces from the Tropical Eastern Pacific region [TEPR]; however, more than 61% of the fish fauna exhibited an affinity with the San Diegan province [CR], whereas the Oregonian province [CR] and the Galapagos province [TEPR] had affinity values of 1.3% each.

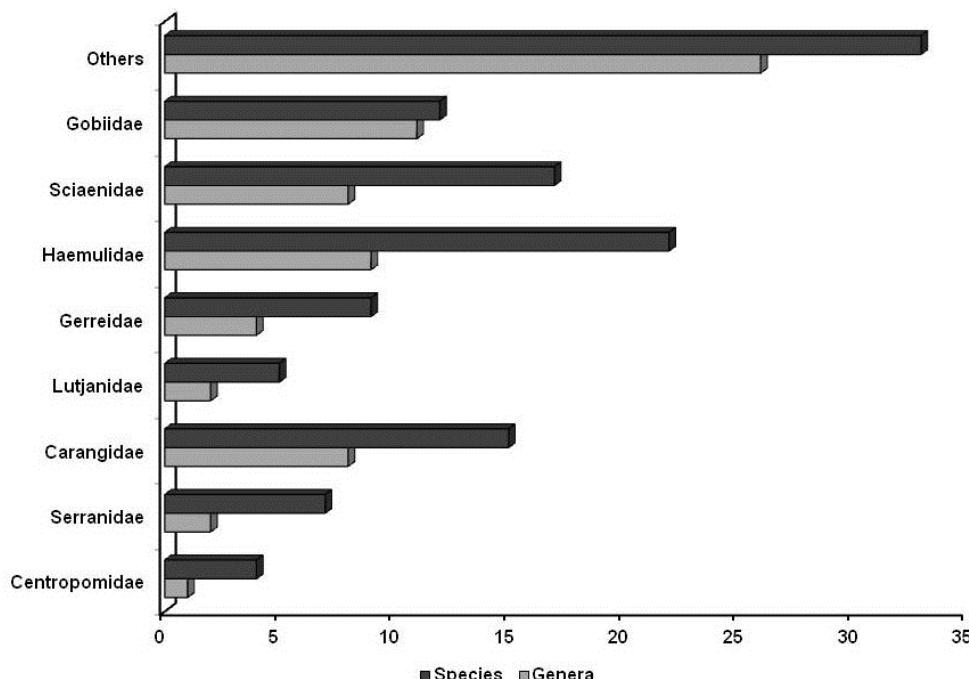


Figure 2. Number of genera and species recorded for the most representative families of the order Perciformes in mangrove estuaries of Baja California Sur.

Based on the distributional ranges of the species, the most of them are representative from the tropical eastern Pacific region, and in few cases with circumtropical (6.1%), amphiamerican (0.4%) and amphipacific (0.4%) distributions (Table 2); also the exotic component [Ex] is portrayed by the families: Poeciliidae (*Poecilia butleri* Jordan, 1889 and *P. reticulata*, Peters, 1860), Sparidae (*Sparus aurata* (Linnaeus, 1758), and Cichlidae (*Tilapia cf. zillii* (Gervais, 1848); whereas, three species of the family Gobiidae: *Aruma histrio* (Jordan, 1884), *Barbulifer pantherinus* (Pellegrin, 1901) and *Gobiosoma chiquita* (Jenkins and Evermann, 1889) are endemic [E] to the Gulf of California.

Table 1. Comparative surveys preformed in mangrove estuaries from Baja California Sur. Taxonomic categories: O (order), F (family), G (genera) and S (species)

Study	CHONDRICHTHYES				ACTINOPTERYGII				Total	
	O	F	G	S	O	F	G	S	Spp.	
Maeda-Martínez (1981)	1	2	2	2	9	26	44	51	53	
Maeda-Martínez et al. (1982)	-	-	-	-	-	-	-	-	41	
Leija-Tristán et al. (1992)	-	-	-	-	7	19	25	25	25	
Castro-Aguirre & Balart (1997)	4	9	11	13	15	43	87	122	135	
González-Acosta et al. (1999)	-	-	-	-	11	22	39	55	55	
González-Acosta et al. (2005)	-	-	-	-	10	16	26	34	34	
Whitmore et al. (2005)	2	5	7	8	12	48	99	152	160	
López-Rasgado et al. (2012)	1	1	1	1	15	29	57	81	82	
Moreno-Sánchez et al. (2012)	-	-	-	-	6	13	19	24	24	
This chapter	5	11	15	20	18	61	132	208	228	

Table 2. Zoogeography, distribution and verified fish records in mangrove estuaries of Baja California Sur. Zoogeographic provinces and regions *sensu* Briggs (1974): Californian region [PO (Oregonian Province), SP (San Diegan Province) and CP (Cortez Province)] and Tropical Eastern Pacific region [PM (Mexican Province), PP (Panamic Province) and PG (Galapagos Province)]; Geographic distribution affinity *sensu* (Castro-Aguirre et al., 2005, 2006): CT (circumtropical), AA (amphiamerican), A (amphipacific), EX (exotic) and E (endemic), localities recorded: CB (Bahía Concepción), GN (Laguna Guerrero Negro), LPB (Bahía de La Paz), MB (Bahía Magdalena), OL (Laguna Ojo de Liebre), (Laguna San Ignacio) and TB (Bahía Tortugas). Percentages on the total number of species are presented in parenthesis

Class	Zoogeographic affinity						Distribution						Verified records					
	PO	SP	CP	PM	PP	PG	CT	AA	AP	EX	E	CB	GN	LPB	MB	OL	SI	TB
CHONDRICHTHYES	2 (0.8)	15 (6.6)	18 (7.9)	15 (6.6)	14 (6.1)	1 (0.4)	-	-	-	-	-	7 (3)	1 (0.4)	5 (2.2)	15 (6.6)	- (4.8)	11 (0.4)	1
ACTINOPTERYGII	1 (0.4)	124 (54.4)	172 (75.4)	152 (66.6)	147 (64.5)	2 (0.8)	13 (5.7)	1 (0.4)	1 (0.4)	4 (1.7)	3 (1.3)	47 (20.6)	10 (2.3)	138 (60.5)	112 (49.1)	9 (3.9)	50 (21.9)	2 (0.8)
Totals	3 (1.3)	139 (60.9)	190 (83.3)	167 (73.2)	161 (70.6)	3 (1.3)	13 (5.7)	1 (0.4)	1 (0.4)	4 (1.7)	3 (1.3)	54 (23.7)	11 (4.8)	143 (62.7)	127 (55.7)	9 (3.9)	61 (26.7)	3 (1.3)

Fish records of the Bahía de La Paz (62.7%) and Bahía Magdalena (55.7%) mangrove ecosystems contribute significantly to the total species richness of BCS (Table 2). The cartilaginous fishes were more common in Bahía Magdalena (6.6%) and Laguna de San Ignacio (4.8%), whereas the actinopterygian fishes were more diverse in Bahía de La Paz (60.5%) than in Bahía Magdalena (49.1%).

DISCUSSION

Mangrove estuaries and sheltered coastal marine habitats are known around the world because its importance for supporting large stocks of fish species (Haedrich and Hall, 1976; McDowall, 1976), some of which are commercially important or represent critical links in the food chains (González-Acosta et al., 2001, 2005). Estuaries represent an important nursery

habitat for many species because enhance growth and survival of juvenile fish due to high food availability, low predation risk and protection against adverse weather conditions (Aboorike et al., 2000). Fish diversity occurring in mangrove estuaries of Baja California Sur (BCS) comprises 43.7% more in the number of taxa (70 species) than that previously reported by Whitmore et al. (2005). Likewise, our checklist adds 25% of the fish fauna previously reported for the Gulf of California (Hastings et al. 2010) and 8.2% of the total number of known species for México (Espinosa-Pérez, 2014).

The high diversity of families such as Haemulidae, Gerreidae, Carangidae, Sciaenidae, Gobiidae and Eleotridae, is a pattern commonly reported for the estuarine mangroves of the BCS (e.g., Castro-Aguirre and Balart, 1997; González-Acosta et al., 1999, 2005; Whitmore et al., 2005; López-Rasgado et al., 2012). This pattern is resulting from the availability of mangrove habitats with soft substrates coupled with the oceanographic conditions generated by the mixture of the tropical and temperate waters of the Pacific Ocean, all of them promoting the dominance of species from tropical derivation over those from temperate origin (e.g., Brusca et al., 2005; Whitmore et al., 2005; Hastings et al., 2010). In addition, this condition could explain the high zoogeographic affinity of the fish fauna of the study area with the Cortez (Californian region) and Mexican and Panamic provinces from the Tropical Eastern Pacific region (TEP). Likewise, more than 60% of the fish species reported in this chapter extend their northern distribution ranges into the San Diegan province (Californian region), denoting the intersection of this coastal region with those provinces of the TEP region. This intersection promotes a transitional zone between temperate and tropical faunas (Briggs, 1974; Castro-Aguirre et al., 1993; Hastings et al., 2010).

The shared presence in the mangrove estuaries from BCS by circumtropical elements such as *Carcharhinus leucas*, *Chanos chanos*, *Mugil cepa*, *M. curema* and *Scarus ghobban*, as well as amphipacific forms like *Fistularia commersonii* or the amphiamerican (*M. hospes*) species are very notable evidences of an ancient connection of the northeastern Pacific and Gulf of California with other biogeographic regions during several glacial episodes or geological-oceanographic events of the peninsular evolution (Castro-Aguirre et al., 1993, 1995; Castro-Aguirre and Balart, 1997). It is important to note the records of three exotic freshwater fish species: Pacific molly [*Poecilia butleri* Jordan] reported by first time by Palacios-Salgado et al. (2011) into the mangrove-root system of Estero Enfermería, Bahía de La Paz, BCS; Guppy [*P. reticulata* Peters] which has been reported as an invasive exotic fish of ample distribution in oases of the Baja California peninsula together with the third exotic species, the Redbelly tilapia [*Tilapia cf. zillii* (Gervais)] (Ruiz-Campos et al., 2003); and the Gilthead seabream [*Sparus aurata* Linnaeus] which was introduced after a failed project to establish its commercial culture into Bahía de La Paz, BCS, then was first recorded by Balart et al. (2008) in the vicinities of the mangrove estuary of Ensenada de La Paz, where every year at least one specimen has been caught by the local fishermen.

Three gobiids were found as endemics of the Gulf of California (Thomson et al., 2000): Slow Goby [*Aruma histrio* (Jordan, 1884)], Panther Goby [*Barbulifer pantherinus* (Pellegrin, 1901) and Sonora Goby [*Gobiosoma chiquita* (Jenkins and Evermann)], which represent 3.4% of the total endemic species reported for the Gulf of California (Brusca et al., 2005; Hastings et al., 2010).

Differences observed between the fish diversity reported here and previous fish surveys could be explained by the extension of the mangrove systems and a higher sampling effort, its geographic location and accessibility, and vicinity to urban areas. Other possible reason is the

fact that some published surveys do not indicate what fishes are or not associated to mangrove ecosystems (e.g., Whitmore et al., 2005) or because the information from fish collections are disperse in the web making inaccessible for a fast review.

The mangrove estuary ecosystems of BCS play an important role as rearing area for many marine fish species that use these biotopes as primary habitat or nursery (ground or feeding and spawning) during some phase of their biological cycle, contributing with the growth, recruitment and conservation of species with commercial value or as critical link in the food chains.

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APPENDIX 1

Systematic checklist of fish from mangrove estuaries of Baja California Sur; the taxonomical arrangement and nomenclature follows Page et al. (2013); ZA is the zoogeographic affinity based on Briggs (1974): provinces Oregonian (PO), San Diegan (SP), Cortez (CP), Mexican (MP), Panamic (PP) and Galapagos (GP), and Castro-Aguirre et al. (2005): CT (circumtropical), AA (amphiamerican), A (amphipacific), Ex (exotic) and E (endemic). LR are the localities of record: CB (Bahía Concepción), GN (Laguna Guerrero Negro), LPB (Bahía de La Paz), MB (Bahía Magdalena), OL (Laguna Ojo de Liebre), (Laguna San Ignacio) and TB (Bahía Tortugas)

TAXON	LR	ZA
Clase CHONDRICHTHYES		
ORDEN Heterodontiformes		
Familia HETERODONTIDAE		
<i>Heterodontus francisci</i> (Girard, 1855)	CB,MB,SI	SP,CP,MP,PP
<i>Heterodontus mexicanus</i> Taylor & Castro-Aguirre, 1972	CB,MB	CP,MP,PP
ORDEN Carcharhiniformes		
Familia TRIAKIDAE		
<i>Mustelus californicus</i> Gill, 1864	SI	SP,CP
<i>Mustelus henlei</i> (Gill, 1863)	LPB,MB	OP,SP,CP,PM,PP
<i>Mustelus lunulatus</i> Jordan & Gilbert, 1882	CB,LPB,MB,SI	SP,CP,MP,PP
<i>Triakis semifasciata</i> Girard, 1855	CB,MB,SI	SP,CP

Appendix 1. (Continued)

TAXON	LR	ZA
Familia CARCHARHINIDAE		
<i>Carcharhinus leucas</i> (Müller & Henle, 1839)	CB,MB	CT
ORDEN Torpediniformes		
Familia NARCINIDAE		
<i>Diplobatis ommata</i> (Jordan & Gilbert, 1890)	MB	CP,MP,PP
<i>Narcine entemedor</i> Jordan & Starks, 1895	CB,MB	CP,MP,PP
ORDEN Rajiformes		
Familia RHINOBATIDAE		
<i>Rhinobatos productus</i> Ayres, 1854	MB,SI	SP,CP
<i>Zapteryx exasperata</i> (Jordan & Gilbert, 1880)	MB	SP,CP,MP,PP
Familia RAJIDAE		
<i>Raja equatorialis</i> Jordan & Bollman, 1890	SI	SP,CP,MP,PP
ORDEN Myliobatiformes		
Familia PLATYRHYNIDAE		
<i>Platyrhinoidis triseriata</i> (Jordan & Gilbert, 1880)	SI	SP
Familia UROTRYGONIDAE		
<i>Urobatis halleri</i> (Cooper, 1863)	LPB,MB,SI	SP,CP,MP,PP
<i>Urobatis maculatus</i> Garman, 1913	GN,LPB,SI	CP,MP
<i>Urotrygon aspidura</i> (Jordan & Gilbert, 1882)	MB	SP,CP,MP,PP
<i>Urotrygon chilensis</i> (Günther, 1872)	LPB,MB	SP,CP,MP,PP
Familia DASYATIDAE		
<i>Dasyatis dipterura</i> (Jordan & Gilbert, 1880)	CB	OP,SP,CP,MP,PP,PG
Familia GYMNURIDAE		
<i>Gymnura marmorata</i> (Cooper, 1864)	MB,SI	SP,CP,MP,PP
Familia MYLIOBATIDAE		
<i>Myliobatis californica</i> Gill, 1865	SI	SP,CP,MP,PP
Clase ACTINOPERYGII		
ORDEN Elopiformes		
Familia ELOPIDAE		
<i>Elops affinis</i> Regan, 1909	LPB	SP,CP,MP,PP
ORDEN Albuliformes		
Familia ALBULIDAE		
<i>Albula esuncula</i> (Garman, 1899)	LPB,MB	SP,CP,MP,PP
ORDEN Angulliformes		
Familia OPHICHTHIDAE		
<i>Myrichthys tigrinus</i> Girard, 1859	LPB	CP,MP,PP
<i>Myrophis vafer</i> Jordan & Gilbert, 1883	LPB	CP,MP,PP
<i>Ophichthus zophochir</i> Jordan & Gilbert, 1882	MB	CP,MP,PP,GP
Familia CONRIDAE		
<i>Heteroconger digueti</i> (Pellegrin, 1923)	LPB	CP,MP
ORDEN Clupeiformes		
Familia ENGRAULIDAE		
<i>Anchoa argentivittata</i> (Regan, 1904)	LPB	CP,MP,PP

TAXON	LR	ZA
<i>Anchoa exigua</i> (Jordan & Gilbert, 1882)	MB	SP,CP,MP,PP
<i>Anchoa ischana</i> (Jordan & Gilbert, 1882)	LPB,MB	CP,MP,PP
<i>Anchoa lucida</i> (Jordan & Gilbert, 1882)	LPB,MB	SP,CP,MP
<i>Anchoa mundeola</i> (Gilbert & Pierson, 1898)	LPB	CP,MP
<i>Anchoa mundeofooides</i> (Breder, 1928)	LPB	CP,MP,PP
<i>Anchoa nasus</i> (Kner & Steindachner, 1867)	LPB,MB	SP,CP,MP,PP
<i>Anchoa scofieldi</i> (Jordan & Culver, 1895)	MB	SP,CP,MP,PP
<i>Anchoa walkeri</i> Baldwin & Chang, 1970	LPB	CP,MP,PP
<i>Anchovia macrolepidota</i> (Kner, 1863)	MB	SP,CP,MP,PP
<i>Cetengraulis mysticetus</i> (Günther, 1867)	LPB,MB	SP,CP,MP,PP
<i>Engraulis mordax</i> Girard, 1854	LPB	SP,CP,MP,PP
Familia CLUPEIDAE		
<i>Etrumeus teres</i> (DeKay, 1842)	LPB,MB	SP,CP,MP,PP
<i>Harengula thrissina</i> (Jordan & Gilbert, 1882)	CB,LPB	SP,CP,MP,PP
<i>Lile nigrofasciata</i> Castro-Aguirre, Ruiz-Campos & Balart, 2000	LPB	CP,MP,PP
<i>Lile stolifera</i> Jordan & Gilbert, 1884	CB,LPB,MB	CP,MP,PP
<i>Opisthonema libertate</i> (Günther, 1867)	LPB	SP,CP,MP,PP
<i>Opisthonema mediterraneum</i> Berry & Barret, 1963	LPB	SP,CP,MP,PP
<i>Sardinops sagax</i> (Jenyns, 1842)	SI	CT
ORDEN Gonorynchiformes		
XVIII Familia CHANIDAE		
<i>Chanos chanos</i> (Forsskål, 1775)	LPB,SI	CT
ORDEN Siluriformes		
Familia ARIIDAE		
<i>Ariopsis seemanni</i> (Günther, 1864)	LPB	CP,MP,PP
<i>Bagre panamensis</i> (Gill, 1863)	CB,LPB	CP,MP,PP
<i>Bagre pinnimaculatus</i> (Steindachner, 1877)	LPB	CP,MP,PP
<i>Notarius planiceps</i> (Steindachner, 1877)	LPB,MB	SP,MP,PP
ORDEN Aulopiformes		
Familia SYNODONTIDAE		
<i>Synodus evermanni</i> Jordan & Bollman, 1890	MB	SP,CP,MP,PP
<i>Synodus lacertinus</i> Günther, 1890	MB	CP,MP,PP
<i>Synodus lucioceps</i> (Ayres, 1855)	MB	SP,CP,MP
<i>Synodus scutuliceps</i> Jordan & Gilbert, 1882	CB,LPB,SI	CP,MP,PP
ORDEN Ophidiiformes		
Familia OPHIDIIDAE		
<i>Ophidion galeoides</i> (Gilbert, 1890)	OL	SP,CP,MP,PP
<i>Ophidion scrippsae</i> (Hubbs, 1916)	SI	SP
ORDEN Batrachoidiformes		
Familia BATRACHOIDIDAE		
<i>Porichthys margaritatus</i> (Richardson, 1844)	MB	SP,CP,MP,PP
<i>Porichthys myriaster</i> Hubbs & Schultz, 1939	MB,SI	SP,CP
ORDEN Mugiliformes		

Appendix 1. (Continued)

TAXON	LR	ZA
Familia MUGILIDAE		
<i>Mugil cephalus</i> Linnaeus, 1758	GN,LPB,MB, SI	CT
<i>Mugil curema</i> Valenciennes, 1836	CB,LPB,MB,SI	CT
<i>Mugil hospes</i> Jordan & Culver, 1895	LPB	AA
ORDEN Atheriniformes		
Familia ATHERINOPSIDAE		
<i>Atherinella eriarcha</i> Jordan & Gilbert, 1882	LPB,MB	SP,MP,PP
<i>Atherinops affinis</i> (Ayres, 1860)	GN,MB,OL,SI	SP,CP
<i>Atherinops californiensis</i> Girard, 1854	MB	SP,CP
ORDEN Beloniformes		
Familia EXOCOETIDAE		
<i>Fodiator acutus</i> (Valenciennes, 1847)	MB	SP,CP,MP,PP
Familia HEMIRAMPHIDAE		
<i>Hemiramphus saltator</i> Gilbert & Starks, 1904	MB	SP,CP,MP,PP
<i>Hyporamphus naos</i> Banford & Collette, 2001	LPB,MB	SP,CP,MP,PP
<i>Hyporamphus rosae</i> (Jordan & Gilbert, 1880)	MB,SI	SP,CP,MP,PP
Familia BELONIDAE		
<i>Abelennes hians</i> (Valenciennes, 1846)	MB	CT
<i>Strongylura exilis</i> (Girard, 1854)	LPB,MB	SP,CP,MP,PP
<i>Tylosurus pacificus</i> (Steindachner, 1876)	MB	CT
ORDEN Cyprinodontiformes		
Familia FUNDULIDAE		
<i>Fundulus parvipinnis</i> Girard, 1854	GN,MB	SP
Familia POECILIIDAE		
<i>Poecilia butleri</i> Jordan, 1889	LPB	Ex
<i>Poecilia reticulata</i> Peters, 1860	CB	Ex
ORDEN Gasterosteiformes		
Familia SYNGNATHIDAE		
<i>Hipocampus ingens</i> Girard, 1858	MB,OL,SI	SP,CP,MP,PP
<i>Pseudophallus starksii</i> (Jordan & Culver, 1895)	LPB	CP,MP,PP
<i>Syngnathus auliscus</i> (Swain, 1882)	GN,LPB,MB	SP,CP,MP,PP
<i>Syngnathus californiensis</i> Storer, 1845	MB	SP
<i>Syngnathus leptorhynchus</i> Girard, 1854	SI	SP
Familia FISTULARIDAE		
<i>Fistularia corneta</i> Gilbert & Starks, 1904	LPB	CP,MP,PP
<i>Fistularia commersonii</i> Rüppell, 1838	LPB,MB	AP
ORDEN Scorpaeniformes		
Familia TRIGLIDAE		
<i>Bellator xenisma</i> (Jordan & Bollman, 1890)	SI	SP,CP,MP,PP
<i>Prionotus stephanophrys</i> Lockington, 1881	LPB,MB,SI	SP,CP,MP,PP
ORDEN Perciformes		
Familia CENTROPOMIDAE		

TAXON	LR	ZA
<i>Centropomus armatus</i> Gill, 1863	LPB	CP,MP,PP
<i>Centropomus medius</i> (Günther, 1964)	MB	SP,CP,MP,PP
<i>Centropomus nigrescens</i> Günther, 1864	CB,LPB	CP,MP,PP
<i>Centropomus viridis</i> Lockington, 1877	LPB	SP,CP,MP,PP,PG
Familia POLYPRIONIDAE		
TAXON	LR	ZA
<i>Stereolepis gigas</i> Ayres, 1859	SI	CT
Familia EPINEPHELIDAE		
<i>Alphestes immaculatus</i> Breder, 1936	SI	CP,MP,PP
<i>Hyporthodus niphobles</i> (Gilbert & Starks, 1897)	LPB	SP,CP,MP,PP
<i>Mycteroperca xenarca</i> Jordan, 1888	MB	SP,CP,MP,PP
Familia SERRANIDAE		
<i>Diplectrum euryplectrum</i> Jordan & Bollman, 1890	CB,LPB	SP,CP,MP,PP
<i>Diplectrum labarum</i> Rosenblatt & Johnson, 1974	CB,MB	CP,MP,PP
<i>Diplectrum pacificum</i> Meek & Hildebrand, 1925	CB,LPB,MB,SI	SP,CP,MP,PP
<i>Diplectrum rostrum</i> Bortone, 1974	SI	CP,MP,PP
<i>Paralabrax clathratus</i> (Girard, 1854)	OL	SP,CP,MP
<i>Paralabrax maculatofasciatus</i> (Steindachner, 1868)	GN,LPB,MB,SI	SP,CP,MP,PP
<i>Paralabrax nebulifer</i> (Girard, 1854)	MB,OL,SI	SP,CP,MP
Familia CARANGIDAE		
<i>Caranx caninus</i> Günther, 1867	LPB	SP,CP,MP,PP
<i>Caranx melampygus</i> Cuvier, 1833	LPB	CT
<i>Caranx sexfasciatus</i> Quoy & Gaimard, 1825	LPB	CP,MP,PP
<i>Caranx vinctus</i> Jordan & Gilbert, 1882	CB,LPB,MB	SP,CP,MP,PP
<i>Chloroscombus orqueta</i> Jordan & Gilbert, 1883	LPB,MB	SP,CP,MP,PP
<i>Hemicaranx leucurus</i> (Günther, 1864)	LPB	SP,CP,MP,PP
<i>Gnathanodon speciosus</i> (Forsskål, 1775)	LPB	CP,MP,PP
<i>Oligoplites altus</i> (Günther, 1868)	CB,LPB,MB,SI	CP,MP,PP
<i>Oligoplites fulgens</i> Gilbert & Starks, 1904	LPB	SP,CP,MP,PP
<i>Oligoplites saurus</i> (Bloch & Schneider, 1801)	CB,LPB,MB	SP,CP,MP,PP
<i>Selene brevoortii</i> (Gill, 1863)	MB	SP,CP,MP,PP
<i>Selene peruviana</i> Steindachner, 1881	LPB,MB	SP,CP,MP,PP
<i>Trachinotus paitensis</i> Cuvier, 1831	CB,MB,SI	SP,CP,MP,PP
<i>Trachinotus rodophorus</i> Gill, 1863	CB,MB	SP,CP,MP,PP
<i>Trachurus symmetricus</i> (Ayres, 1855)	LPB	SP,CP,MP,PP
Familia LUTJANIDAE		
<i>Hoplopagrus guentherii</i> Gill, 1862	CB, LPB, MB	SP,CP,MP,PP
<i>Lutjanus aratus</i> (Günther, 1864)	CB,LPB	SP,CP,MP,PP
<i>Lutjanus argentiventralis</i> (Peters, 1869)	CB,LPB	SP,CP,MP,PP
<i>Lutjanus colorado</i> Jordan & Gilbert, 1882	CB,LPB	SP,CP,MP,PP
<i>Lutjanus novemfasciatus</i> Gill, 1862	CB,LPB	SP,CP,MP,PP
Familia GERREIDAE		
<i>Diapterus aureolus</i> (Jordan & Gilbert, 1882)	CB,LPB,MB	SP,CP,MP,PP
<i>Diapterus brevirostris</i> (Sauvage, 1879)	CB,LPB,MB	SP,CP,MP,PP

Appendix 1. (Continued)

TAXON	LR	ZA
<i>Eucinostomus currani</i> Zahuranec, 1980	CB,LPB,MB	SP,CP,MP,PP
<i>Eucinostomus dowii</i> (Gill, 1863)	CB,LPB,MB	SP,CP,MP,PP
<i>Eucinostomus entomelas</i> Zahuranec, 1980	CB,LPB,MB	SP,CP,MP,PP
<i>Eucinostomus gracilis</i> (Gill, 1862)	CB,LPB,MB	SP,CP,MP,PP
<i>Eugerres axillaris</i> (Günther, 1864)	LPB	CP,MP,MP
<i>Eugerres lineatus</i> (Humboldt, 1821)	CB,LPB,MB	CP,MP,PP
<i>Gerres cinereus</i> (Walbaum, 1792)	CB,LPB,MB	SP,CP,MP,PP
Familia HAEMULIDAE		
<i>Anisotremus davidsonii</i> (Steindachner, 1876)	SI	SP,CP
<i>Conodon serrifer</i> Jordan& Gilbert, 1882	LPB	CP,MP,PP
<i>Genyatremus dovii</i> (Günther, 1864)	MB	SP,CP,MP,PP
<i>Haemulon californiensis</i> (Steindachner, 1876)	SI	SP,CP
<i>Hemulon flaviguttatum</i> Gill, 1862	LPB	SP,CP,MP,PP
<i>Haemulon maculicauda</i> (Gill, 1862)	LPB	CP,MP,PP
<i>Haemulon scudderii</i> Gill, 1862	LPB	CP,MP,PP
<i>Haemulon sexfasciatum</i> Gill, 1862	LPB,MB	SP,CP,MP,PP
<i>Haemulon steindachneri</i> (Jordan & Gilbert, 1882)	LPB,MB	CP,MP,PP
<i>Haemulopsis axillaris</i> (Steindachner, 1869)	LPB	CP,MP,PP
<i>Haemulopsis elongatus</i> (Steindachner, 1879)	LPB,MB	SP,CP,MP,PP
<i>Haemulopsis leuciscus</i> (Günther, 1864)	CB,LPB,MB	CP,MP,PP
<i>Haemulopsis nitidus</i> (Steindachner, 1869)	LPB	CP,MP,PP
<i>Microlepidotus inornatus</i> Gill, 1862	LPB	CP,MP,PP
<i>Orthopristis chalceus</i> (Günther, 1864)	LPB,MB,SI	CP,MP,PP
<i>Orthopristis reddingi</i> Jordan & Richardson, 1895	LPB,MB,SI	CP,MP,PP
<i>Pomadasys bayanus</i> Jordan & Evermann, 1898	LPB,MB	CP,MP,PP
<i>Pomadasys branickii</i> (Steindachner, 1879)	LPB	CP,MP,PP
<i>Pomadasys macracanthus</i> (Günther, 1864)	LPB,MB	CP,MP,PP
<i>Pomadasys panamensis</i> (Steindachner, 1875)	LPB	CP,MP,PP
<i>Xenichthys xanti</i> Gill, 1863	CB,LPB	SP,CP,MP,PP
<i>Xenistius californiensis</i> (Steindachner, 1876)	CB,MB	SP,CP,MP,PP
Familia SPARIDAE		
<i>Calamus brachysomus</i> (Lockington, 1880)	CB,MB,SI	SP,CP,MP,PP
<i>Sparus aurata</i> Linnaeus, 1758	LPB	Ex
Familia POLYNEMIDAE		
<i>Polydactylus approximans</i> (Lay & Bennett, 1839)	LPB,MB,SI	SP,CP,MP,PP
Familia SCIAENIDAE		
<i>Atractoscion nobilis</i> (Ayres, 1860)	MB	OP,SP
<i>Bairdiella icistia</i> (Jordan & Gilbert, 1882)	LPB,MB,SI	CP,MP,PP
<i>Cynoscion parvipinnis</i> Ayres, 1861	LPB,MB,SI	SP,CP
<i>Cynoscion phoxocephalus</i> Jordan & Gilbert, 1882	LPB,MB	SP,CP,MP,PP
<i>Cynoscion stolzmani</i> (Steindachner, 1879)	MB	SP,CP,MP,PP
<i>Cynoscion xanthulus</i> Jordan & Gilber, 1882	LPB,MB	CP,MP,PP
<i>Elattarchus archidium</i> (Jordan& Gilbert, 1882)	LPB	CP,MP,PP

TAXON	LR	ZA
<i>Menticirrhus elongatus</i> (Günther, 1864)	MB	SP,CP,MP,PP
<i>Menticirrhus nasus</i> (Günther, 1868)	MB	SP,CP,MP,PP
<i>Menticirrhus undulatus</i> (Girard, 1854)	MB,SI	SP,CP,MP,PP
<i>Micropogonias altipinnis</i> (Günther; 1864)	LPB,MB	SP,CP,MP,PP
<i>Micropogonias ectenes</i> (Jordan & Gilbert, 1881)	LPB	CP,MP,PP
<i>Ophioscion strabo</i> Gilbert, 1897	LPB	CP,MP,PP
<i>Umbrina analis</i> Günther, 1868	MB	SP,CP,MP,PP
<i>Umbrina roncador</i> Jordan & Gilbert, 1882	CB,LPB,MB	SP,CP
<i>Umbrina wintersteeni</i> Walker & Radford, 1992	LPB	CP,MP,PP
<i>Umbrina xanti</i> Gill, 1862	MB	SP,CP,MP,PP
Familia MULLIDAE		
TAXON	LR	ZA
<i>Pseudopeneus grandisquamis</i>	LPB,MB	CP,MP,PP
Familia KYPHOSIDAE		
<i>Girella nigricans</i> (Ayres, 1860)	SI	SP,CP
<i>Hermosilla azurea</i> Jenkins & Evermann, 1889	SI	SP,CP
<i>Kyphosus analogus</i> (Gill, 1863)	CB,LPB	SP,CP,MP,PP
Familia CHAETODONTIDAE		
<i>Chaetodon humeralis</i> Günther, 1860	SI	CP,MP,PP
Familia POMACANTHIDAE		
<i>Pomacanthus zonipectus</i> (Gill, 1862)	CB,LPB,MB	SP,CP,MP,PP
Familia CICHLIDAE		
<i>Tilapia cf. zilli</i> (Gervais, 1848)	CB	Ex
Familia POMACENTRIDAE		
<i>Abudefduf troschelii</i> (Gill, 1862)	CB,LPB,MB	SP,CP,MP,PP
Familia LABRIDAE		
<i>Halichoeres semicinctus</i> (Ayres, 1859)	CB,MB,SI	SP,CP
<i>Oxyjulis californica</i> (Günther, 1861)	CB,MB,SI	SP,PP
<i>Scarus ghobban</i> Forskål, 1775	LPB	CT
Familia BLENIIDAE		
<i>Hypsoblennius gentilis</i> (Girard, 1854)	LPB,SI	SP,CP
Familia LABRISOMIDAE		
<i>Exerpes asper</i> (Jenkins & Evermann, 1889)	LPB,MB,SI	SP,CP
<i>Paraclinus mexicanus</i> (Gilbert, 1904)	LPB	SP,CP,MP,PP
<i>Paraclinus sini</i> Hubbs, 1952	LPB	SP,CP
Familia ELEOTRIDAE		
<i>Dormitator latifrons</i> (Richardson, 1844)	CB,LPB	CP,MP,PP
<i>Eleotris picta</i> Kner, 1863	LPB	SP,CP,MP,PP
<i>Erotelis armiger</i> (Jordan & Richardson, 1895)	CB,LPB	CP,MP,PP
<i>Gobiomorus maculatus</i> (Günther, 1859)	LPB,MB	SP,CP,MP,PP
Familia GOBIIDAE		
<i>Aruma histrio</i> (Jordan, 1884)	CB,LPB	E
<i>Barbulifer pantherinus</i> (Pellegrin, 1901)	CB,LPB	E
<i>Bathygobius ramosus</i> Ginsburg, 1947	LPB,MB	SP,CP,MP,PP
<i>Ctenogobius manglicola</i> (Jordan & Starks, 1895)	LPB	CP,MP,PP

Appendix 1. (Continued)

TAXON	LR	ZA
<i>Ctenogobius sagittula</i> (Günther, 1861)	CB,LPB,MB,SI	SP,CP,MP,PP
<i>Evorthodus minutus</i> Meek & Hildebrand, 1928	LPB	CP,MP,PP
<i>Gillichthys mirabilis</i> Cooper, 1864	GN,LPB	SP,CP
<i>Gobionellus microdon</i> (Gilbert, 1892)	LPB	SP,CP,MP,PP
<i>Gobiosoma chiquita</i> (Jenkins & Evermann, 1889)	CB,LPB	E
<i>Ilypnus gilberti</i> (Eigenmann & Eigenmann, 1889)	MB,SI	SP,CP
<i>Parrella maxillaris</i> Ginsburg, 1938	LPB	SP,CP,MP,PP
<i>Quietula y-cauda</i> (Jenkins & Evermann, 1889)	LPB,SI	SP,CP
Familia EPHIPPIDAE		
<i>Chaetodipterus zonatus</i> (Girard, 1858)	LPB,MB,SI	SP,CP,MP,PP
Familia SPHYRAENIDAE		
<i>Sphyraena argentea</i> Girard, 1854	SI	SP
<i>Sphyraena lucasana</i> Gill, 1863	MB	SP,CP,MP,PP
Familia SCOMBRIDAE		
<i>Auxis thazard</i> (Lacepède, 1800)	LPB	CT
<i>Sarda chiliensis</i> (Cuvier, 1832)	MB	SP,CP,MP,PP
<i>Scomber japonicus</i> Houttuyn, 1782	SI	CT
<i>Scomberomorus sierra</i> (Jordan & Starks, 1895)	MB	SP,CP,MP,PP
Familia STROMATEIDAE		
<i>Peprilus simillimus</i> (Ayres, 1860)	SI	SP,CP
ORDEN Pleuronectiformes		
Familia PARALICHTHYDAE		
<i>Citharichthys gibberti</i> Jenkins & Evermann, 1889	CB,LPB	SP,CP,MP,PP
<i>Cyclopsetta panamensis</i> (Steindachner, 1876)	LPB,MB	CP,MP,PP
<i>Cyclopsetta quernea</i> (Jordan & Bollman, 1890)	MB	SP,CP,MP,PP
<i>Etropus crossotus</i> Jordan & Gilbert, 1882	LPB,MB	SP,CP,MP,PP
<i>Hippoglossina stomata</i> Eigenmann & Eigenmann, 1890	SI	SP,CP,MP
<i>Hippoglossina tetrophthalmia</i> (Gilbert, 1890)	MB	SP,CP,MP,PP
<i>Paralichthys californicus</i> (Ayres, 1859)	GN,MB,SI	SP,CP
<i>Paralichthys woolmani</i> Jordan & Williams, 1897	LPB,MB	SP,CP,MP,PP
<i>Syacium ovale</i> (Günther, 1864)	MB	SP,CP,MP,PP
<i>Xystreurus liolepis</i> Jordan & Gilbert, 1880	MB,SI	SP,CP
Familia PLEURONECTIDAE		
<i>Hypsopsetta guttulata</i> (Girard, 1856)	CB,GN,MB, TB, OL, SI	SP,CP
<i>Pleuronichthys ritteri</i> Starks & Morris, 1907	SI	SP
<i>Pleuronichthys verticalis</i> Jordan & Gilbert, 1880	MB,SI	SP,CP
Familia BOTHIDAE		
<i>Bothus constellatus</i> (Jordan, 1889)		
Familia ACHIRIDAE		
<i>Achirus mazatlanus</i> (Steindachner, 1869)	CB,LPB,MB,SI	SP,CP,MP,PP
<i>Trinectes fonsecensis</i> (Günther, 1862)	LPB	CP,MP,PP

Familia CYNOGLOSSIDAE		
<i>Sympodus atricaudus</i> (Jordan & Gilbert, 1880)	SI	SP,CP
<i>Sympodus chabanaudi</i> Mahadeva & Munroe, 1990	MB	SP,CP,MP,PP
<i>Sympodus fasciolaris</i> Gilbert, 1892	MB	CP,MP,PP
ORDEN Tetraodontiformes		
Familia BALISTIDAE		
<i>Balistes polylepis</i> Steindacher, 1876	MB	SP,CP,MP,PP
Familia TETRAODONTIDAE		
<i>Sphoeroides annulatus</i> (Jenyns, 1842)	LPB,MB	SP,CP,MP,PP
<i>Sphoeroides lispus</i> Walker, 1996	LPB,GN, MB,OL	SP, CP
<i>Sphoeroides lobatus</i> (Steindachner, 1870)	LPB,GN,MB,SI	CP,MP,PP
Familia DIODONTIDAE		
<i>Diodon holocanthus</i> Linnaeus, 1758	LPB	CT
<i>Diodon hystrix</i> Linnaeus, 1758	LPB	CT

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Chapter 6

A SYSTEMATIC LIST OF FISHES FROM AN INSULAR MANGROVE ECOSYSTEM IN THE GULF OF CALIFORNIA

**Adrián F. González-Acosta^{1*}, Jeb Art Rabadan-Sotelo¹,
Gorgonio Ruiz-Campos², Luis F. Del Moral-Flores³
and José Manuel Borges-Souza¹**

¹Centro Interdisciplinario de Ciencias Marinas-IPN,

Col. Playa Palo de Santa Rita, La Paz, BCS, México

²Facultad de Ciencias, Universidad Autónoma de Baja California, Ensenada, Baja California, México

³Posgrado en Ciencias Biológicas, Instituto de Biología,
Universidad Nacional Autónoma de México, Universitaria, México, D.F., México

ABSTRACT

We provided a taxonomic list of fishes caught in the mangrove-lagoon ecosystem of the Isla San José, Baja California Sur, México, based on samplings performed during October 2004 to June 2009. Fifty-three species belonging to 40 genera and 29 families were recorded, of which the most diverse families were Haemulidae (six species), Lutjanidae and Gerreidae (four species each). Biogeographically, the fish fauna of San José estuary has a higher affinity with the Cortez province (77.4%), followed by the Mexican (75.5%) and Panamic (71.7%) provinces. Moreover, almost 70% (37 spp.) of the fish species reported here have commercial value. The King Angel fish, *Holacanthus passer*, is subject to special protection by the Mexican laws (NOM-059-SEMARNAT-2010). The use of the mangrove ecosystem of Isla San José by juveniles and adults of the species caught by artisanal fisheries and those subject to conservation programs, revealed the importance of this estuary as a critical habitat or nursery ground for several fish species in the Gulf of California.

* E-mail address: aacosta@ipn.mx.

Keywords: Gulf of California, fish diversity, insular mangrove ecosystem, Sea of Cortez, Taxonomy

INTRODUCTION

The Gulf of California is a large and a semi-enclosed sea that encompasses the presence of more than 900 islands and islets that constitute a region with high habitat diversity and characteristics that promote the forces of evolution to shape its flora and fauna (Brusca et al., 2005). Thus, the insular ecosystems in the Gulf of California support a high density, biomass and diversity of fish species with wide distribution and dispersal capabilities, as well as endemics and/or species restricted in their patterns of distribution and with limited dispersion (Thomson and Gilligan, 1983, 2002).

The Isla San José (ISJ) is a rugged island of volcanic origin uplift lying parallel to the Baja California Peninsula (BCP) and both separated by the San José Channel. The island littoral is characterized by the presence of a succession of high rocky bluffs (E side) with some beaches (W side) and lagoons bordered by mangroves (Muñoz-Lumbier, 1946). An extensive lagoon with an intricate maze of tidal channels, known as “Laguna o Estero de la Isla San José” is located at the south point of the island between Bahía Amortajada and Punta Ostiones. This lagoon is one of the most beautiful spots in the Gulf of California, which is bordered by the largest mangrove insular swamp in the gulf (Lewis and Ebeling, 1971; Bourillón-Moreno et al., 1988). This ecosystem provides a variety of habitats for several species of invertebrates, fishes and birds that nest and feed in the area.

Due to its proximity with La Paz city, the ISJ lagoon ecosystem is subject to an intensive touristic activity as well as artisanal and sport fisheries. However, in spite of being an area with great biodiversity few studies had been performed on this issue; some of these surveys are focused on birds (Carmona and Lozano, 2002), cartography (Hetzinger, 2003; Ruiz-Geraldo, 2005), paleozoology (Aguirre-Fernández, 2006), rodents (Espinosa-Gayoso and Álvarez-Castañeda, 2006) and soft-bottom invertebrates (Holguín-Quiñones et al., 2008). Several ichthyological studies deal on fish assemblages (e.g., Uscanga-Aguilar, 2002; Villegas-Sánchez, 2004; Villegas-Sánchez et al., 2009; Barjau-González et al., 2012) and list of fish species collected or reported in rocky and sand beach habitats around the island (e.g., González-Acosta, 2006; Del Moral-Flores, 2010; Palacios-Salgado et al., 2012; Del Moral-Flores et al., 2013). However, until now, no study on the fish fauna inhabiting the mangrove lagoon ecosystems of the island have been performed. Therefore, this chapter provides a systematic checklist of the fish that inhabit the mangrove-lagoon ecosystem (Estero) of Isla San José, with notes on its zoogeography.

MATERIAL AND METHODS

Study area. The Laguna (locally named Estero) of Isla San José, comprises a system of lagoon with an intricate maze of tidal channels which is situated at the south point of Isla San José ($24^{\circ} 53'12''$ N - $110^{\circ} 33'48''$ W), Baja California Sur between Bahía Amortajada and Punta Ostiones (Figure 1). The mangrove-lagoon system drains through a narrow tidal

channel into the main tidal channel and then into the bay, having two principal mouths influenced by the tidal cycle. An extensive coverage of red (*Rhizophora mangle*) and black (*Avicennia germinans*) mangrove swamp bordering the lagoon-channel system, providing a great variety of habitats for several species of invertebrates, fishes and birds that nest and feed in the area.

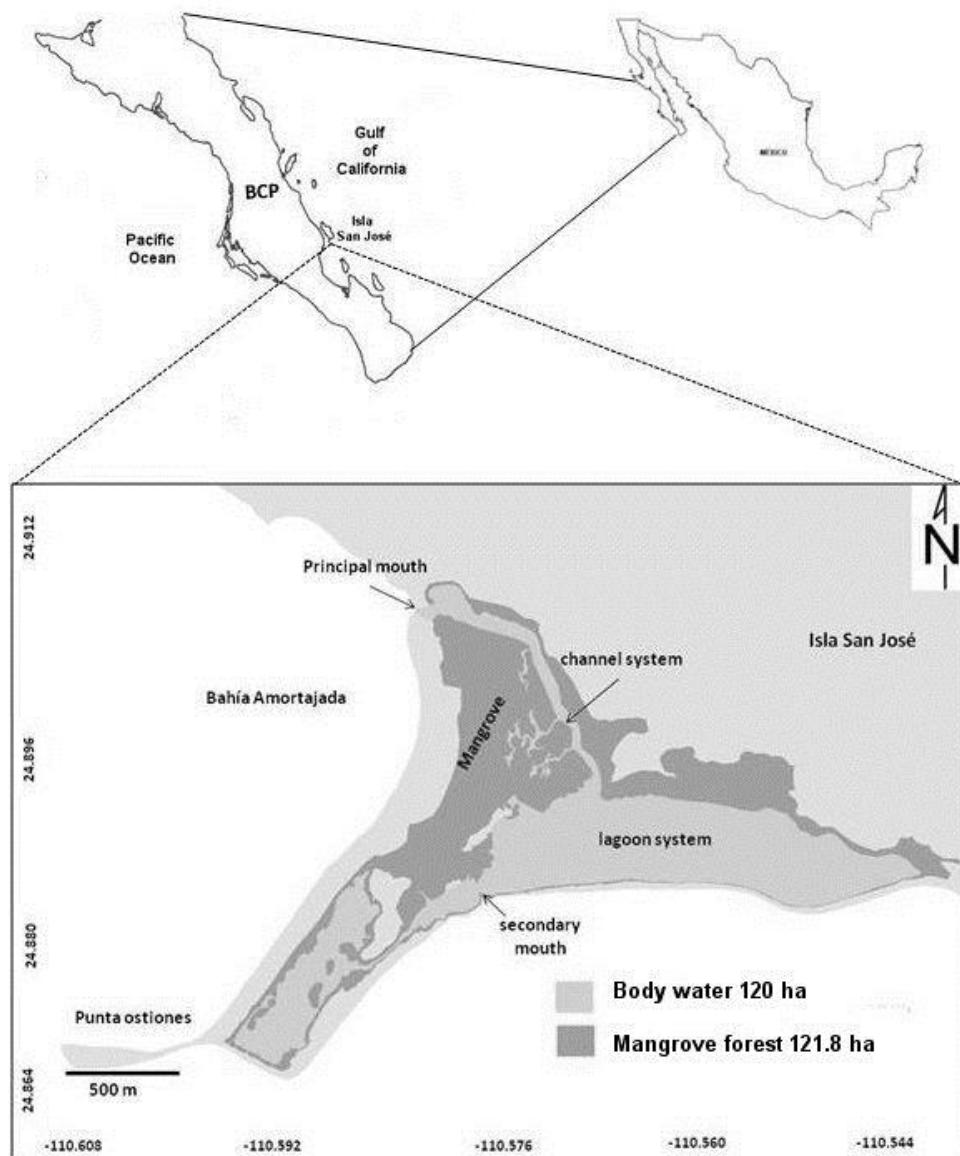


Figure 1. Study area on Isla San José, Baja California Sur, México. BCP = Baja California Peninsula.

The substrate is of mud and sand in the channel, whereas in the middle part of the lagoon is composed of sand-rocky patches. The main tidal channel is 1.5 km long and 4 to 10 m wide with 2 to 5 m in depth, whereas the main lagoon body is 2.68 km long with depth varying from 0.3 m (W side) to 8 m (E side). The tide is semidiurnal and is influenced by lunar cycles,

its range varying from 1.3 to 2.1 m, with a current velocity interval of 0.27 to 1.0 m^{-s}. The water temperature ranges from 24 to 27 °C in relation to the hour of the day and the tidal phase regarding the moon cycle; the salinity is similar to the adjacent sea with values ranging from 34.6 to 34.9 psu, in accordance to the ebb (reflux) or low (flux) tides.

Sampling fish methods. Fish were collected during the study period from October 2004 to June 2009, in several sites along the mangrove lagoon-channel system (Figure 1). Different fishing gears such as gill nets (50 m long, 2 m high and 3" and 6" mesh size), seine net (100 m long, 1.5 m high and 3 mm of mesh size), dip nets, traps, and speargun while scuba diving were employed.

All caught specimens were frozen and then transported to the fish laboratory at Centro Interdisciplinario de Ciencias Marinas (CICIMAR), where they were fixed with 10% formalin injected in the abdominal cavity and later preserved in 70% ethanol. The fish identification was performed using specialized taxonomic keys and fish catalogs (e.g., Burgess and Axelrod, 1984; Allen and Robertson, 1994; Fischer et al., 1995; Castro-Aguirre et al., 1999; Thomson et al., 2000). Some specimens of each species were housed as vouchers in the Fish Collection of the Universidad Autónoma de Baja California (CI-UABC) and the Colección Nacional de Peces of the Instituto de Biología-Universidad Nacional Autónoma de México (CNP-IBUNAM).

Taxonomy and zoogeography. The systematic checklist follows to Page et al. (2013) for the correct use of nomenclature and classification. The zoogeography was based on the Briggs's regionalization (Briggs, 1974) and the distributional patterns of the fish fauna following to Castro-Aguirre et al. (2005, 2006). Ecology and conservation status of the fish species were determined based on the critical review of related literature.

RESULTS

A total of 344 fish specimens corresponding to two classes, 14 orders, 29 families, 40 genera and 53 species, was caught in the mangrove-lagoon ecosystem of Isla San José (see Appendix I). The class Chondrichthyes depicted 4% (two species from the same genus, family and order) of the total species reported here, whereas 96% (51 spp., 39 genera, 29 families and 13 orders) corresponding to ray-finned fishes (class Actinopterygii).

Additionally, the order Perciformes was the most diverse group within the class Actinopterygii with 14 families, 22 genera and 31 species (Figure 2), followed by the orders Tetraodontiformes (2 families, 4 genera and 5 spp.) and Beloniformes (2 fam., 3 gen. and 3 spp.). Moreover, the richest families of the order Perciformes were Haemulidae (6 genera and 6 species), Lutjanidae and Gerreidae (2 genera and 4 species each), and Carangidae (2 genera and 3 species; Figure 3). Biogeographically, the fish fauna of the Isla San José lagoon showed a higher affinity with the Cortez (77.4 %) province [CP] from the California region [CA], followed by the Mexican [PM] (75.5%) and Panamic [PP] (71.7%) provinces that are belonging to the tropical eastern Pacific region [TEP]. Whereas, a lower affinity (47.2%) was corresponding to the San Diegan province [SP] from CA region (see Appendix 1). Likewise, 13.2% of the fish species included here was determined as Circumtropical [CT] based upon its patterns of distribution, others (7.5%) displayed a distribution in both coast of the Pacific Ocean [Amphipacific = AP], and 3.7% are Amphiamerican [AA].

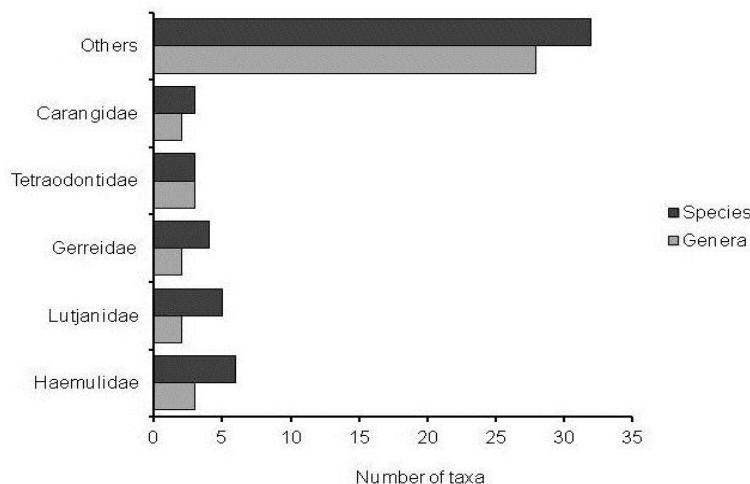


Figure 2. Taxonomic categories of the fish fauna caught in the mangrove-lagoon ecosystem of Isla San José, Baja California Sur, México.

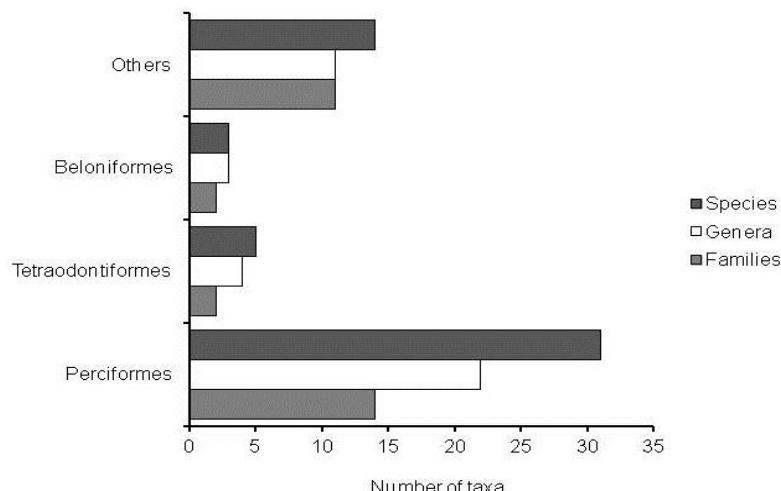


Figure 3. Taxonomic composition for the most representative families of fish species caught in the mangrove-lagoon ecosystem of Isla San José, Baja California Sur, México.

Thirty-seven (68.8%) fish species reported in this list are of commercial value because they support artisanal fisheries developed in La Paz bay. Of this total, only the King Angel fish, *Holacanthus passer* Valenciennes, 1846, is subject to special protection by the Mexican laws [NOM-059-SEMARNAT-2010] (DOF, 2010).

DISCUSSION

Comparatively the fish diversity recorded in the Isla San José mangrove-lagoon ecosystem depicts 5.8% of the total of fish species (911 spp.) known for the Gulf of California (Hastings et al., 2010). The great diversity of cartilaginous fishes (class

Chondrichthyes) reported by Hastings and Findley (2007) in the Upper Gulf Biosphere Reserve (northern Gulf), contrast with the higher fish diversity of ray-finned fishes (class Actinopterygii) here reported in addition to the paucity of sharks and ray fish species presents in the mangrove-lagoon ecosystem of ISJ. The limited presence of cartilaginous fish in the island is due to the reduced size of the tidal-channel mouth, coupled with the poverty of suitable habitats with soft-bottom substrates.

Moreover, the specific richness in our survey for the mangrove lagoon ecosystem of Isla San José (53 spp.) was lesser than those reported in other islands of the Gulf of California with a higher number of habitat sampled, such as Isla Cerralvo (Galván-Magaña et al. 1996) and Espíritu Santo (Rodríguez-Romero et al., 2005) with 174 and 120 fish species, respectively. Because of the recent interest for the study of the fish diversity of ISJ, the number of species has been increased from 62 (Uscanga-Aguilar, 2002) to 298 (Palacios-Salgado et al., 2012). Therefore, the systematic checklist of the fish fauna from the island could be updated on the basis of new studies in the area that increase the catch effort as well as the sources of information from fish collections (verified taxonomic and systematically) and the critical analysis of related literature. An example is the recent checklist of 243 species documented by Del Moral-Flores et al. (2013). Our list include exclusively those fishes collected in different types of habitats represented in the mangrove ecosystems of the Isla San José. Thus, main differences in the species number of the taxonomic lists here mentioned could be attributed to the use of several collecting methods (e.g., visual census, gill nets, seine nets, etc.), as well as differences in the sampling effort both in time and space and the sources of information (e.g., fish catalogs, checklist, review of specimens deposited in fish collections, etc.). Other factors that can increase our knowledge on the fish diversity in the mangrove-ecosystem of ISJ is the exploration of inaccessible areas using more specialized fishing gears (e.g., anesthetics, suction pumps, etc.) or the incidence on benthic groups (e.g., gobies, blennies), the discovery of new species or the taxonomic and systematic clarification of those previously known (Hastings et al., 2010).

The highest biogeographic affinities showed by the fish fauna from the mangrove-lagoon ecosystem of ISJ are with the Cortez (Californian region) and the Mexican and Panamic provinces (tropical eastern Pacific region); these higer affinities could be resulting of the strong linkage between the temperate and tropical regions of the eastern Pacific that determine a condition as a subtropical system of the Gulf of California (Brusca et al., 2005) and reflect the predominance of fish species from tropical derivation (Mexican and Panamic provinces) mixed with species from temperate areas from the Californian region (Hastings et al., 2010). Additionally, the low percentages of circumtropical, amhipacific and amphiamerican fish assemblages could be explained by the presence of a reduced number of species with ample distribution in the eastern Pacific and Gulf of California, like the milkfish *Chanos chanos* Forskål, 1775 which enters to the mangrove-lagoon ecosystem of ISJ to use the resources provided by the estuarine habitat. In this sense, Thomson and Gilligan (1983, 2002) mentioned that the “insular” fish fauna include species with wide distribution and dispersal capabilities.

The insular mangrove estuaries in the Gulf of California have been recognized as important nursery, refuge and feeding grounds for species coming of lagoons and estuarine ecosystems of the mainland coast (Thomson and Gilligan, 1983, 2002; Brusca et al., 2005). Therefore, do not be surprising the use of the insular mangrove-lagoon ecosystem of ISJ by juveniles and adults of the species caught by artisanal fisheries as well as those subject to

protection by the Mexican laws [NOM-059-SEMARNAT-2010] (DOF, 2010); like the King Angel fish *Holacanthus passer* Valenciennes, 1846, highlighting the importance of the San José estuary as critical habitat or nursery ground for several fish species in the Gulf.

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APPENDIX I

Systematic checklist of the fish species collected in the mangrove lagoon-channel system of Isla San José, Baja California Sur, México; the systematic arrangement and nomenclature follows Page et al. (2013). The zoogeographic affinity follows the scheme of regionalization and provinces established by Briggs (1974): Californian region: San Diegan (SP) and Cortez (CP), and the tropical eastern Pacific region: Mexican (MP) and Panamic (PP); likewise, patterns of geographic distribution follows to Castro-Aguirre et al. (2005, 2006): Circumtropical (CT), Amphiamerican (AA) and Amphipacific (AP). *

Species under conservation status of Mexican laws (NOM-059-SEMARNAT-2010)

TAXON	ZA
Clase CHONDRICHTHYES	
ORDEN Myliobatiformes	
Familia UROTRYGONIDAE	
<i>Urotrygon concentrica</i> Osburn & Nichols, 1916	CP,MP
<i>Urotrygon maculata</i> Garman, 1913	CP,MP
Clase ACTINOPTERYGII	
ORDEN Elopiformes	
Familia ELOPIDAE	
<i>Elops affinis</i> Regan, 1909	SP,CP,MP,PP
ORDEN Albuliformes	
Familia ALBULIDAE	
<i>Albula esculenta</i> (Garman, 1899)	SP,CP,MP,PP
ORDEN Anguilliformes	
Familia MURAENIDAE	
<i>Gymnothorax castaneus</i> (Jordan & Gilbert, 1883)	CP,MP,PP

Appendix 1. (Continued)

TAXON	ZA
ORDEN Clupeiformes	
Familia CLUPEIDAE	
<i>Opisthonema libertate</i> (Günther, 1867)	SP,CP,MP,PP
ORDEN Gonorynchiformes	
Familia CHANIDAE	
<i>Chanos chanos</i> (Forsskål, 1775)	AP
ORDEN Siluriformes	
Familia ARIIDAE	
<i>Occidentarius platypogon</i> (Günther, 1864)	SP,CP,MP,PP
ORDEN Mugiliformes	
Familia MUGILIDAE	
<i>Mugil cephalus</i> Linnaeus, 1758	CT
<i>Mugil curema</i> Valenciennes, 1836	AA
ORDEN Beloniformes	
Familia HEMIRAMPHIDAE	
<i>Hyporamphus naos</i> Banford & Collette, 2001	SP,CP,MP,PP
Familia BELONIDAE	
<i>Platybelone argalus</i> (Lesueur, 1821)	CT
<i>Tylosurus pacificus</i> (Steindachner, 1876)	CT
ORDEN Beryciformes	
Familia HOLOCENTRIDAE	
<i>Sargocentron suborbitalis</i> (Gill, 1863)	SP,CP,MP,PP
ORDEN Gasterosteiformes	
Familia SYNGNATHIDAE	
<i>Fistularia comersonii</i> Rüppell, 1838	AP
<i>Fistularia corneta</i> Gilbert & Starks, 1904	CP,MP,PP
ORDEN Scorpaeniformes	
Familia SCORPAENIDAE	
<i>Scorpaena mystes</i> Jordan & Starks, 1895	SP,CP,MP,PP
ORDEN Perciformes	
Familia NEMATISTIIDAE	
<i>Nematus pectoralis</i> Gill, 1862	SP,CP,MP,PP
Familia CARANGIDAE	
<i>Caranx caballus</i> Günther, 1869	SP,CP,MP,PP
<i>Caranx sexfasciatus</i> Quoy & Gaimard, 1825	CP,MP,PP
<i>Trachinotus rodophorus</i> Gill, 1863	SP,CP,MP,PP
Familia LUTJANIDAE	
<i>Hoplogagrus guentheri</i> Gill, 1862	SP,CP,MP,PP
<i>Lutjanus aratus</i> (Günther, 1864)	SP,CP,MP,PP
<i>Lutjanus argentiventralis</i> (Peters, 1869)	SP,CP,MP,PP
<i>Lutjanus colorado</i> Jordan & Gilbert, 1882	SP,CP,MP,PP
<i>Lutjanus novemfasciatus</i> Gill, 1862	SP,CP,MP,PP
Familia GERREIDAE	

TAXON		ZA
<i>Eucinostomus currani</i> Zahuranec, 1980		SP,CP,MP,PP
<i>Eucinostomus dowii</i> (Gill, 1863)		SP,CP,MP,PP
<i>Eucinostomus entomelas</i> Zahuranec, 1980		SP,CP,MP,PP
<i>Gerres cinereus</i> (Walbaum, 1792)		AA
Familia HAEMULIDAE		
<i>Anisotremus interruptus</i> (Gill, 1862)		CP,MP,PP
<i>Hemulon flaviguttatum</i> Gill, 1862		SP,CP,MP,PP
<i>Haemulon scudderii</i> Gill, 1862		CP,MP,PP
<i>Haemulon sexfasciatum</i> Gill, 1862		SP,CP,MP,PP
TAXON		ZA
<i>Haemulon steindachneri</i> (Jordan & Gilbert, 1882)		CP,MP,PP
<i>Microlepidotus inornatus</i> Gill, 1862		CP,MP,PP
Familia SPARIDAE		
<i>Calamus brachysomus</i> (Lockington, 1880)		SP,CP,MP,PP
Familia MULLIDAE		
<i>Mulloidichthys dentatus</i> (Gill, 1862)		SP,CP,MP,PP
Familia KYPHOSIDAE		
<i>Kyphosus analogus</i> (Gill, 1863)		SP,CP,MP,PP
Familia POMACANTHIDAE		
<i>Holacanthus passer</i> Valenciennes, 1846		CP,MP,PP
Familia POMACENTRIDAE		
<i>Abudefduf troschelii</i> (Gill, 1862)		SP,CP,MP,PP
<i>Stegastes rectifraenum</i> (Gill, 1862)		CP,MP,PP
Familia LABRIDAE		
<i>Scarus ghobban</i> Forskål, 1775		AP
<i>Thalassoma lucasanum</i> (Gill, 1862)		CP,MP,PP
Familia EPHIPPIIDAE		
<i>Chaetodipterus zonatus</i> (Girard, 1858)		SP,CP,MP,PP
Familia ACANTHURIDAE		
<i>Acanthurus xanthopterus</i> Valencianne, 1835		AP
<i>Prionurus punctatus</i> Gill, 1862		CP,MP,PP
Familia SCOMBRIDAE		
<i>Auxis rochei</i> (Risso, 1810)		CT
ORDEN Tetraodontiformes		
Familia TETRAODONTIDAE		
<i>Arothron meleagris</i> (Lacèpede, 1798)		CT
<i>Canthigaster punctatissima</i> (Günther, 1870)		CP,MP,PP
<i>Sphoeroides lobatus</i> (Steindachner, 1870)		CP,MP,PP
Familia DIODONTIDAE		
<i>Diodon holocanthus</i> Linnaeus, 1758		CT
<i>Diodon hystrix</i> Linnaeus, 1758		CT

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Chapter 7

**SEASONAL VARIATION
IN THE COMPOSITION AND ABUNDANCE
OF MOJARRA SPECIES (TELEOSTEI: GERREIDAE) IN
A MANGROVE ECOSYSTEM IN THE GULF OF
CALIFORNIA, MÉXICO**

***Leticia Jareny Ramos-Lozano¹, Adrián F. González-Acosta^{1,*},
Gustavo de la Cruz-Agüero¹ and Gorgonio Ruiz-Campos²***

¹Centro Interdisciplinario de Ciencias Marinas-IPN. Av. Instituto Politécnico Nacional s/n, Colonia Playa Palo de Santa Rita, La Paz, Baja California Sur, México

²Facultad de Ciencias, Universidad Autónoma de Baja California, Ensenada, Baja California, México

ABSTRACT

Seasonal variation in the composition and abundance of a guild of seven gerreid species: *Diapterus brevirostris* (Sauvage, 1879), *Eucinostomus currani* Zahuranec, 1980, *Eucinostomus entomelas* Zahuranec. 1980, *Eucinostomus gracilis* (Gill, 1862), *Eucinostomus dowii* (Gill, 1863), *Eugerres lineatus* (Humboldt, 1821), and *Gerres cinereus* (Walbaum, 1792) was examined in order to assess the effect of the hydrological dynamics of El Conchalito estuary, a mangrove swamp ecosystem located in the southern Gulf of California. Monthly samples were collected from June 1996 to August 1997 using a stationary flume net deployed across the tidal channel. Densities were recorded for each species, as well as the standard length and weight of each individual. Hydrological parameters (water temperature, salinity, tidal level and dissolved oxygen) were measured *in situ* during each sampling event. The most abundant species in terms of density was *D. brevirostris* (62.6 %), while *E. entomelas* was the most prevalent species in terms of biomass (34.6%). A significant correlation was found between temperature and species abundance (including *E. currani*). Seasonal environmental variability in the ecosystem prompts changes in the composition and abundance of gerreids; the high

* E-mail address: aacosta@ipn.mx.

incidence of juveniles during summer and winter highlights the importance of this ecosystem as a nursery for fish fauna.

Keywords: Ecology, gerreids, El Conchalito estuary, Baja California Sur, mangrove biotope

INTRODUCTION

Coastal ecosystems, like estuaries and coastal lagoons, are highly productive biotopes created by the influx of nutrients resulting from the degradation of mangrove litter (Haedrich and Hall, 1976; Cyrus and Blaber, 1983; Espinosa Pérez et al., 2011); where a variety of species take advantage of the seasonality of these unique ecosystems, using them for nursery, protection, and feeding.

Fish of the family Gerreidae, commonly known as mojarras, typically inhabit coastal lagoons and estuaries bordered by mangroves in the tropical and subtropical regions (Albarete and Desfosses, 1988; Bussing, 1995; Gilmore and Greenfield, 2002; McEachran and Fechhelm, 2005); with some species living partially or permanently in freshwater ecosystems (Castro-Aguirre et al., 1999; González-Acosta et al., 2007; González-Acosta and Rodiles-Hernández, 2013).

The use of estuarine habitats by gerreid fish has been observed in studies of recruitment patterns, seasonal abundance and frequency (Aguirre-León et al., 1982; Aguirre-León and Yáñez-Arancibia, 1986; Albaret and Desfosses, 1988; Araújo and De Alcantara-Santos, 1999), and feeding ecology (Cyrus and Blaber, 1983; Kerschner et al., 1985; Arenas-Granados and Acero, 1992; Chávez-Comparán and Hammann, 1989; Cunningham and Maciel, 1995; Rivas et al., 1999).

Gerreids are among the most common and abundant fish in El Conchalito estuary (González-Acosta et al., 1999, 2005), a mangrove swamp in Baja California Sur, México. As a result, they are threatened by anthropogenic activities, including garbage disposal, pollution, recreational activities, and the transit of vehicles (González-Acosta et al., 2005). The main goal of this chapter was to assess the importance of this biotope as a nursery area for a guild of seven gerreid fish and to determine how the hydrological dynamics affect their seasonal abundance and recruitment patterns.

MATERIAL AND METHODS

Study site: Fish were sampled monthly in El Conchalito estuary (Figure 1), a mangrove swamp ecosystem located in La Paz Lagoon in Baja California Sur, México ($24^{\circ}08'34''$ - $24^{\circ}07'40''$ N and $110^{\circ}21'04''$ - $110^{\circ}20'35''$ W) from June 1996 to August 1997.

Study system: A modified, stationary 15 m x 15 m “flume net” (cf. McIvor and Odum, 1986) with 6.3 mm mesh size equipped with a net bag (1.5 m x 1.5 m x 4 m long) was deployed across the mouth of the tidal channel ($24^{\circ}08'17''$ N and $110^{\circ}20'51''$ W) during full moons in order to take advantage of the resulting high tide. Individuals were collected at 2-h intervals during the net operation (6-h of the tidal reflux), then preserved in 10% formalin and

identified to species level using taxonomic keys. All specimens were counted and individually weighed (0.01 g) and measured in standard length (0.01 mm).

Estimation of abundance: Fish abundance was estimated monthly and averaged annually in order to calculate the density (individuals/m²) and biomass (g/m²) of each sample. The relative frequency of individuals for each species was also determined monthly. The most abundant species were grouped by size in order to assess the length-frequency distribution of gerreids in this mangrove ecosystem.

Measurement of physicochemical parameters: Environmental parameters such as water temperature (°C), water salinity (UPS), tidal range, current velocity (cm·s⁻¹), and dissolved oxygen concentration were also recorded during each sampling period. The current velocity was measured with an Ocean Instruments S4 current meter (programmed to take 120 readings per minute, every four minutes), while the dissolved oxygen concentration was recorded using Winkler's method (Strickland and Parsons, 1972). The Pearson's correlation coefficient (Roscoe, 1975) was used to analyze the correlation between the environmental parameters and fish abundance.

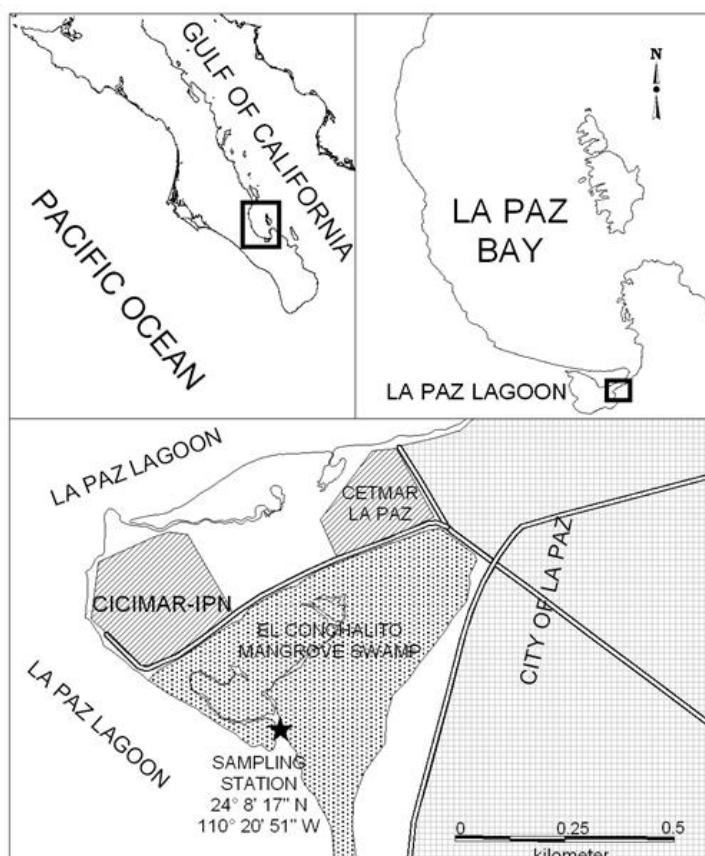


Figure 1. Collecting area of gerreid fishes in El Conchalito mangrove swamp, Baja California Sur, México.

RESULTS

Measurement of relationship environmental factors and fish abundance: The Spearman's rank non parametric correlation coefficient (ρ) was used to examine the degree of association between the environmental parameters and species abundance (Siegel and Castellan, 1988). Here, $\rho = 1$ reflects a positive association, $\rho = -1$ is a negative correlation, and $\rho = 0$ indicates there is no correlation (Yáñez, 2005).

Environmental parameters: The average water temperature varied seasonally from 18°C in January to 31.8°C in September, with an annual average of 25°C. The water salinity varied diurnally and seasonally from 37 UPS (January, September, and October) to 41 UPS (June), with an annual average of 38.2 UPS, exceeding the average sea salinity of 35 UPS. The dissolved oxygen (DO) concentration showed diurnal changes but without seasonal pattern. The DO concentration varied from 3.3 mg/l (February) to 6.3 mg/l (March), with an annual average of 5.2 mg/l. The tidal level exhibited a range from 1.0 m in February to 3.0 m in January, with an annual average of 2.4 m. The current velocity was markedly influenced by tidal range and the semidiurnal cycle, varying between 13 cm^{-s} in June and 32.9 cm^{-s} in September, with an annual average of 24.7 cm^{-s}.

Relative abundance: A total of 10 005 individuals (weighing a total of 6 659 g) representing four genera and seven species of the family Gerreidae were collected. The collected species, from the most to the least abundant were: *Diapterus brevirostris*, *Eucinostomus currani*, *Eucinostomus entomelas*, *Eucinostomus dowii*, *Eucinostomus gracilis*, *Gerres cinereus*, and *Eugerres lineatus* (Table 1). Two species contributed 81.5% of the total fish abundance by number: *D. brevirostris* (62.6 %) and *E. currani* (18.9 %); whereas *E. entomelas* (34.6 %) and *E. currani* (30.0 %), contributed 64.6% by weight to the total biomass of Gerreidae caught in this mangrove biotope (Table 1).

Table 1. Density, biomass and frequency of gerreid species caught in El Conchalito mangrove swamp. N = number of individuals, W = biomass, RF = relative frequency

Species	N	N/m ²	%	W	g/m ²	%	RF	%
<i>Diapterus brevirostris</i>	6 269	0.056	62.6	1 228	0.011	18.5	8	19.05
<i>Eucinostomus currani</i>	1 900	0.012	18.9	1 997	0.018	30	9	21.43
<i>Eucinostomus entomelas</i>	967	0.008	9.6	2 302	0.02	34.6	8	19.05
<i>Eucinostomus dowii</i>	293	0.002	2.9	642	0.005	9.65	6	14.29
<i>Eucinostomus gracilis</i>	546	0.004	5.4	439	0.004	6.6	5	11.9
<i>Eugerres lineatus</i>	4	3x10 ⁻⁵	0.03	24	3x10 ⁻⁴	0.42	2	4.76
<i>Gerres cinereus</i>	26	3x10 ⁻⁴	0.25	27	2x10 ⁻⁴	0.31	4	9.52
Total	10 005	0.0994	100	6 659	0.0605	100		100

Table 2. Monthly density (ind.m⁻²) and biomass (g.m⁻²) of the gerreid species (in brackets). Boldface indicates higher abundances

Species	J	F	M	A	M	J	J	A	S	O	N	D	Total	%
<i>D. brevirostris</i>	-	-	0.000045	-	0.000472	0.000627	0.000436	0.00072	0.00058	0.0534	0.00076	-	0.05699	62.6
			[0.000164]	-	[0.00143]	[0.00365]	[0.0015]	[0.00135]	[0.00064]	[0.00145]	[0.00094]	-	[0.01116]	[18.4]
<i>E. currani</i>	-	-	0.000081	-	0.00084	0.00027	0.00072	0.00095	0.00194	0.00714	0.00525	0.000063	0.0172	18.9
			[0.000018]	-	[0.00089]	[0.0002]	[0.00069]	[0.0026]	[0.00096]	[0.0056]	[0.007]	[0.00014]	[0-01815]	[29.9]
<i>E. entomelas</i>	-	-	0.000036	-	0.00042	0.00042	0.000027	0.00087	0.00031	0.00209	0.00462	-	0.00879	9.6
			[0.00013]	-	[0.00106]	[0.00173]	[0.00014]	[0.0017]	[0.00224]	[0.00495]	[0.00896]	-	[0.02093]	[34.6]
<i>E. dowii</i>	-	-	0.000009	-	0.00022	0.00016	0.00013	-	0.00056	0.0016	-	0.00266	2.9	
			[0.000026]	-	-	[0.0023]	[0.0003]	[0.00014]	-	[0.00099]	[0.00205]	-	[0.00584]	[9.6]
<i>E. gracilis</i>	-	-	-	-	0.000009	-	0.000045	0.000163	-	0.001254	0.0035	-	0.004963	5.4
			-	-	[0.000012]	-	[0.00006]	[0.00008]	-	[0.00098]	[0.00286]	-	[0.00399]	[6.6]
<i>E. lineatus</i>	-	0.000009	-	-	-	-	-	-	-	-	-	0.000027	0.000036	0.04
		[0.00003]	-	-	-	-	-	-	-	-	-	[0.00019]	[0.00019]	[0.3]
<i>G. cinereus</i>	-	-	-	-	0.000009	-	0.000009	-	-	0.000009	0.00021	-	0.000236	0.26
		-	-	-	[0.000045]	-	[0.000003]	-	-	[0.000024]	[0.00018]	-	[0.00025]	[0.4]
Total		-	0.00017	-	0.00173	0.00153	0.00134	0.00267	0.00282	0.06314	0.01222	0.000063	0.09095	
		-	[0.00338]	-	[0.00344]	[0.00791]	[0.00272]	[0.00591]	[0.00385]	[0.01402]	[0.022]	[0.00033]	[0.06052]	100

Eucinostomus currani was the most frequent species in the ecosystem (RF = 21.4%), followed by *D. brevirostris* and *E. entomelas* each with 19.05% of relative frequency. In contrast, *E. lineatus* was the gerreid species with the lower frequency in the estuary (RF = 4.76%).

Seasonal abundance: The density of the species varied seasonally. The most abundant species, *D. brevirostris*, increased in density in autumn (September–November); other gerreid species also showed a similar pattern during this season, but were less abundant (Table 2). Moreover, the most important species in terms of biomass, *E. entomelas*, was the prevalent species during the winter (October–December); the remaining species presented multimodal patterns throughout the study period (Table 2).

Relationship between environmental parameters and fish abundance: The Pearson's correlation coefficient between water salinity and current velocity indicated a significant negative correlation ($r = -0.747$; $p > 0.05$). Moreover, the Spearman's rank correlation indicated a significant positive correlation between temperature and the abundance of *D. brevirostris* ($\rho = 0.74$), *E. currani* ($\rho = 0.782$) and *E. entomelas* ($\rho = 0.645$) (Table 3).

Monthly length-frequency distributions: Monthly length-frequencies for *D. brevirostris* ranged from 16.5 to 88.7 mm SL (average 38.8 mm SL), indicating that small fish are recruiting to the estuary during summer and autumn. Interestingly, *D. brevirostris* was absent from the ecosystem in the winter (Figure 2).

For the Pacific flagfin mojarra, *E. currani*, monthly length-frequencies ranged from 9.2 to 88.8 mm SL (average 29.2 mm SL), with recruitment of juveniles to the estuary during the spring, summer, and autumn (Figure 3). Moreover, the monthly length-frequency distribution of *E. entomelas* varied between 15.6 and 106.3 mm SL (average 40.5 mm SL), with recruitment of juveniles occurring in summer and autumn (Figure 3).

Recruitment of juvenile individuals of *E. dowii* and *G. cinereus* was observed during autumn, whereas *E. gracilis* recruited during summer. Patterns of recruitment for *E. lineatus* could not be identified due to the low number of individuals caught during the study period.

Table 3. Pearson's correlation matrix for environmental factors and gerreid fish density. Boldface indicated the significant correlation ($P < 0.05$). °C (temperature), UPS (salinity), O₂ (dissolved oxygen), TL (tidal level) and CV (current velocity)

Species	°C	UPS	O ₂	TL	CV
<i>D. brevirostris</i>	0.74	0.272	0.082	-0.313	0.285
<i>E. currani</i>	0.782	0.113	0.113	-0.197	0.479
<i>E. entomelas</i>	0.645	0.312	0.21	-0.371	0.168
<i>E. dowii</i>	0.437	0.372	0.246	-0.388	-0.007
<i>E. gracilis</i>	0.468	0.141	0.351	-0.172	0.351
<i>E. lineatus</i>	-0.511	-0.287	-0.43	-0.011	0.086
<i>G. cinereus</i>	0.269	0.237	0.382	-0.336	0.084

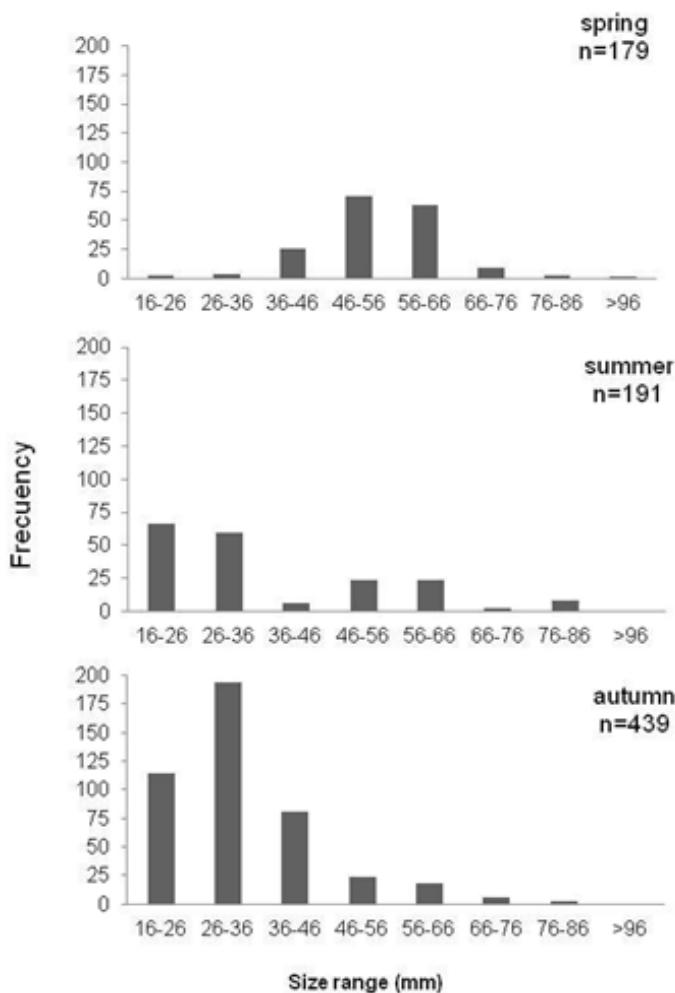


Figure 2. Seasonal length-frequency distribution of *Diapterus brevirostris* in El Conchalito mangrove swamp, Baja California Sur, México.

DISCUSSION

El Conchalito mangrove ecosystem in the southern Gulf of California is important because of the diversity of marine fish that use this biotope as a primary habitat for nursery, foraging, and recruitment, as well as for its conditions of low anthropic disturbance (González-Acosta et al., 1999, 2001, 2005).

The predominance of *D. brevirostris* in density, and *E. entomelas* in biomass, reflects the abundance pattern previously reported by González-Acosta et al. (2005) and Ramos-Lozano (2010). The other gerreid species exhibited seasonal changes in density and biomass, alternating with other commercially and ecologically important fish species that regularly enter the ecosystem to feeding and spawn (González-Acosta et al. 2001, 2005). Moreover, less abundant species, such as *E. lineatus* and *G. cinereus*, are more frequent in habitats with open or deep waters (Allen and Robertson, 1994; Bussing, 1995).

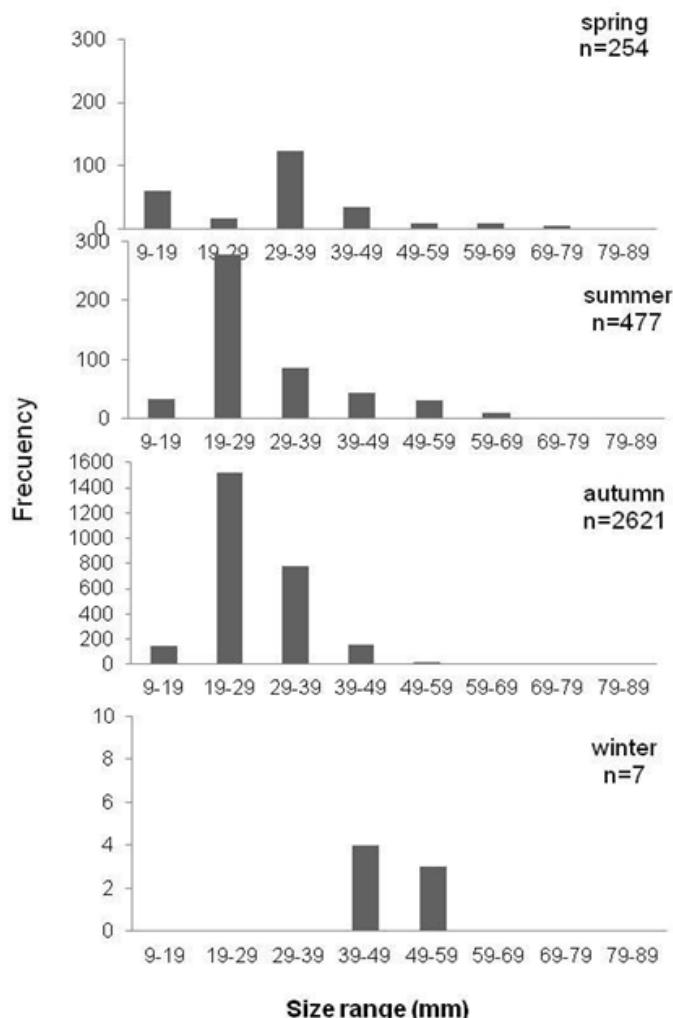


Figure 3. Seasonal length-frequency distribution of *E. currani* in El Conchalito mangrove swamp, Baja California Sur, México.

The seasonality of the composition and abundance patterns of gerreids in tropical estuarine biotopes has been linked to local factors (e.g., salinity, rainfall), unfavorable environmental conditions or migratory movements related to spawning and recruitment (Araújo and De Alcantara-Santos, 1999; Leal de Castro, 2001) as well as feeding and nursery purposes (Cyrus and Blaber, 1983; Albaret and Defossez, 1989; Chávez-Comparan and Hamman, 1989). The seasonality of the composition of the most abundant gerreid species in El Conchalito mangrove swamp, may be result of varying competition between species making use of the available resources in this estuarine environment at different times (cf. Tzeng and Wang, 1992).

The significant correlations found between temperature and abundance for *D. brevirostris*, *E. currani* and *E. entomelas* reflects their preferences for tropical or subtropical habitats (Bussing, 1995). Despite the fact that no other environmental parameters were significantly correlated with gerreid abundance in the study area, variability in environmental

factors (e.g., salinity, temperature and distance from the mouth) might influence seasonal changes in the composition of fish communities or the relative abundance of species with different life cycles (Loneragan, 1992).

El Conchalito mangrove swamp is used by a variety of fish species as a primary habitat for nursery and feeding activities. Of particular importance are those species, like the seven mojarra species considered here, that are the focus of the artisanal and recreational fisheries in La Paz Bay. Any disturbance to this important ecosystem would affect not only the fish species that inhabit it, but also the artisanal fisheries that depend of them. Likewise, our results are a first step towards creating a data base focused exclusively on local species as we excluded from our study species not distributed in the area like *Eucinostomus argenteus* (e.g., Vázquez-Hurtado et al., 2010), a species from the western Atlantic (Eschmeyer, 2013).

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Chapter 8

EFFECTS OF CLIMATE CHANGE ON MANGROVES: TWO CASE STUDIES

Martha Patricia Mora Flores^{1*} and Sara Cecilia Díaz Castro²

¹Centro Interdisciplinario de Investigación para el Desarrollo Integral Regional, Instituto Politécnico Nacional, Unidad Oaxaca (CIIDIR-IPN Oaxaca), Becaria COFAA-IPN, Santa Cruz Xoxocotlán, Oaxaca, Mexico

²Centro de Investigaciones Biológicas del Noroeste, S.C. Avenida Instituto Politécnico Nacional No. 195, La Paz, México

ABSTRACT

Mangroves are capable of absorbing or reorganizing stress effects to maintain their function, processes, and structure. Nevertheless, in spite of their ecosystem resilience and environmental adaptation, they have to face climate change risk. At world level, the forecast is an increase in CO₂, CH₄, and N₂O concentrations, as well as in temperature, a higher impact in runways, drought, ocean acidification, sea level rise, and hurricanes. However, vulnerability of mangrove systems is not only according to biophysical interactions with environmental risks but also to links among socio-economical structures, cultural norms, and social values.

Particularly, an accelerated speed of sea level rise will probably have an impact in vegetation loss toward the sea, an inland or a lateral shift that will depend on the contribution of hydrology, sediments, and anthropogenic modifications to the landscape. The vulnerability degree of the mangrove ecosystem will depend on the particular regional conditions. In this chapter we show case studies on two lagoon systems of the eastern Gulf of California coast where we identify and assess the principal environmental and socioeconomic factors that cause mangrove vegetation vulnerability.

We analyzed changes in mangrove cover in the two lagoon complexes Guásimas-LosAlgodones-Lobos (GLL), Sonora and Navachiste-San Ignacio-Macapule (NSM), Sinaloa, using LANDSAT (1990 and 2003) satellite images and SPOT 5 (2009). Forest cover maps were generated with ERDAS Imagine and Arc/Gis 9.1 programs. We used unsupervised (isodata algorit) and supervised (classes) classification methods; after that,

* Corresponding author: Teléfono y fax (951) 5170400 y 5170610. Correo electrónico: mmoraf@ipn.mx.

we could detect changes and used the Puyravud Deforestation Analysis Rate (2003) during the 1990-2003 and 2003-2009 periods.

As a result, we detected coverage changes in the GLL lagoon complex, a decrease in specific areas from 1990-2003 where the lost rate was positive (from 0.5 to 2.1). From 2003-2009 the main values were negative, with the highest rate of -0.22 where there was an important increase in areas with little anthropogenic influence, possibly related to the increase of rainfall in that period. Changes during the first period were due to precipitation increase and cover loss and to natural cycles more than to socioeconomic risks in the second period.

In the case of the NSM lagoon complex, we observed mangrove cover reduction in 1990, 2003, and 2009. Negative rates were higher from 2003-2009 (-0.22) even when the weather was less arid with important cyclonic events including a major contribution of rainfall. The socioeconomic vulnerability analysis showed affectation by agricultural activities and contaminants draining to the system.

INTRODUCTION

Mangroves are capable of absorbing or reorganizing stress effects to maintain their function, processes, and structure (Gilman, et al., 2008); in spite of their ecological resilience and their capability to adapt to environmental conditions, they face the risk of climate change (Morris et al., 2002; IPCC, 2007). In this respect, the expected effects of this change on a worldwide scale are increasing concentrations of CO₂, CH₄, and N₂O, such as an increase in temperature, a higher incidence of water filtration into soil, drought, ocean acidification, sea level rise, and incidence of hurricanes (IPCC, 2007). Particularly, an accelerated speed of sea level rise will probably have an impact on vegetation loss toward the sea, and inland or a lateral shift that will depend on the contribution of hydrology, sediments, and anthropogenic effects on landscape. Besides, the vulnerability of mangrove systems does not depend exclusively on the interaction with environmental danger, so the bonding between socioeconomic structures, as well as cultural norms and social principles, should also be considered.

All along the Gulf of California (GC) coastline, an extensive distribution of mangroves are subjected to a greater pressure as a consequence of intensified anthropogenic activities, such as farming, ranching, fishing, aquaculture, tourism, terrestrial forestry, thermoelectric industries, salt mine construction, urban and industrial development, distribution of energy routes, as well as the construction of large scale reservoirs, highways, navy piers, or even eolian project development. It is important to recognize the interaction between nature and culture in order to evaluate ecosystem vulnerability, thus bonding between socioeconomic structures and environmental danger, as well as cultural norms and social principles, should be considered (Oliver-Smith, 2003; Shiva, 2007). Knowledge of ecosystem integrity and its exposure both to natural and anthropogenic dangers constitutes the fundamental basis in terms of ecosystem vulnerability. Therefore to calculate its vulnerability, the components that may allow obtaining quantitative estimations about ecological resource condition, stress magnitude and the amount of exposure under variant conditions must be defined. Vulnerability evaluations fulfill the purpose of integrating social, ecological, and economic information (Pratt et al., 2004).

As the level of ecosystem vulnerability depends on particular regional conditions, in this chapter we discuss case studies of two lagoon complexes located at the eastern coast of the

Gulf of California, both sharing the characteristics of being located at arid zones where main environmental and socioeconomic factors responsible for mangrove vegetation vulnerability are identified and evaluated.

MATERIALS AND METHODS

Guasimas-Los Algodones-Lobos lagoon complex is located at $27^{\circ}11'$ and $27^{\circ}55'$ LN and $110^{\circ}17'$ and $110^{\circ}40'$ LW at southern Sonora, Mexico; it occupies a territorial extension of 35,198 ha. The main weather in that area is arid with an evaporation rate of about 2,500 mm per year and a low total precipitation lower than 300 mm per year, and annual average temperature of 24°C . The second lagoon complex is Navachiste-San Ignacio-Macapule located at $1^{\circ}17'$ and $25^{\circ} 40'$ LN and $108^{\circ}25'$ and $109^{\circ} 05'$ LW at Sinaloa, Mexico with a territorial extension of 79,873 ha. Weather in this area tends to be very warm and dry with certain rainfall during summer. Annual average precipitation is approximately from 365 to 450 mm with an evaporation rate of 2,000 mm per year, and annual environmental temperature averaging around 23.5°C . To determine variation in mangrove vegetation on each system, we used LANDSAT (April 1990 and May 2003) and SPOT 5 (March, April, and May 2009) satellite images; forest cover maps were generated with ERDAS Imagine and Arc/Gis 9.1 programs by using unsupervised (isodata algorithm) and supervised (classes) classification methods, by which values for Deforestation Rate $r = (1/(t_2-t_1)) * \ln(A_2/A_1)$ (Puyravaud 2003) were obtained (1990-2003 and 2003-2009). In the case of Vulnerability Index, an extensive compilation of available information for that specific region was made based on Pratt et al., (2004) criteria. As a fact evaluating mangrove vulnerability implies several risks together with poor data availability; ecological resilience and integrity of an ecosystem are very complex to be determined by using indicators which provide information to estimate vulnerability (Table 1). Measurement units were converted to intensity or frequency algorithms on a scale where 1 means the lowest possible and 7 meaning the highest possible appearance.

RESULTS

When analyzing mangrove coverage variations on the lagoon complex GLL, two contrasting variations in landscape were detected; while during the period 1990-2003 forestation rates were highly positive (green), which means a vegetation increase (0.5 – 2.1), during the period 2003-2009 forestation rates were preferentially negative (up to 0.22) (Figure 1). A most evident vegetation increase was observed in areas with low anthropogenic influence: Guasimas lagoon and Los Algodones estuary (Yaqui's ethnic group settlement), in contrast with the Lobos lagoon that showed a higher number of negative forestation areas where human activities have more influence (human settlement both native and racially mixed, farming drainage systems, and fishing spots) (Figure 1). In respect to vulnerability determining factors, rainfall was higher from 1990 to 2003 than in other years, which means a greater amount of fresh water input favoring growth in mangroves during that period. On the second period from 2003-2009, negative forestation rates were detected (yellow and red),

agreeing with a register of a seasonal drought particularly in that region in 2005 (CONAGUA, 2008). Besides, human activities implied a greater demand on fresh water (drinking water and for intensive agricultural use), tendency that could drastically affect mangroves in arid zones in case it continued. The Yaqui River natural flow used to provide enough fresh water during rainy seasons. Nevertheless, this fact has been dramatically modified since 1941 as a consequence of the construction of the Lázaro Cárdenas or Angostura, Plutarco Elías Calles or El Novillo and Oviáchic or Álvaro Obregón reservoirs; same case as the grooving in the Yaqui agricultural valley that now makes fresh water flow among canals and farming waste water pipes which unload sediments, fertilizers as well as pesticides right into the lagoons. Currently, lagoons constitute the only fresh water input for Los Algodones, La Luna, and Lobos; the last one receives an important amount of agricultural and industrial wastes directly from the main canal coming from the Yaqui valley and Obregón City. Pest-control substances are applied to almost all agricultural crops in the area, just as in the case of the Yaqui valley where 355,438 L of the pesticide *DIMETOATO* 40® were applied during the 2007-2008 seasons. On the other hand, presence of organic chloride, phosphorus and carbon compounds, as well as agricultural products and several organisms in water were detected showing collateral effects in residents and aquatic fauna (Macías-Zamora et al., 2008). Such a drastic variation in the river flow implies an important change in the amount of sediment input, as well as a variation of the GLL lagoon complex water cycle pattern. This fact actually agrees with some other regions where water cycle variations, consequence of reservoir construction and incidence of anthropogenic activities as farming, causes ecological and socioeconomic degradation due to river flow changes, which transform mangrove habitats, distribution, floral composition, and biodiversity by means of hydro-geomorphic alterations (Dahdouh-Guebas et al., 2005, 2006).

Likewise, underground water extraction has caused soil salinization and important phreatic mantle disturbances all along the valley (Nauman, 2006). Nowadays, the ambition of diverting part of the Yaqui River water resource, which naturally flows into the lagoons, toward the city of Hermosillo, could cause a greater increase in vulnerability.

Particularly in the case of NSM, the forestation rates obtained for both analyzed periods were mainly negative, representing a continuous loss in a period of 29 years; the highest deforestation rates were estimated during the second period (2003-2009) where high values were reached = -0.22 (Figure 2) in spite of the difference in weather conditions. NSM is less arid than GLL and in addition a higher number of cyclonic events occurred as well as a greater amount of rainfall in that region. Important landscape variations were detected specially in areas contiguous to drainage system canals, swamp zones at Navachiste lagoon and into the area where most of the shrimp production farms were located; those variations were even more noticeable at Batamote, Novobampo, and San Antonio drains which flow directly into the lagoons. When comparing mangrove coverage loss and socioeconomic vulnerability, the results of the analysis show that agricultural activities are responsible for the great vulnerability.

Agricultural activities not only imply changes in the amount of sediment input as well as a reduction in fresh water supply to mangrove ecosystems but also and most importantly in the use of pesticides, weed killer substances, and inorganic fertilizers. These harmful substances are scattered from a plane by using dispersal mechanisms, causing their residues to flow directly through the canals into the lagoons, which is the reason why significant amounts

of DDT, DDD, DDE, and HCB have been detected on smooth tissues of mollusks living in these lagoon systems (Osuna et al., 2009).

Table 1.

Year	Name	System	Category	Wind(mph)
1981	Lidia	SNM	TS	40.00
1982	Paul	SNM	2	100.00
1985	Waldo	SNM	1	75.00
1986	Newton	Both	1	75.00
1987	Paine	SNM	1	90.00
1990	Rachel	SNM	TS	60.00
1993	Hilary	GLL	TD	35.00
1995	Ismael	both	1/TS	70.00
1996	Fausto	both	1/TS	75.00
1998	Isis	both	TS	70.00
2001	Juliette	GLL	TD	35.00
2004	Javier	GLL	TD	25.00
2006	Paul	SNM	TD	30.00
2006	Lane	SNM	TS	40.00
2007	Henriette	GLL	TS	70.00
2008	Norbert	both	1	85.00

GLL= Guasimas-Los Algodones-Lobos, Son. SNM=San Ignacio-Navachiste-Macapule, Sin. TS= storm, TD= Tropical depression.

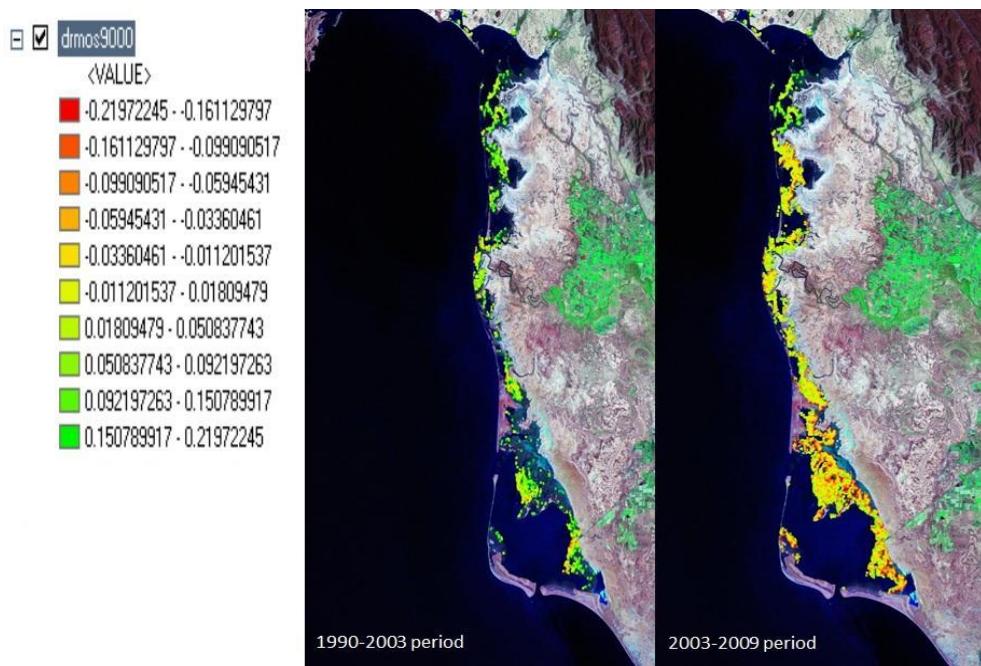


Figure 1. Forestation rates of Laggons Guásimas-Los Algodones-Lobos, Son. Mexico during 1990-2003 and 2003-2009, from images LANDSAT and SPOT. Green colours represent positive values, yellow and red colours represent negative values.

Another socioeconomic activity that affects vulnerability directly is aquaculture by an increasing number of shrimp production farms during both analyzed periods; all of them are located near mangrove vegetation (about 10,000 ha), mainly on swamp areas and seasonal flooding plains. Now only 50% of these farms remain active while all the others have been abandoned due to a drastic decrease on shrimp commercial value as well as by the presence of a virus known as “white spot”. It has been widely documented that aquaculture has contributed to mangrove fragmentation by cutting down important mangrove extensions along with herbal and bushy vegetation; in addition, supplying canals and access paths change water cycle patterns at flooding areas while reducing potential areas for mangrove displacement (swamp areas), which are considered as cushioning zones for vegetation in case of a possible sea level rise as a consequence of climatic change (Agráz-Hernández et al., 2001; DUMAC 2005; Gilman et al. 2008).

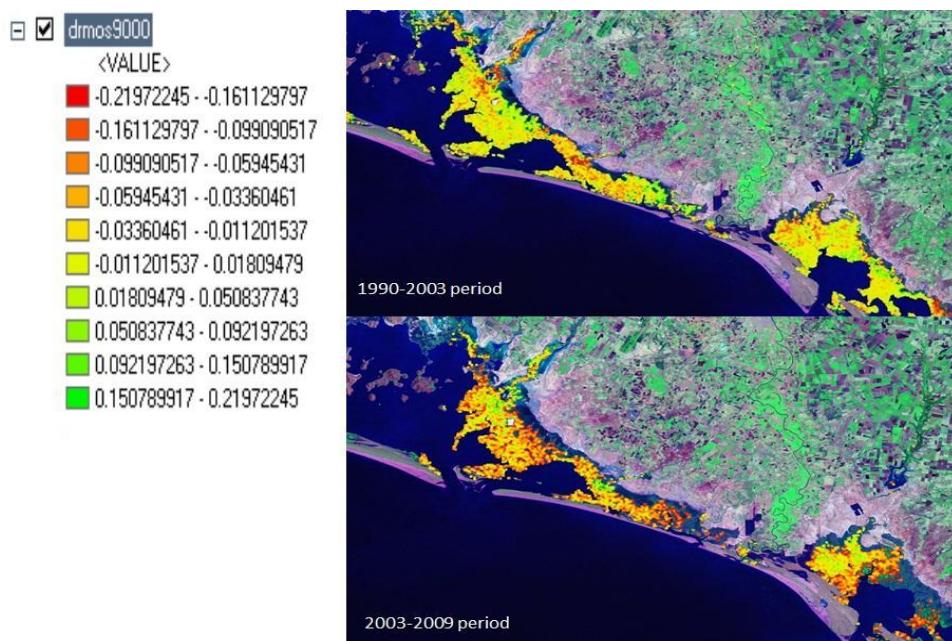


Figure 2. Forestation rates of Laggons Navachiste-San Ignacio-Macapule, Sinaloa, Mexico during 1990-2003 and 2003-2009, from images LANDSAT and SPOT. Green colours represent positive values, yellow and red colours represent negative values.

Hurricanes affect mangrove ecosystems through four different mechanisms: winds, waves, sediment burial, and water levels. Each one of them has a frequency and intensity of its own, which is the reason why the analysis was made independently from the specific value given at the scale of hurricanes (Table 2) because the impact response and resilience also depend on the particular characteristics of each cyclonic event. In this respect, hurricanes do not pass directly into the region, but they contribute indirectly in the amount of water input into the catchment area, whose indirect effect in terms of water supply depends on the ecosystem position in relation to watersheds; strong rains that may cause erosion from the highlands bringing about flooding and rubble flow down to lower territories. Structural ecosystem changes, as well as some other arrangement variations, occur as a result of

hurricane incidence on land besides other alterations that occur, as in age, distribution, size, composition, biomass, species diversity, succession processes, nutrient cycles in plants and animal interactions between different species.

Researchers have reported that mangroves do not fit perfectly into the concept of ecological succession, which can be explained by pointing out that these ecosystems have suffered periodic disturbances along geological periods and their recovery depends on the seasonal incidence of a natural disturbance (hurricane). Negative deforestation rates, particularly for the NSM lagoon complex besides all the other reasons previously analyzed, show that anthropogenic activities such as agriculture, aquaculture, water cycle modifications, and seasonal raining patterns as well as hurricane incidence due to climate change are greatly contributing to the loss of this ecosystem as a result of alterations in natural disturbance incidence (Dahdouh-Guebas et al., 2005, 2006).

CONCLUSION

Coverage variations for the GLL complex lagoon are mainly attributed to natural cycles than to socioeconomic risks; nevertheless, an increasing tendency in human activities for last period including climate change would have probably increased vegetation loss which could be responsible for an increment in vulnerability of the ecosystem. It is also important to point out the continuous loss in mangroves for the NSM lagoon complex where a high vulnerability due to human activities was registered. In both cases, coverages lost has been related to human activities which turn out to be particularly serious because nowadays public policies are heading toward progress and development where ecosystems are considered as merchandise with service values without considering the real complexity of their culture (Illich, 2008; Shiva, 2008).

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Chapter 9

STRUCTURE OF THE ICHTHYOFaUNA ASSOCIATED WITH SOFT BOTTOMS IN SAN IGNACIO LAGOON, BAJA CALIFORNIA SUR, MEXICO

***E. Barjau-González,^{*1} F. Galván-Magaña^{2#}
and E. Romero-Vadillo¹***

¹Academic Department of Marine Biology, UABCs. La Paz, Mexico

²Interdisciplinary Center of Marine Sciences. Baja California Sur, Mexico

ABSTRACT

San Ignacio Lagoon is located within the Vizcaíno Biosphere Reserve. There has been limited study of its ichthyofauna to date; the objective of this chapter is to describe the composition and distribution, as well as the temporal and spatial abundance of the ichthyofauna associated with soft bottoms in this coastal lagoon. A total of 44 trawls were carried out over a 346,500 m² area. A total of 11 trawls per season were carried out, covering an 86,625 m² area (7,875 m² per trawl). A total of 1,361 organisms were collected, with a total biomass of 67,118 g. The ichthyofauna of this lagoon was composed of 21 families, 35 genera, and 44 species. The highest abundance of organisms and biomass was obtained during the autumn (34.6% and 30.4%, respectively), while the lowest abundance and biomass were obtained in the spring (16.5% and 13.5%, respectively). Of the 44 recorded species, eight were present during the overall study period (*S. annulatus*, *S. lispus*, *E. dowii*, *P. maculatofasciatus*, *Urobatis halleri*, *Hypsopsetta guttulata*, *Hippocampus ingens*, and *Paralichthes californicus*). The Biological Value Index showed that 25 species were dominant in this ecosystem. The species diversity was highest in winter (2.28 bits/ind.) and lowest during the autumn (0.41 bits/ind.). Water temperature was similar in spring and autumn (21.1°C and 20.9°C, respectively), and was higher in the summer (24.9°C). The zoogeographic affinity analysis showed that the provinces with the highest number of records were the Panamic (36%) and San Diego (31.8%) Provinces.

* ebarjau@uabc.mx.

galvan.felipe@gmail.com.

Keywords: San Ignacio Lagoon, soft bottoms, fish community structure

INTRODUCTION

The western coast of the Baja California Peninsula is one of the most important coasts in Mexico in terms of fishery production, representing 36% of the total volume of catches in Mexico. Of the over 100 coastal water systems (coastal lagoons and estuaries) in Mexico, 22 lagoon-estuary systems are located in the Baja California Peninsula (Lankford, 1977). Coastal lagoons and estuaries are physically unstable areas, characterized by spatial and temporal variations in water temperature, salinity, oxygen concentration, turbidity, and other factors (Yañez-Arancibia, 1977; Day, et al. 1989). The coastal lagoons of Baja California Sur have no freshwater influx due to low precipitation, and are therefore considered anti-estuarine.

The environmental complexity of coastal lagoons, the low incidence of large piscivorous predators, and the continual influx of nutrients provide numerous benefits for fish communities in these areas (Claridge, et al. 1989). As a general objective, the present chapter includes a description of the composition, distribution, temporal, and spatial abundance and aims to determine the biogeographic affinities of the ichthyofauna associated with soft bottoms in San Ignacio Lagoon, B.C.S.

MATERIALS AND METHODS

San Ignacio Lagoon is located on the western coast of the Baja California peninsula, Mexico, between $26^{\circ}43'$ and $26^{\circ}58'$ North Latitude and $113^{\circ}08'$ and $113^{\circ}16'$ West Longitude. It is a shallow lagoon, with depths between 2 and 4 m, and 20 m deep channels that connect with the ocean (Swart & Cummings, 1978). The inside of the lagoon has a system of channels that are separated by extensive areas of sand banks, which are exposed during low tide (Swart and Cummings, 1978; Reitherman and Storror, 1981). The coast has sand beaches, muddy banks, mangroves, marshes, and a few rocky areas. The northern part of the lagoon consists of the main water body, while a secondary branch extends towards the west part and includes most of the mangrove areas. San Ignacio Lagoon is situated within the Vizcaíno Biosphere Reserve, which is of great national importance, as grey whales come to this area to reproduce and because migratory marine birds seasonally inhabit the area (Figure 1).

Four seasonal sampling trips were carried out in the spring, summer, and autumn of 1998 and the winter of 1999. A total of 44 trawls lasting 30 minutes in duration were carried out over 11 stations. Organisms were caught using an experimental trawl net that was 9 m long, with a 4.5 m opening, 1.5-inch mesh size, and 95x50 cm metallic doors. The towing speed was 3.5 km/h; the net was towed using a 22-foot boat with a 75 HP outboard motor.

The bottom temperature ($\pm 0.1^{\circ}\text{C}$) and dissolved oxygen concentration ($\pm 0.1\text{mg/L}$) were measured using a YSI model 63; salinity was measured using a refractometer Atago ATC-S /Mill-E on a scale of 0‰ to 100‰ ($\pm 1\%$). These parameters were measured at each station at the end of the trawl.

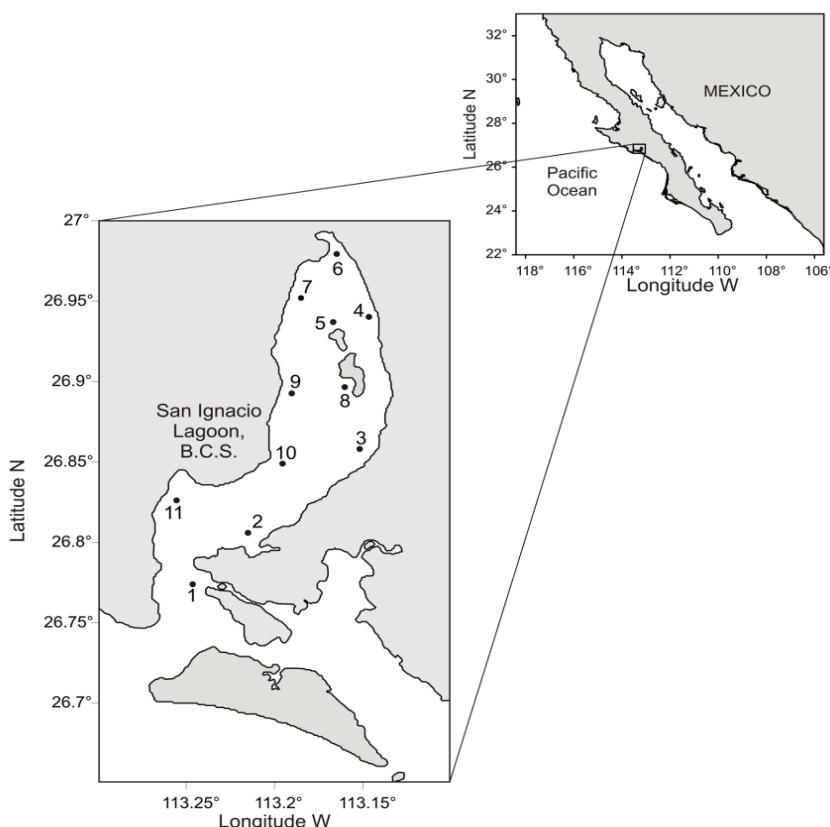


Figure 1. Location of sampling stations in San Ignacio Lagoon, Baja California Sur, Mexico. Canal del Cardón (1), La Freidera (2), La Base (3), El Anegado (4), Norte Isla Garza (5), El Remate (6), Cantil Cristal (7), La Choya (8), Los Cerritos (9), Las Islitas (10), El Mapache(11).

The length and weight of each fish captured at San Ignacio Lagoon were recorded and the fish were transported to the Autonomous University of Baja California Sur for species identification using specialized keys [Jordan & Evermann, (1896,1900); Meek & Hildebrand, (1923,1928); Miller & Lea, (1972); Norman, (1934); Curran, 1942; Zahuranec,(1967); Whitehead, (1985); Allen & Robertson, (1994); Fischer, et al. (1995); Walker & Bussing, (1996); and Thomson, et al. (2000)].

Relative abundance is a mathematical expression used by several authors, including Horn and Allen (1985), Subrahmanyam (1985), Ramirez (1989), Barjau-Gonzalez (2003), and Barjau-Gonzalez, et al. (2012) to show, using percentages, which species conform to the community structure and its variations over time. The relative abundance index was estimated based on the number and total biomass of the captured fish. The index of relative abundance was estimated by the expression: $RA = n / TN * 100$, where n = number of organisms of each species and NT = total number of organisms.

To determine the dominant species in San Ignacio Lagoon, the Biological Value Index proposed by Sanders (1960) was used.

To calculate the species diversity, the Shannon-Wiener index (H') was used (Shannon and Weaver, 1963), as it is widely used in cases where samples are obtained using nets, traps,

or transects (Krebs, 1978), and because it allows for the comparison of results with similar studies carried out in coastal lagoons. This index is represented by the following expression:

$$H' = \sum_{i=1}^s p_i \log^2 p_i$$

where: H' = Shannon-Wiener diversity index, p_i = proportion of species i , and s = number of species in the sample.

The relative frequency of occurrence allows for the direct measurement of constancy of species occurrence per trawl, and is considered a complementary index to those previously mentioned by Perez Mellado and Findley (1985).

To determine the Biological Value Index (BVI) proposed by Sanders (1960), one of the features of this index is the assignment of points based on the relative abundance of species in each sample. A species is considered dominant when it shows consistency in its abundance and frequency of occurrence (Loya-Salinas & Escofet 1990).

An analysis of species presence/absence was conducted to determine the distribution of species in each locality of the study area.

The zoogeographic affinity of fish was analyzed according to the basic scheme proposed by Briggs (1974), with modifications by Hastings (2000), Robertson et al. (2004), and Horn et al. (2006), and considered the following provinces and affinity:

1. San Diego Province: from Point Conception, California to Magdalena Bay off the western coast of Baja California Sur. Corresponding affinity: warm-temperate.
2. Cortez Province: endemic fish from the Gulf of California, taking Cabo San Lucas on the western coast and Mazatlán on the eastern coast of the Gulf of California as the southern limit. Corresponding affinity: warm-temperate and subtropical.
3. Mexican Province: fish found from Magdalena Bay or Cabo San Lucas to the Gulf of Tehuantepec, Oaxaca. Corresponding affinity: subtropical.
4. Panamic Province: fish found in the subtropical-tropical zone, between 23° N and 5° S (Peru). Corresponding affinity: tropical.
5. Eastern Pacific: fish with a wide distribution in the Tropical Eastern Pacific from California to Peru.
6. Circumtropical: fish with a wide distribution in tropical areas.
7. Indo-Pacific: fish with a transpacific distribution.

RESULTS

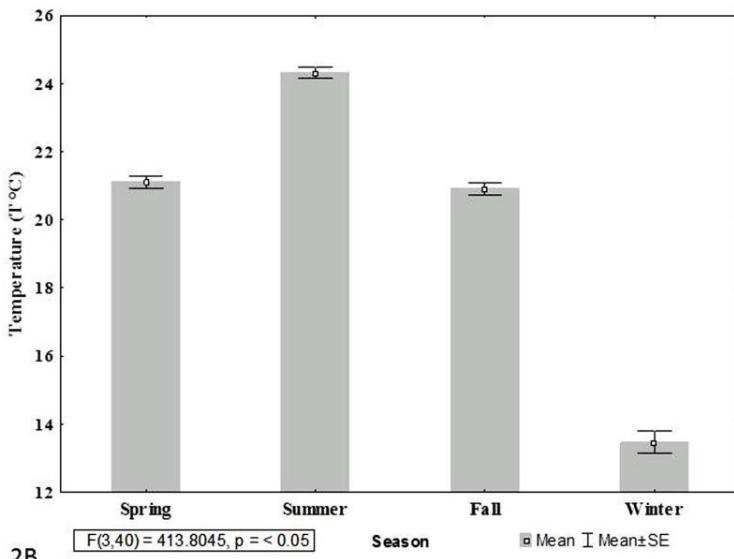
Temperature: A seasonal variation in bottom temperature over the sampling period was observed. The maximum temperature was 25°C in summer and the minimum was 11.2°C in winter; the thermal variation was $\pm 13.8^\circ\text{C}$. In general, temperature had a well-defined seasonal pattern, increasing gradually to reach the highest values in summer (Figure 2A).

Salinity: San Ignacio Lagoon is classified as a hypersaline lagoon, which implies that water with higher salinity is denser and flows at the bottom of the water column towards the ocean, while less saline water flows towards the inside of the lagoon on the surface. During

summer and fall, maximum salinity was found on the eastern side of the islands towards the bottom (station 6 “El Remate”), reaching values of 42‰ (Figure 2B).

There was a pronounced gradient in dissolved oxygen (DO) over the annual cycle at San Ignacio Lagoon. The highest values occurred in spring and fall (4.18 ml/l), and the lowest values were recorded in winter (1.4 ml/l). The average DO for the overall study period was 3.33 ml/l (Figure 2C).

2A



2B

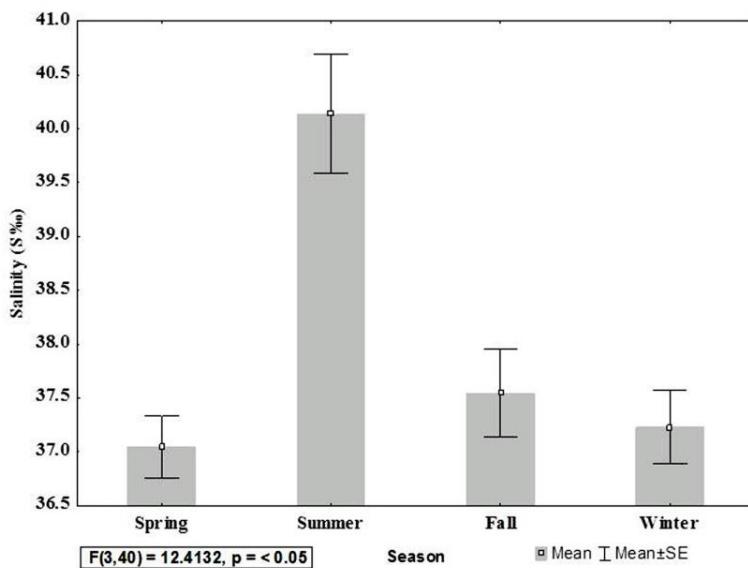


Figure 2A, 2B and 2C. (Continued).

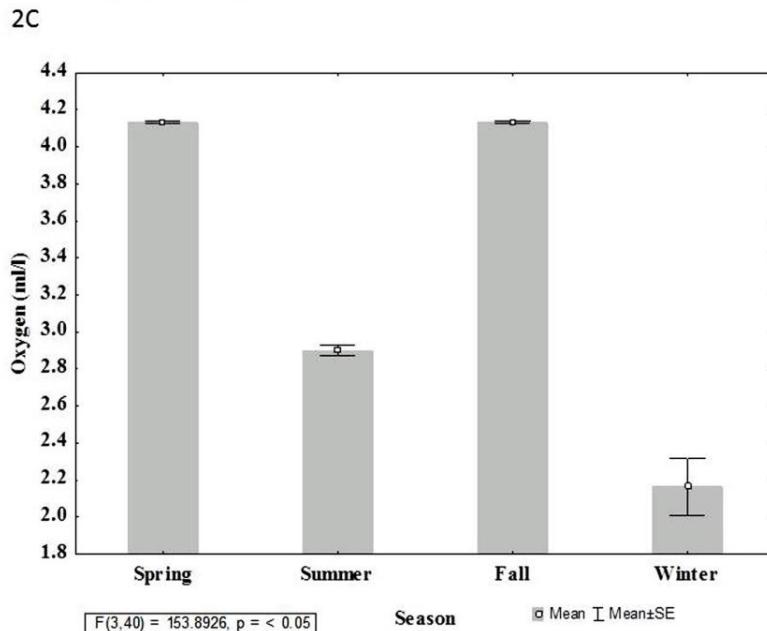


Figure 2A, 2B and 2C. Water temperature, salinity and dissolved oxygen by season in San Ignacio Lagoon, Baja California Sur, Mexico.

A total of 1,361 organisms was caught during the study period, which is equivalent to a biomass of 67,118 g. Total biomass density was 1,938.3 g/ha, individual sizes ranged from 6.6 mm to 544 mm in total length, and weight ranged from 2 g to 1,874 g.

The fish composition consisted of 44 species belonging to two classes, seven orders, 21 families and 35 genera. The families with the highest numbers of species were Serranidae, Haemulidae, and Sciaenidae with four species each. Engraulidae, Bothidae, Gerreidae, and Tetraodontidae had three species each, and the remaining families had either one or two species.

The relative abundance (RA) showed that 16 species were the most abundant, comprising 86.3% of the total. Of these, nine species made up 79.9%, as follows: *Paralabrax auroguttatus* (16.1%), *Sphoeroides lispus* (12.9%), *P. maculatofasciatus* (9.8%), *P. nebulifer* (8.1%), *S. annulatus* (5.5 %), *Eucinostomus dowii* (5.2%), *Bairdiella icistia* (4.9%), *Exerpes asper* (4.6%), and *Orthopristis reddingi* (4.5%) (Table 1).

The specific composition by season showed that of the total biomass (67,118 g), 20,433.4 g were caught during autumn and 19,968.7 g were caught in winter. The biomass in the summer was 17,786.2 g and in spring it was 8,929.7 g (Table 1).

The Biological Value Index showed that there were 25 dominant species; species comprising more than four percent, according to BVI, were as follows: *P. auroguttatus*, *S. lispus*, *P. maculatofasciatus*, *P. nebulifer*, *E. dowii*, *S. annulatus*, *B. icistia*, *Exerpes asper*, *Orthopristis reddingi*, *Hypsoblennius gentilis*, *Cyclopsetta panamensis*, and *Hypsopsetta guttulata* (Figure 3, Table 1).

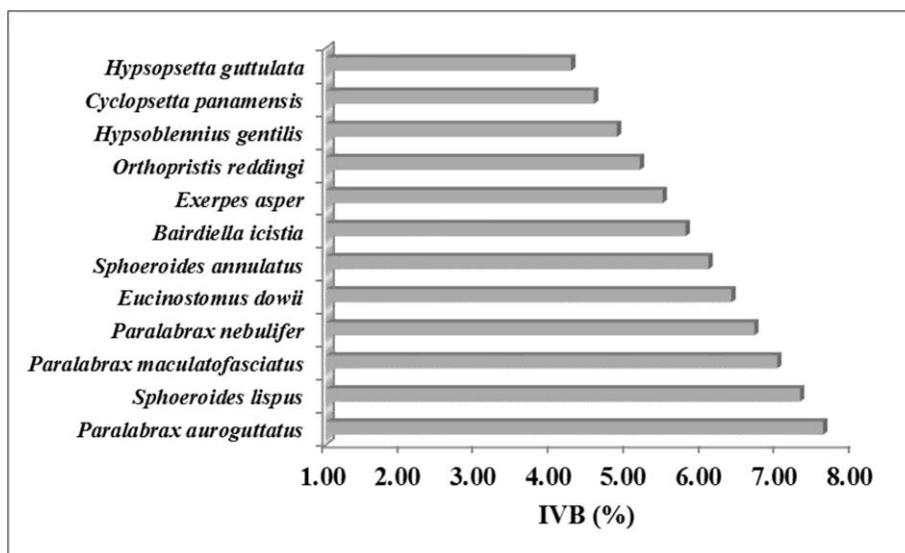


Figure 3. Dominant species during the sampling period, according to the Biological Value Index, in San Ignacio Lagoon, Baja California Sur, Mexico.

Table 1. General relative abundance (RA) and relative weight (RW) by season. Biological Value Index (BVI) of fish captured in San Ignacio Lagoon, Baja California Sur, Mexico

Species/ Season	Spring		Summer		Autumn		Winter		All Seasons		IBV
	RA	RW	RA	RW	RA	RW	RA	RW	RA	RW	
<i>Achirus mazatlanus</i>	0.89	0.93	0	0	1.68	1.84	2.95	4.56	18	1369.5	9
<i>Anchoa compressa</i>	8.44	4.32	0	0	0.00	0.00	0.00	0.00	19	386	10
<i>Anchoa ischana</i>	0.00	0.00	1.54	0.62	0.00	0.00	0.00	0.00	6	111	
<i>Atherinops affinis</i>	0.00	0.00	2.05	0.06	0.00	0.00	0.00	0.00	8	10.7	1
<i>Bairdiella icistia</i>	13.78	12.20	5.64	3.60	2.95	0.37	0.00	0.00	67	1804.6	19
<i>Calamus brachysomus</i>	0.00	0.00	0.51	0.12	0.63	0.12	0.37	0.12	6	67.7	
<i>Cetengraulis mysticetus</i>	0.89	0.00	0.00	0.00	1.89	0.00	0.00	0.00	11	1.3	5
<i>Chaetodipterus zonatus</i>	0.00	0.00	0.77	0.77	0.84	0.29	0.00	0.00	7	195.4	
<i>Cyclopsetta panamensis</i>	0.00	0.00	0.00	0.00	0.00	0.00	12.55	0.85	34	169	15
<i>Etropus crossotrus</i>	2.67	0.33	0.26	0.05	0.21	0.17	0.00	0.00	8	74.3	
<i>Eucinostomus dowii</i> *	4.89	1.20	10.51	3.22	1.26	0.20	4.80	1.36	71	991.5	20
<i>Eucinostomus entomelas</i>	0.00	0.00	0.00	0.00	0.00	0.00	4.06	0.85	11	170.5	5
<i>Eucinostomus gracilis</i>	8.44	2.91	0.77	0.08	0.00	0.00	0.37	0.02	23	277	12
<i>Exerpes asper</i>	2.22	0.11	0.00	0.00	11.79	0.16	0.74	0.01	63	44.7	18
<i>Haemulon californiensis</i>	1.78	0.54	0.51	0.11	0.00	0.00	0.00	0.00	6	67	
<i>Halichoeres semicinctus</i>	0.00	0.00	0.00	0.00	0.21	0.22	0.00	0.00	1	44	
<i>Heterodontus francisci</i>	0.44	9.97	0.26	2.96	0.00	0.00	0.37	3.58	3	2131.5	
<i>Hippocampus ingens</i> *	0.44	0.49	0.77	0.74	0.84	0.70	0.74	0.38	10	395.7	2
<i>Hoplopagrus geuntheri</i>	0.00	0.00	0.00	0.00	0.21	0.89	0.00	0.00	1	180.9	
<i>Hypsoblennius gentilis</i>	0.00	0.00	0.26	0.12	9.47	2.57	0.00	0.00	46	545.6	16

Table 1. (Continued)

Species/ Season	Spring		Summer		Autumn		Winter		All Seasons		IBV
	RA	RW	RA	RW	RA	RW	RA	RW	RA	RW	
<i>Hypsopsetta guttulata</i> *	8.00	0.75	0.77	2.02	1.05	3.49	1.48	3.39	30	1816.8	15
<i>Menticirrhus undulatus</i>	0.44	1.88	0.00	0.00	0.00	0.00	0.00	0.00	1	168	
<i>Microlepidotus inornatus</i>	0.00	0.00	1.03	0.35	0.00	0.00	0.00	0.00	4	62	
<i>Mycteroptera xenarcha</i>	0.00	0.00	0.00	0.00	0.63	0.42	0.00	0.00	3	86.2	
<i>Ophistonema libertate</i>	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.43	1	86	
<i>Orthopristis reddingi</i>	2.22	1.16	7.95	3.99	5.47	1.00	0.00	0.00	62	1018.2	17
<i>Paralabrus integripinnis</i>	0.44	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1	1	
<i>Paralabrax auroguttatus</i>	4.44	0.94	0.00	0.00	34.95	27.66	15.87	6.39	219	7010.3	25
<i>Paralabrax maculatofasciatus</i> *	4.44	12.44	11.28	2.46	15.16	25.55	2.95	4.33	134	7633.2	23
<i>Paralabrax nebulifer</i>	5.78	1.38	24.87	10.24	0.00	0.00	0.00	0.00	110	1944.5	22
<i>Paralichthys californicus</i> *	2.67	0.35	0.77	0.15	0.84	4.89	2.95	15.97	21	4245.3	11
<i>Pleuronichthys verticalis</i>	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.22	1	44	
<i>Pomadasys panamensis</i>	0.00	0.00	0.00	0.00	0.00	0.00	4.06	0.42	11	84.4	5
<i>Pseudopenneus grandisquamis</i> .	3.11	2.71	2.05	2.53	0.00	0.00	0.00	0.00	15	691.8	6
<i>Quietula y-cauda</i>	1.33	0.03	0.00	0.00	0.21	0.00	0.37	0.00	5	3.7	
<i>Scorpaena mystes</i>	0.44	0.47	0.00	0.00	0.21	1.03	0.00	0.00	2	252.4	
<i>Sphoeroides annulatus</i> *	6.67	16.48	4.10	12.00	2.95	17.36	11.07	19.13	75	10972.8	21
<i>Sphoeroides lispus</i> *	12.89	14.56	9.74	10.34	5.89	6.82	29.89	19.31	176	8387.2	24
<i>Sphoeroides lobatus</i>	0.00	0.00	0.51	0.51	0.00	0.00	0.00	0.00	2	90.5	
<i>Syngnathus auliscus</i>	0.44	0.02	0.00	0.00	0.21	0.00	0.00	0.00	2	2.8	
<i>Umbrina roncador</i>	0.00	0.00	4.62	2.74	0.00	0.00	0.00	0.00	18	487.5	9
<i>Umbrina xanti</i>	0.00	0.00	1.79	0.70	0.00	0.00	0.00	0.00	7	124	
<i>Urobatis halleri</i> *	0.44	2.82	4.36	30.30	0.42	4.26	1.85	10.39	25	8587.4	13
<i>Urobatis maculatus</i>	1.33	11.00	2.31	9.23	0.00	0.00	1.85	8.30	17	4280.1	7

In the analysis of diversity, the overall diversity measured was 2.97 bits/ind.; the climate season with the highest diversity was spring, with 2.83 bits/ind., then summer at 2.57 bits/ind., followed by winter at 2.28 bits/ind. and autumn, with the lowest, at 2.19 bits/ind.

The relative frequency of occurrence gives a direct measure of the consistency of species occurrence in the 11 trawls carried out during each season. Of the 44 species recorded over the course of all the sampling, eight species were present during the entire study: *S. annulatus*, *S. lispus*, *E. dowii*, *P. maculatofasciatus*, *Urobatis halleri*, *Hypsopsetta guttulata*, *Hippocampus ingens*, and *Paralichthys californicus* (marked with asterisks (*) in Table 1).

To determine the distribution of the 44 reported species, an analysis of presence/absence was carried out. This analysis indicated that 11 species were present during three seasons, six species were present in two seasons, and the remaining species were present during only one of the four seasons. Of the 44 species, 61.4% were captured in the spring, 59.1% in the summer, 54.5% in the fall, and 47.7% in the winter (Table 2).

Biogeographic affinity: The zoogeographic analysis yielded 16 species (36%) belonging to the Panamic Province, 14 species (32%) belonging to the San Diego Province, 8 species (18%) belonging to the Eastern Pacific, 5 species (11%) belonging to the Mexican Province,

and one species (2%) listed as amphipeninsular, since it was found on both sides of the Baja California Peninsula (Figure 4).

Table 2. Spatial and temporal distribution of fish captured in San Ignacio Lagoon, Baja California Sur, Mexico. Spring (Sp.), Summer (Su.), Autumn (Au.), and Winter (Wi.)

Species	1	2	3	4	5	6	7	8	9	10	11	Sp.	Su.	Au.	Wi.
<i>Achirus mazatlanus</i>	P	P	P	P	P	P				P	P	P	P	P	P
<i>Anchoa compressa</i>				P	P						P				
<i>Anchoa ischana</i>				P				P				P			
<i>Atherinops affinis</i>									P		P	P	P	P	P
<i>Bairdiella icistia</i>	P			P	P			P	P	P	P	P	P	P	P
<i>Calamus brachysomus</i>	P	P		P	P		P	P	P	P	P	P	P	P	P
<i>Cetengraulis mysticetus</i>	P	P	P	P	P					P		P	P	P	
<i>Chaetodipterus zonatus</i>		P	P	P	P							P	P		
<i>Cyclopsetta panamensis</i>	P	P	P	P	P			P							P
<i>Etropus crossotus</i>		P		P	P			P		P	P	P	P	P	P
<i>Eucinostomus dowii</i>	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
<i>Eucinostomus entomelas</i>				P									P		
<i>Eucinostomus gracilis</i>	P				P	P			P	P	P	P			P
<i>Exerpes asper</i>					P				P	P	P	P	P	P	P
<i>Haemulon californiensis</i>		P		P				P	P	P	P	P			
<i>Halichoeres semicinctus</i>						P	P		P	P	P	P		P	
<i>Heterodontus franciscus</i>			P		P				P	P	P	P	P		P
<i>Hippocampus ingens</i>	P	P						P	P	P	P	P	P	P	P
<i>Hoplopagrus guentheri</i>	P								P		P	P	P	P	P
<i>Hypsoblennius gentilis</i>		P							P	P	P	P	P	P	P
<i>Hypsopsetta guttulata</i>	P	P	P	P				P	P	P	P	P	P	P	P
<i>Menticirrhus undulatus</i>		P		P					P	P	P				
<i>Microlepidotus inornatus</i>									P		P				
<i>Mycteroperca xenarca</i>									P			P			P
<i>Ophistionema libertate</i>						P									
<i>Orthopristis reddingi</i>				P	P	P		P	P	P	P	P	P	P	P
<i>Paraclinus integrifinnis</i>				P						P	P				
<i>Paralabrax auroguttatus</i>	P	P	P	P	P	P	P	P	P	P	P		P	P	P
<i>Paralabrax maculatofasc.</i>	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
<i>Paralabrax nebulifer</i>		P	P	P	P				P	P	P	P	P		
<i>Paralichthys californicus</i>	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
<i>Pleuronichthys verticalis</i>							P							P	P
<i>Pomadasys panamensis</i>						P									P
<i>Pseudupeneus grandisq.</i>		P	P		P	P	P	P	P		P	P			
<i>Quietula y-cauda</i>		P						P			P		P		P
<i>Scorpaena mystes</i>			P		P						P			P	
<i>Sphoeroides annulatus</i>	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
<i>Sphoeroides lispus</i>	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
<i>Sphoeroides lobatus</i>		P						P			P				
<i>Syngnathus auliscus</i>		P							P	P				P	
<i>Umbrina roncador</i>									P				P		
<i>Umbrina xanti</i>									P			P			
<i>Urobatis halleri</i>		P	P	P	P	P	P	P		P	P	P	P	P	P
<i>Urobatis maculatus</i>		P	P	P	P	P	P	P	P	P	P	P	P	P	P
Total species	4	7	15	19	17	20	23	11	8	21	20	27	26	24	21
Ratios												61.4	59.1	54.5	47.7

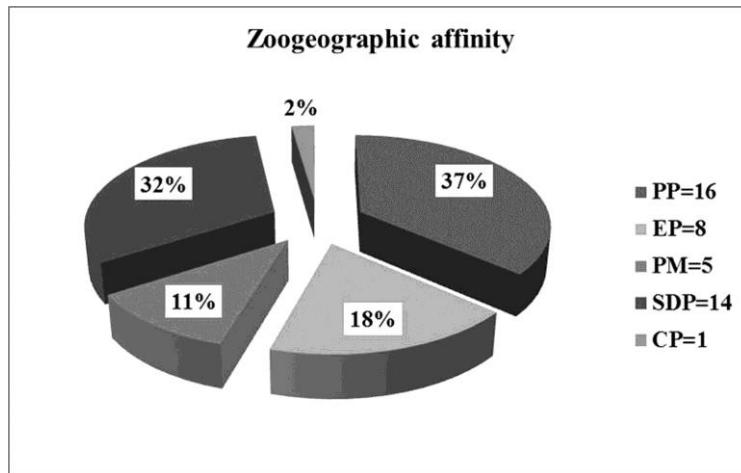


Figure 4. Zoogeographic affinity of fish species captured in San Ignacio Lagoon, Baja California Sur, Mexico.

DISCUSSION

Temperature had a well-defined temporal gradient; the maximum bottom temperature was recorded in summer and the minimum was in winter. The intermediate seasons, spring and fall, had similar temperatures (an average of 21°C). These results are consistent with the pattern recorded in Bahía Magdalena by Acevedo-Cervantes (1997) and Gutierrez-Sánchez (1997). These authors pointed out that bathymetry, tidal currents, warming, and non-uniform evaporation are the main factors that make this lagoon such a complex location to study. Nikolski (1963) determined that variation in surface temperature throughout the year in Ojo de Liebre Lagoon, B.C.S., was 13.5°C to 26°C. The results from San Ignacio Lagoon were relatively similar. The spring and fall seasons had a fluctuation of 0.2°C between sampling stations, while summer and winter temperatures varied an average of 10.8°C. Water temperature increased from spring to summer and then decreased towards winter, making spring and fall the transitional seasons between summer and winter, respectively. This type of situation was also recorded by Gutierrez-Sánchez (1997) in Magdalena Bay. The spatial temperature distribution showed that stations close to the mouth of the lagoon had lower temperatures than those towards the inside of the lagoon, which is also consistent with results reported by Gutierrez-Sánchez (1997) for Magdalena Bay and by Acevedo-Cervantes (1997) for Ojo de Liebre Lagoon.

The bottom salinity had a spatial and seasonal distribution gradient. The stations closest to the mouth of the lagoon were influenced by the ocean, making them less saline, while stations in the inside of the lagoon had higher salinity. The seasonal distribution had a similar pattern to the spatial distribution, in that stations close to the mouth of the lagoon were influenced by the ocean, with higher salinity at stations in the inside of the lagoon. A well-marked differentiation was observed, with maximum salinity during the summer and minimum salinity in the winter. Similar results were recorded by Gutierrez-Sánchez (1997) in Magdalena Bay, where salinities similar to those found in oceanic waters were recorded at the mouth of the lagoon, and increased towards the canal zone of Almejas Bay. The values of

salinity reported in this study support the observations of Largier et al. (1995), who classified this lagoon type as hypersaline.

Because the San Ignacio Lagoon is hypersaline (Largier et al. 1995), the dissolved oxygen (DO) in the bottom water behaved differently; the seasonal gradient was more pronounced than the spatial gradient. From values obtained during different seasons, it could be inferred that during spring, summer and fall, levels were relatively normal, perhaps due to an increase in temperature. Values obtained during the winter, however, were low, possibly due to the residence time of the water mass, as the time of renovation of the lagoon is slow (3 to 5 months). Gutierrez-Sanchez (1997) recorded low oxygen values in the inner margins of the Bahía Magdalena complex that were similar to those found in this study during the winter. The spatial distribution showed that in spring, oxygen concentrations did not change much, with the minimum concentration being recorded at Station 6 during the winter, which can be interpreted as there being higher oxygen consumption than production. In the fall, there was an increase in DO; a concentration of 3.5 mL/L was recorded at Station 6. In general, there was a gradient of DO with values that diminished from the mouth to the inside of the lagoon; in the middle part of the lagoon, there was no defined pattern (Barjau-Gonzalez 2003).

Variations in the recorded parameters were reflected in the distribution of the fish species captured, consistent with the results of Horn et al. (2006), who mentioned that temperature and salinity are the main factors that regulate the annual cycle of the number of species in California and adjacent waters. These factors also influence the distribution of the specific richness of fish populations (Nikolski, 1963). Temperatures and salinities were consistently higher towards the inside of the lagoon than towards the mouth, which indicates a hypersaline condition, where the anti-estuarine situation (evaporation) exceeds the freshwater inputs from drainage or precipitation. The gradients of temperature and salinity were generally consistent with the bathymetry of the lagoon, showing high values in shallow waters.

The taxonomic analysis of collected organisms showed that the fish community structure in San Ignacio Lagoon was composed of two classes, seven orders, 21 families, 35 genera, and 44 species. Of these species, only eight were present throughout the year, while the rest of the species were present during one, two, or three seasons. These results were compared with studies carried out in other coastal lagoons in Baja California Sur. In Ojo de Liebre Lagoon, a total of 59 species was identified using three methods (gillnet, trawl, and seine net) (Acevedo-Cervantes 1997). Of the species total, 29 were collected by trawling, which is lower than the number of species reported in this study. In El Coyote (adjacent to our study area), Ramirez De Aguilar Azpiroz (2001) recorded 43 species using three fishing methods (seine net, gillnet, and trawl); 15 species were collected by trawling. In the lagoon complex of Bahía Magdalena/Almejas, 75 species were captured using a trawl net taking monthly samples and a total of 197 trawls (Gutierrez-Sanchez, 1997). In Bahía Magdalena-Bahía Almejas, Torres Orozco et al. (1992) reported a total of 85 species, identified from the bycatch of shrimp. Ecologically, Magdalena Bay and Almejas Bay form an ecotone (Gutierrez-Sánchez, 1997); therefore, the number of species recorded was higher than that recorded at San Ignacio Lagoon. A total of 55 species were captured with trawl nets in Concepción Bay (Rodriguez-Romero et al., 1998). In this study, 44 trawls were conducted over four seasons (11 trawls per season); therefore, San Ignacio Lagoon can be considered as an area with high species richness, taking into account that the sampling effort was lower than in other studies.

In this study, we recorded a total numerical abundance of 1,361 individuals. This number was compared to other studies carried out in other areas of Baja California Sur. In Ojo de Liebre Lagoon, an abundance of 8,412 individuals was recorded, of which 611 were collected with a trawl net (Acevedo-Cervantes, 1997), lower than the total reported for San Ignacio Lagoon. A total abundance of 871 organisms was obtained in Tortugas Bay using a gillnet (Moreno-Sánchez, 2002). A study in El Coyote salt marsh reported 2,648 organisms, 91 of which were caught using a trawl net (Ramirez De Aguilar Azpiroz, 2001). A total of 854 organisms were captured in San Ignacio Lagoon using a gillnet (Cruz-Escalona, 1998). In the Magdalena Bay/Almejas Bay lagoon complex, Gutierrez-Sánchez (1997) reported 6,510 individuals due to a larger number of throws using a trawl net. Romero-Rodríguez et al. (1998) captured 1,336 organisms in Bahía Concepción, B.C.S. using a trawl net. Comparing these results with those obtained in the present study, we observed a certain similarity, and therefore infer that this lagoon maintains an abundance of organisms. In addition to its species richness, we can also say that it has not yet been altered or that exploitation of these resources has not been intense.

The total seasonal abundance in San Ignacio Lagoon showed a certain variation, with the highest catches in the fall and the lowest in the spring. This variation could be explained in terms of environmental changes (temperature, salinity, tides, etc.), by the use of different available resources in the lagoon, or by one of a number of other factors (reproduction, feeding, breeding, etc.). It is important to note that this abundance could also have been influenced by El Niño in 1998, because 21 species that have tropical origins were reported, which is uncommon in a temperate lagoon. There was an unusually high abundance of these typically tropical species, including *Bairdiella icistia*, *Cetengraulis mysticetus*, *Exerpes asper*, *Sphoeroides lispus*, *Urobatis maculatus*, *Cyclopsetta panamensis*, *Umbrina xanti*, and *Microlepidotus inornatus*. 14 species with temperate origins were also reported at a lower abundance than what would normally be expected. Analysis of the presence and abundance of these species, in addition to considering that this gap is within the province of California (Brigs, 1974), we can infer that temperature increase was an influential factor.

When calculating the density of catches by numerical abundance in studies carried out at three coastal lagoons, we detected a clear differentiation from south to north on the western coast of B.C.S. In the Magdalena Bay/Almejas Bay lagoon complex, Gutierrez-Sánchez (1997) calculated a density of 79.2 ind/ha; in this study, we reported a density of 39.3 ind/ha, and in Ojo de Liebre Lagoon, Acevedo-Cervantes (1997) calculated a density of 25.2 ind/ha. This latitudinal difference may be due to Magdalena Bay being part of a transition zone and having more environmental variety and differing depths (Gutierrez-Sánchez, 1997). This means that with decreasing latitude, there is higher diversity and lower abundance.

Relative abundance values indicate which species provide the highest proportion of individuals to the system. Of the 44 reported species in this study, the 21 having the highest relative abundance made up 91.41%. Nine species had relative abundance values over 4%, jointly making up 71.79%; other species had a relative abundance under 3%. These results were compared to those obtained by Gutierrez-Sánchez (1997) for the Magdalena Bay/Almejas Bay lagoon complex, where of the 71 identified species, five had abundances over 4% (*Etropus crossotus*, *Paralabrax maculatofasciatus*, *Eucinostomus dowii*, *Urobatis maculatus* and *Arius platypogon*). The study carried out by Acevedo-Cervantes (1997) in Ojo de Liebre Lagoon reported that of the 29 species collected with a trawl net, six represented 80.8%: *Sphoeroides* sp., *Paralabrax maculatofasciatus*, *Urobatis halleri*, *Urobatis*

maculatus, *Hypsopsetta guttulata*, and *Sphoeroides annulatus*. In El Coyote salt marsh, Ramirez De Aguilar Azpiroz (2001) identified 15 species that were captured with a trawl net, six of which were primarily abundant (*Sphoeroides annulatus*, *Sphoeroides lispus*, *Urobatis maculatus*, *Paralabrax maculatofasciatus*, *Eucinostomus dowii*, and *Bairdiella icistia*). In Concepción Bay (Gulf of California), Rodriguez-Romero (1992) reported 55 species, four of which had abundances over 4% (*Paralabrax maculatofasciatus*, *Etropus crossotus*, *Urobatis halleri*, and *Sphoeroides lispus*). From this comparison, we infer that relative abundance showed little variation for similar species caught in the four coastal lagoons. The variation may be due to environmental conditions at each site, to physiographic characteristics, and to sampling effort.

The total fish biomass in San Ignacio Lagoon was 67,118 g, compared with the Magdalena Bay/Almejas Bay lagoon complex, where the total was 416,693 g (Gutierrez-Sanchez, 1997), and Ojo de Liebre Lagoon, where total biomass was 86,147 g (Acevedo-Cervantes, 1997). This difference may be due to sampling effort, seasonality, abundance, fish distribution, and the size of organisms (Lagler et al., 1984; Torres Orozco et al., 1992). Of the 44 species reported, 12 contributed 89.66%; and seven of those 12 made up 76.15% (*Sphoeroides annulatus*, *Urobatis halleri*, *Sphoeroides lispus*, *Paralabrax maculatofasciatus*, *Paralabrax auroguttatus*, *Urobatis maculatus*, and *Paralichthys californicus*). In the Magdalena Bay/Almejas Bay lagoon complex, seven species were reported with values over 4% (*Paralabrax maculatofasciatus*, *Sphoeroides annulatus*, *Urobatis halleri*, *Urobatis maculatus*, *Eucinostomus dowii*, *Dasyatis brevis*, and *Etropus crossotus*) (Gutierrez-Sanchez, 1997). In Ojo de Liebre Lagoon, Acevedo-Cervantes (1997) reported a biomass of 86.6%; the species that contributed to this biomass were *Sphoeroides* sp., *Paralabrax maculatofasciatus*, *Urobatis halleri*, *Urobatis maculatus*, *Hypsopsetta guttulata*, and *Sphoeroides annulatus*. Similar species contributed to the high biomass in the three lagoons.

Twenty-five species were dominant in all four seasons. 12 species were recorded at over 4%, according to the BVI. In the Magdalena Bay/Almejas Bay lagoon complex, Gutierrez-Sanchez (1997) reported 12 dominant species: *Arius platypogon*, *Sphoeroides annulatus*, *Paralichthys californicus*, *Diplobatis ommata*, *Pleuronichthys ritteri*, *Urobatis halleri*, and *Achirus mazatlanus*. In Ojo de Liebre Lagoon, north of San Ignacio Lagoon, Acevedo-Cervantes (1997) reported that the dominant species were *P. maculatofasciatus*, *Urobatis halleri*, and *Urobatis maculatus*. There was no overlap in the recorded species, except for *P. maculatofasciatus* and *S. annulatus*, which are the most dominant species, and remain within these lagoon systems (resident species) for most of the year. This supports the hypothesis of Loya and Escofet (1990), who stated that constancy in space and time are the main factors that influence species dominance.

Diversity calculated within San Ignacio Lagoon showed a marked seasonality, which was influenced by annual composition, distribution, and abundance of species. In general, some species typify seasons and location, while other species show higher distribution throughout the ecosystem. Diversity values obtained here may be due to fishing gear selectivity, consistent with results reported by Alvarez-Rubio et al. (1986) for a lagoon system in Nayarit. Changes in diversity may also be a function of season, location, fishing gear, and intrinsic characteristics of the community. Comparing the analysis of this ecological attribute with the one reported by Gutierrez-Sanchez (1997), who recorded values fluctuating between 1.9 bits/ind and 3.6 bits/ind for the Magdalena Bay/Almejas Bay area, we can observe that these diversity values are higher than those obtained at San Ignacio Lagoon (between 0.56 and 2.28

bits/ind), for abundance as well as biomass. This was probably due to ecological, oceanographic, and physiographic characteristics of the system. In Ojo de Liebre Lagoon, Acevedo-Cervantes (1997) reported higher values (1.5 to 2.5 bits/ind) due to the influence of seagrass. In southern California, Horn and Allen (1985) reported a numeric diversity of 1.2 bits/ind, and a biomass of 2.15 bits/ind. These results are similar to the results reported in the present study. According to both reports, we observe that ecosystems located north of San Ignacio lagoon have relatively similar diversities, which are consistent with its zoogeographic location.

Based on its geographic location, San Ignacio lagoon is classified as a temperate lagoon, according to Hubbs (1960), who reported that for the western coast of the Baja California peninsula, 44.7% of species had temperate affinity, 22% had tropical affinity, and 33% of species were endemic to the Californian province. According to the zoogeographic affinity recorded in this study, the Panamic Province accounted for 36% of the species, the Eastern Pacific Province for 18%, the Mexican Province for 11%, the Cortez Province (one species, *Exerpes asper*, is listed as amphipeninsular, since it is found on both sides of the Baja California Peninsula) for 2% (accounting for 67% for tropical affinity fish), and the San Diego Province, where San Ignacio Lagoon is located, for 32%. This difference could be mainly due to the El Niño event that occurred during the spring, summer, and fall of 1998. Acevedo-Cervantes (1997) reported that in Ojo de Liebre Lagoon, the percentage of fish with temperate affinity was similar to that reported by Hubbs (1960). In the Magdalena Bay/Almejas Bay lagoon complex, Gutierrez-Sanchez (1997) established differences between species that were close to the bay and had temperate affinity, and those with tropical affinity at the end of the lagoon complex. The percentage of species in the Magdalena Bay lagoon complex that belonged to each province was 25% Californian, 8% Mexican, 1% endemic from the Gulf of California, 49% Panamic, 12% Eastern Pacific, 1% Circum-pacific, and 5% Circumtropical. Dominguez et al. (2000) reported changes in coastal fish community structure during El Niño-La Niña events in Navidad Bay, off the coast of the state of Jalisco, and found that 68% of the species caught had an affinity for the Panamic Province, 21% had an affinity for the Eastern Pacific, 8% were circumtropical and 1.6% had Indo-Pacific affinity. In a study of five coastal lagoons in the state of Baja California Sur, Galvan Magaña et al. (2000) reported that off the western coast of Baja California Sur, the species with Panamic affinity increased with decreasing latitude, while those of Californian affinity decreased with decreasing latitude.

Comparing these results with those of the present study, there were differences in the percentages. This could mean that oceanographic events influenced species distribution in this lagoon, which has more tropical affinity species than temperate affinity species.

The spatio-temporal distributions of temperature and salinity at San Ignacio Lagoon showed that values increased from the mouth of the lagoon towards the inside of the lagoon. There was also a pattern in dissolved oxygen distribution, with decreasing values from the mouth towards the inside of the lagoon, and no defined pattern in the middle. Fish diversity was within acceptable values when compared to similar study areas to the north and south of San Ignacio Lagoon, considering that this study was present during the El Niño event. According to the BVI, eight species were considered resident and were found throughout all four seasons. Temperature also played an important role in species distribution. Geographically, San Ignacio Lagoon is located in a temperate zone; therefore, species with more affinity to that type of region would be expected. However, there were more species

with tropical and subtropical affinity, accounting for 67% of species, while San Diego Province was represented by 32% of species. We therefore conclude that this distribution was influenced by the El Niño event.

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Chapter 10

LONG-TERM EVALUATION OF FISH ASSEMBLAGES AND THEIR ANTHROPOGENIC IMPACT IN THREE ARID MANGROVE SYSTEMS OF THE GULF OF CALIFORNIA

Francisco Javier López Rasgado^{1,2,*}

¹Centro de Investigaciones Biológicas del Noroeste (CIBNOR), La Paz,
Baja California Sur, Mexico

²Centro del Cambio Global y la Sustentabilidad en el Sureste (CCGSS),
Centenario del Instituto Juárez 128, Villahermosa, Tabasco, Mexico

ABSTRACT

Mangrove systems are of great importance worldwide for their high primary and secondary productivity because they sustain fish abundance and diversity. The aim of this chapter was to compare the current structure and composition (2009-2010) of the fish community in three mangrove systems with different degrees of anthropogenic influence in the Bay of La Paz, Baja California Sur, with estimates made 30 years ago (1980-1981). Ecological attributes were estimated (number and weight density, species richness, Shannon-Wiener diversity, Pielou evenness), average taxonomic distinctness (ATD) and the variation in taxonomic distinctness (VarTD) from data collected monthly during each period. A total of 39,123 fish were recorded in 1980-1981 and 16,740 fish in 2009-2010. During both periods, the species that dominated the density in the three systems were *Eucinostomus dowii*, *Diapterus brevirostris*, and *Mugil curema*. The most dramatic changes in the fish structure community was between the periods in the system that has had more human intervention. In contrast, no significant differences between periods were found in the less impacted system, suggesting on the one hand that the fish community is a good indicator of healthy systems, and on the other that local effects of human intervention outweigh the climatic influence regionally.

Keywords: Diversity, Bay of La Paz, fish structure, mangrove

* E-mail: francisco.lopez@ccgss.org.

INTRODUCTION

Mangrove communities play a major role in the interface (ecotone) between terrestrial and marine systems, where a large structural and functional complexity is recognized. They provide various environmental services, such as coastal defense against erosion and hurricanes, as well as nursery areas for commercial fish and carbon fixation (Bouillon et al. 2008a, Feller et al. 2010). In Mexico, mangrove systems cover a total area of 770,057 ha (Conabio 2009). Its distribution reaches the Pacific coastal arid zone at latitude 29° 19' N in the Gulf of California (GC, Rzedowski 1983).

The Gulf of California (GC) is located along Mexico's western coast. It is separated from the Pacific Ocean by the Baja California Peninsula and is ca. 1100 km long and 50–250 km wide. The GC is the northern geographic limit for mangrove species found in the tropical eastern Pacific (Rzedowski 1983). Inside the GC, mangrove habitats have a discontinuous distribution and are separated by up to hundreds of kilometers (Whitmore et al. 2005). They are found from Los Cabos at the southernmost peninsula to Los Angeles Bay on the peninsular coast in the central Gulf and along the northeastern continental coastline, as well as on some islands, covering a total area of 24,327 ha in Baja California Sur (Conabio 2009). Mangrove systems of the Bay of La Paz, Baja California Sur, have a tendency to degrade due to environmental changes. It is assumed that the accelerated and disordered development of the city of La Paz and the surrounding metropolitan area has produced effects on coastal vegetation, particularly on mangrove communities, causing fragmentation and reduction (Mendoza et al. 1984; De la Cruz 1996). In this regard, Mendoza et al. (1984) reported a loss of 44 ha of mangrove in the Ensenada de La Paz during the period of 1973–1981, assuming that human impact was the main factor of the degradation.

Mangrove systems are recognized as important sources of productivity for coastal fisheries (Manson et al. 2005, Nagelkerken 2007). A mangrove system is a unique habitat that provides a structured environment for fishes that leads to reduced predation pressure (Laegdsgaard and Johnson 2001) and favorable foraging conditions (Sheridan and Hays 2003, Nagelkerken and van der Velde 2004). In this regard, it is recognized that in certain regions, the degradation of these ecosystems is a cause of declining fish populations at local and regional levels (Chambers 1992). For example, studies in the Bahamas showed that mangrove systems fragmented by road construction exhibited a decline in habitat quality and showed changes in the fish community structure compared to nearby unfragmented mangrove systems (Layman et al. 2004, Taylor et al. 2007, Rypel and Layman 2008). This chapter is focused on determining the current status of the fish community structure in three mangroves with different degree of anthropogenic habitat influence that have been modified over time in La Paz Bay, México, through comparison with characterizations made in the past (1980-81), and the establishment of a base line for future comparisons. Thus this chapter evaluates whether these systems show evidence of long-term changes in fish community structure. This is carried out specifically by comparing the data collected in the early 1980s with a survey performed almost 30 yrs later (2009–2010) using virtually identical methods. The information generated is relevant for the decision making process of land use and environmental health assessment.

BACKGROUND

Long-term data sets of fish community structure are limited mostly to cases for which fishery-dependent data are available and are mostly generated by the need to assess the effects of the fishery on commercial species (e.g., Greenstreet and Hall 1996, Potter et al. 2001, Bradshaw et al. 2002, O'Connell et al. 2004, Tian et al. 2006). For example, Genner et al. (2004) examined changes in the fish assemblages in marine and estuarine systems in the United Kingdom and observed drastic changes in community composition. Staglicic et al. (2011) evaluated changes in the fish community structure along the eastern Adriatic Sea over a 16-yr period and found that littoral assemblages exhibited drastic temporal changes in structural communities due to overfishing. McHugh et al. (2011) studied long-term changes in fish assemblages of the western English Channel throughout the 20th century and found significant differences in species composition assemblages. The greatest changes in distribution have included increases in the number of larger species with slow-maturation due to the influence of both commercial fishing and regional climate change.

Studies of fish community assemblages associated with mangrove ecosystems in the Mexican Pacific are scarce, although the predominance of juvenile and sub-adult fishes suggest that mangrove systems play an important ecological role in recruitment and in supporting fishery resources on the adjacent inner continental shelf (e.g., Warburton 1978, D'Croz and Kwiecinski 1980, Álvarez-Rubio et al. 1986, Amezcuá-Linares et al. 1987, Flores-Verdugo et al. 1990). Aburto-Oropeza et al. (2008) reported a positive relationship between coastal fish landings and size of nearby mangrove systems in the Gulf of California. Other authors have linked various measurements of fish community structure with mangrove ecosystem health in the GC, including salinity fluctuations in lagoonal systems (Flores-Verdugo et al. 1990, Mendoza et al. 2009), changes in mangrove coverage, and proximity to urban areas (Balart et al. 1997), as well as coastal pollution (Trejo and Mayoral 1984) and the introduction of exotic species (Balart et al. 2009, Palacios-Salgado et al. 2011). Nevertheless, a serious limitation to the assessment of the effect of mangrove habitat loss and modification on fish community structure is the lack of historical data.

Particularly for the La Paz Bay, the fish fauna reported for La Paz Bay includes 522 species (Abitia-Cárdenas et al. 1994, Balart et al. 1995), and 31% of them are known to be inhabitants of mangrove systems during at least part of their life cycle (González-Acosta et al. 1999, Whitmore et al. 2005) for feeding, growth, reproduction, protection and nursery grounds (Chávez 1985a, Castro-Aguirre et al. 1999). It is proposed that these fish assemblages of mangrove systems are actively involved in the regulation of energy balance in the region, importing or exporting energy to neighboring ecosystems (Yáñez-Arancibia and Nugent 1977, Castro-Aguirre et al. 1994).

There are few studies about the fish fauna associated with mangroves in the region. For example, Maeda-Martínez (1981) evaluated fish species associated to three mangroves of Bahía de La Paz, Balandra, Zacatecas and Enfermería, describing its composition, abundance and diversity, in addition to food categorization which focused on primary consumers. González-Acosta et al. (2005) evaluated seasonal changes in the structure of fish assemblages of mangrove the El Conchalito, finding a seasonal alternation in abundance and biomass of the dominant species. De la Cruz-Sosa (2004) evaluated the fish assemblages of four mangroves in the Bay of La Paz, he found an alternation in abundance and diversity, mainly

due to the temperature, and the dominant members of species were of the families Mugilidae, Gerreidae and Lutjanidae.

MATERIALS AND METHODS

Study Area

La Paz Bay is the largest protected coastal system in the eastern side of the Baja California Peninsula, covering 2000 km² (Chávez 1985b). The area is considered arid and has rainfall that is limited to sporadic winter storms or hurricanes. The bay hosts about 20 discrete mangrove habitats ranging in size from 0.64 to 52 ha (González-Zamorano 2002; Fig. 1). This chapter focused on the three mangrove habitats for which historical information on fish communities exists. They include red, black, and White mangrove species (*Rhizophora mangle*, *Avicennia germinans* and *Laguncularia racemosa*) that are present at the three sites, which are separated by tens of kilometers (Figure 1).

The mangrove forest at Balandra Lagoon (24° 19' 5" N, 110° 19' 22" W; Figure 1B) is a protected cove covering 52 ha of which 22 ha. are occupied by mangroves. The area covered by the mangrove habitat is channel shaped, with a length of 990 m and a mean width of 324 m. The connection between the cove and the bay occurs through a 180 m wide mouth that has a channel with mean depth of 1.1 m. This is considered to be the most pristine mangrove system in the region because it has suffered little anthropogenic habitat modification (Mendoza-Salgado et al. 2011). The mangrove system of Zacatecas (24° 10' 27" N, 110° 26' 6"W; Figure 1C) consists of a narrow and long winding channel system of 57 ha, of which 22 ha are covered with mangroves. The connection with the bay occurs through a mouth 36 m wide and 2.5 m deep. This system currently exhibits relatively limited impacts from human activities, but there is urban development close by and its future is uncertain. The mangrove system of Enfermería (24° 13' 48" N, 110° 18' 23" W; Figure 1D) has a micro-basin shape covering a total area of 8 ha of which 3 ha are covered by mangroves (Padilla-Arredondo et al. 1984). In the early 1980s, it was connected to La Paz Bay through a 6 m wide mouth (Mendoza-Salgado 1983). Enfermería is one of the best-studied mangrove systems in the bay, mainly because of the well documented damage caused by human activities. The mangrove forest is located 6 km north of the city of La Paz, and was first modified during the construction of the La Paz-Pichilingue freeway that was finished in 1964 (Cervantes del Río 1967). Between 1994 and 2000, a shrimp-farming project was developed inside the mangrove system, affecting neighboring areas and modifying the natural exchange of water with the rest of the bay.

In 2005, an additional section of freeway was built bordering the eastern side that further impeded the scarce inflow of fresh water during the limited rainy season. These modifications have caused changes in the dynamics of water exchange, which has led to increased sediment accumulation.

The present-day conditions are not conducive to the establishment of new mangrove seedlings, this has contributed to the decay in the long-term condition of the mangrove forest and additional losses of aquatic habitat.

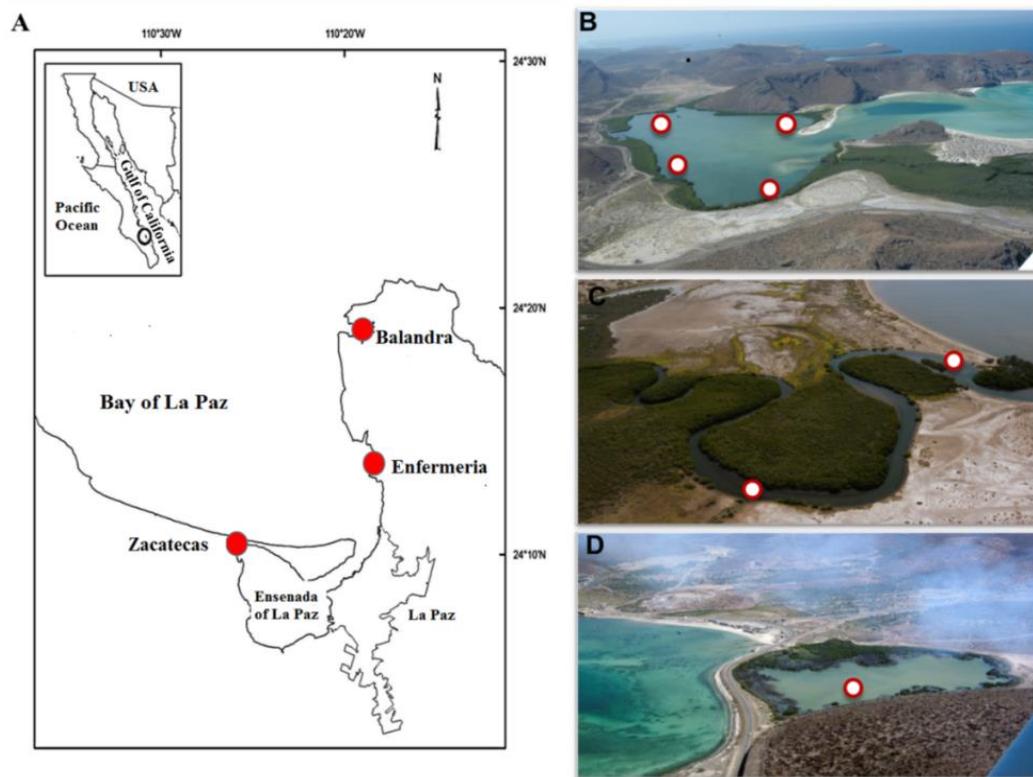


Figure 1. (A) Geographic location and aerial photographs of the three mangrove study systems in La Paz Bay, Gulf of California Mexico, including the specific locations in which sampling was conducted in 1980–1981 and 2009–2010. (B) Balandra with sampling sites, (C) Zacatecas with sampling sites. (D) Enfermeria, sampling site, and the major construction impacting the system.

Sampling Methods

Historical Samples (1980-1981)

Maeda-Martínez (1981) sampled the fish community of the three mangrove systems on a monthly basis from March 1980 to February 1981 using a beach seine (35 m length, 2.15 m height, and 1 cm mesh size). A single tow was conducted at specific sites within each system along the edge of the prop roots ($n = 4$, 2, and 1 for Balandra, Zacatecas, and Enfermeria, respectively; Figure 1). Fish samples collected by Maeda-Martínez (1981) were stored in vials with 10% formaldehyde and archived in the Ichthyology Laboratory of the Biology Department of the Universidad Autónoma de Nuevo León, Nuevo León, Mexico (Dr. María de Lourdes Lozano-Vilano, curator). For the present study, we revisited that collection and verified the identification of all individuals using recent identification keys (Allen and Robertson 1994, Fischer et al. 1995, Thomson et al. 2000). The standard length (SL, mm) and weight (g) of each individual was recorded. The 1980-1981 study period will be called hereafter as 1980.

Present-Day Samples (2009-2010)

To allow for a robust comparison of the fish community structure through time, we repeated the same sampling strategy used by Maeda-Martínez (1981) during a 1-yr period (July 2009–June 2010). The fishing gear consisted of a similar, but shorter, beach seine (20 m length, 2 m height, and 1 cm mesh size). Sampling was carried out at the same frequency and at the specific stations described by Maeda-Martínez (1981). Each captured individual was identified, weighed and measured (SL). Specimens were incorporated into the Ichthyology Collection of CIBNOR. The 2009-2010 study period will be referred to hereafter as 2010.

Data Analysis

Comparing the structure of the fish community in 1980 and 2010 was performed with data collected monthly in each of the three mangrove systems throughout the year. Richness (S) was computed for each period as the total number of fish species captured each month in each mangrove system throughout the entire year. Due to differences between the historical and present-day sampling in the area covered by each tow due to the seine length (97.5 m² in 1980 and 65 m² in 2010), the densities and biomass were estimated for each species and location by standardizing per unit area. Density is reported as the ind m⁻² and biomass as g m⁻² (Varnell and Havens 1995).

We estimated indices of diversity (Shannon-Wiener, H') and evenness (Pielou, J' ; Pielou 1975, Brower and Zar 1977, Ludwig and Reynolds 1988). These indices are considered to be rather robust due to the constant sampling effort. Richness, diversity, evenness, density, and biomass were compared between periods for each location using individual Student's t-test.

We also calculated the average taxonomic distinctness (ATD) and the variation in taxonomic distinctness (VarTD) for each community for each sampling period (Clarke and Warwick 2001). ATD is the mean number of steps up a hierarchy that are necessary to reach a taxonomic rank common to two species, and is computed across all possible pairs of species in an assemblage. Equal step lengths are assumed, with branch lengths standardized so that the shortest and longest paths in the tree are set from 0 to 100. VarTD is simply the variance of these pairwise path lengths and reflects the unevenness of the taxonomic tree. Additionally, both can be used to explore anthropogenic impacts and compared with classic species richness indices as a measure of biodiversity (Warwick and Clarke 1995, 1998).

Species densities data were square root-transformed to down-weight the influence of dominant species. These data were used to recalculate the similarity among samples using the Bray-Curtis coefficient. A non-metric multidimensional scaling (MDS) representation was used to detect patterns of similarity between sampling units. The measurement of this distance is called stress, and it is considered that values of up to 20% show a proper goodness of fit. One-way analysis of similarities (ANOSIM) was performed for each study area to identify differences in community structure between periods; these tests were run with the software PRIMER ver. 6.0 (Clarke 1993, Clarke and Warwick 2001).

We classified the fish species collected at each mangrove and period based on their frequency of occurrence and abundance using the Olmstead-Tukey classification (Sokal and Rohlf 1997). This analysis allows for an ecological and quantitative classification of species based on four species categories: (1) dominant species have values of relative abundance and relative frequency of occurrence higher than the arithmetic mean of both parameters; (2)

frequent species are those whose relative frequency of occurrence is higher than the arithmetic mean and that have a lower than average abundance; (3) occasional species that have a relative abundance higher than the corresponding arithmetic mean and lower than average frequency of occurrence; and (4) rare species, where values of relative abundance and relative frequency of occurrence are lower than their respective arithmetic means

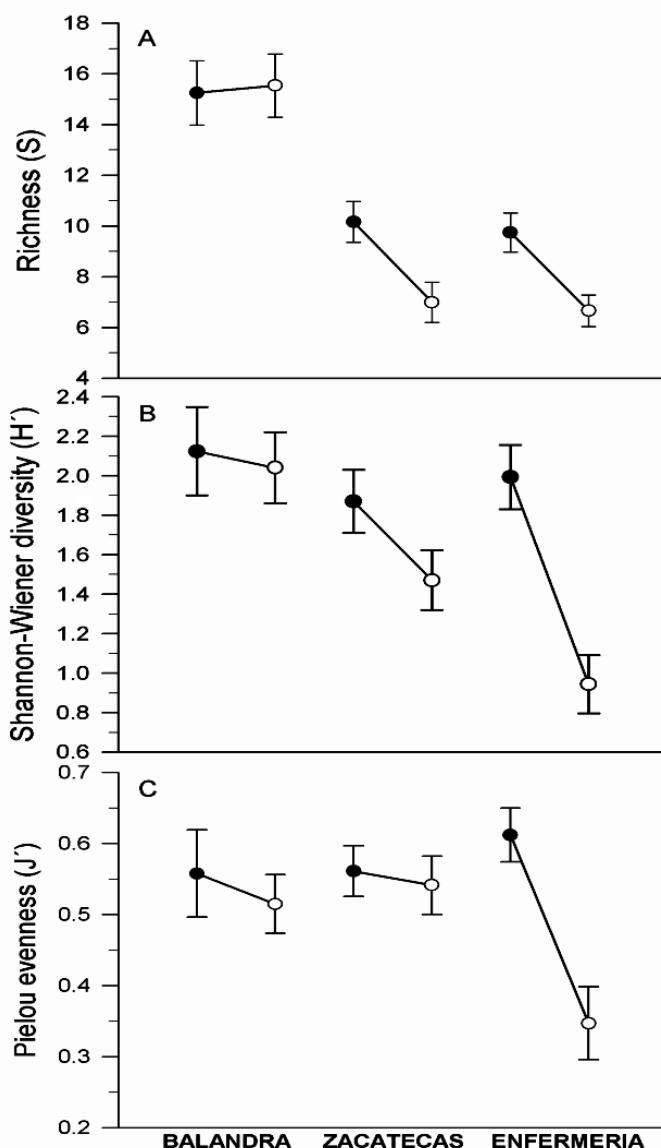


Figure 2. Comparison of A) richness B) Shannon-Wiener diversity and C) Pielou evenness for three mangroves forest estuaries sampled in La Paz Bay, southwestern Gulf of California, Mexico, 30 yrs apart. Black points = 1980–1981, white points = 2009–2010. Mean (SE) of data collected monthly are presented.

RESULTS

We recorded a total of 39,123 individuals for the 1980 historical collection and 16,740 for the field sampling carried out during 2010. Major differences in fish densities between the historical data set and the present-day data were evident in a few formerly abundant species. The species that exhibited high mean monthly densities for 1980 were: *Eucinostomus dowii*, *Anchoa ischana*, and *Mugil curema* for Balandra; *Pomadasys branickii*, *E. dowii* and *Diapterus brevirostris* for Zacatecas; and *D. brevirostris*, and *E. dowii* and *Gerres simillimus* for Enfermeria (Table 1). For 2010 period, the species with higher mean densities were *E. dowii*, *G. simillimus* and *Eucinostomus sp.*, for Balandra; *E. dowii*, *D. brevirostris* and *E. currani* for Zacatecas; and *D. brevirostris*, *M. curema* and *Lile stolifera* for Enfermeria.

Regarding the biomass, species that contributed most in 1980 were *Hyporhamphus naos*, *M. curema* and *E. dowii* to Balandra, *Pomadasys branickii*, *M. curema* and *E. dowii* to Zacatecas and *D. brevirostris*, *G. simillimus* and *E. dowii* for Enfermeria. The species that had higher mean biomass in 2010 were *M. curema*, *E. dowii* and *D. brevirostris* for Balandra, *Diodon hystrix*, *E. dowii* and *D. brevirostris* for Zacatecas; and *D. brevirostris*, *Centropomus viridis* and *M. curema* for Enfermeria. In general, the species with higher densities and biomass were from the mangrove dependent families, Gerreidae and Mugilidae.

Richness was consistently higher in Balandra during both sampling periods compared to the other mangrove systems. Zacatecas and Enfermeria exhibited significantly lower values in 2010 than in the historical collection of 1980 ($t = 2.81$, $p = 0.01$, $t = 3.12$, $p < 0.01$, for Zacatecas and Enfermeria, respectively, Figure 2A).

Diversity index Shannon-Wiener showed no statistically significant differences in Balandra and Zacatecas in both periods ($t = 0.29$, $p = 0.77$, $t = 1.81$, $p = 0.08$, for Balandra and Zacatecas, respectively), in contrast, Enfermeria showed significantly lower values in the period 2010 than 1980 ($t = 4.77$, $p < 0.01$, Figure 2B). Pielou evenness values showed that there were no significant differences between periods in Balandra ($t = 0.59$, $p = 0.56$) nor in Zacatecas ($t = 0.36$, $p = 0.72$, Figure 2C), except Enfermeria that showed the most dramatic changes between periods, with a significant reduction in the period 2010 ($t = 4.16$, $p < 0.01$). Enfermeria exhibited the most drastic change in its community structure over time with significant changes evident in species richness, evenness, and diversity. Fish density did not differ significantly between sampling periods in each of the mangrove systems ($t = 1.38$, $P = 0.181$; $t = 1.064$, $P = 0.298$; and $t = -0.065$, $P = 0.947$; for Balandra, Zacatecas, and Enfermeria, respectively; Figure 4A). Biomass did change significantly between periods for Balandra ($t = 2.550$, $P = 0.017$) and Zacatecas ($t = 2.207$, $P = 0.038$), but not for Enfermeria ($t = -1.402$, $P = 0.175$). The highest mean monthly density and biomass were found at Enfermeria in the present-day collection [5.68 (SE 1.21) ind m^{-2} and 20.70 (SE 4.75) g m^{-2}], while Balandra and Zacatecas displayed lower mean density and biomass in the present-day collection compared to the historical data. The biomass estimates for the historical data set were similar among the three mangrove systems (ca. 10–12 g m^{-2}). The expected average value of ATD, calculated by pooling all monthly samples from all sites, was close to 60 (Figure 4A). ATD values that fall outside of the 95% samples (monthly ATD s) fell within the 95% probability limits, except for some samples from the historical collection for Zacatecas (January) and Enfermeria (June, July, September, December, and January), and a few from 2009 to 2010 (October for Zacatecas and June and January for Enfermeria). Interestingly, for Enfermeria, in five out of 12 monthly samples there was a greater taxonomic

distinctness in the historical collection than was found in the majority of the collections from all sites and both periods. In contrast to the analysis of ATD, VarTD values are greater than the average expected value and are closer to the limit of 95% for both periods (Figure 4B). The MDS analysis showed that the fish community collected in Balandra in 2010 was distinct from the same system's historical data set as well as the other mangrove systems during both study periods (Figure 5). The value of the stress of management was 0.19, indicating an adequate goodness of fit. However, analysis of similarity (ANOSIM) indicated significant differences in the fish community structure between periods for Balandra ($G = 0.41$, $p < 0.01$) and Enfermería ($G = 0.47$, $p < 0.01$), but not for Zacatecas ($G = 0.10$, $p = 0.03$).

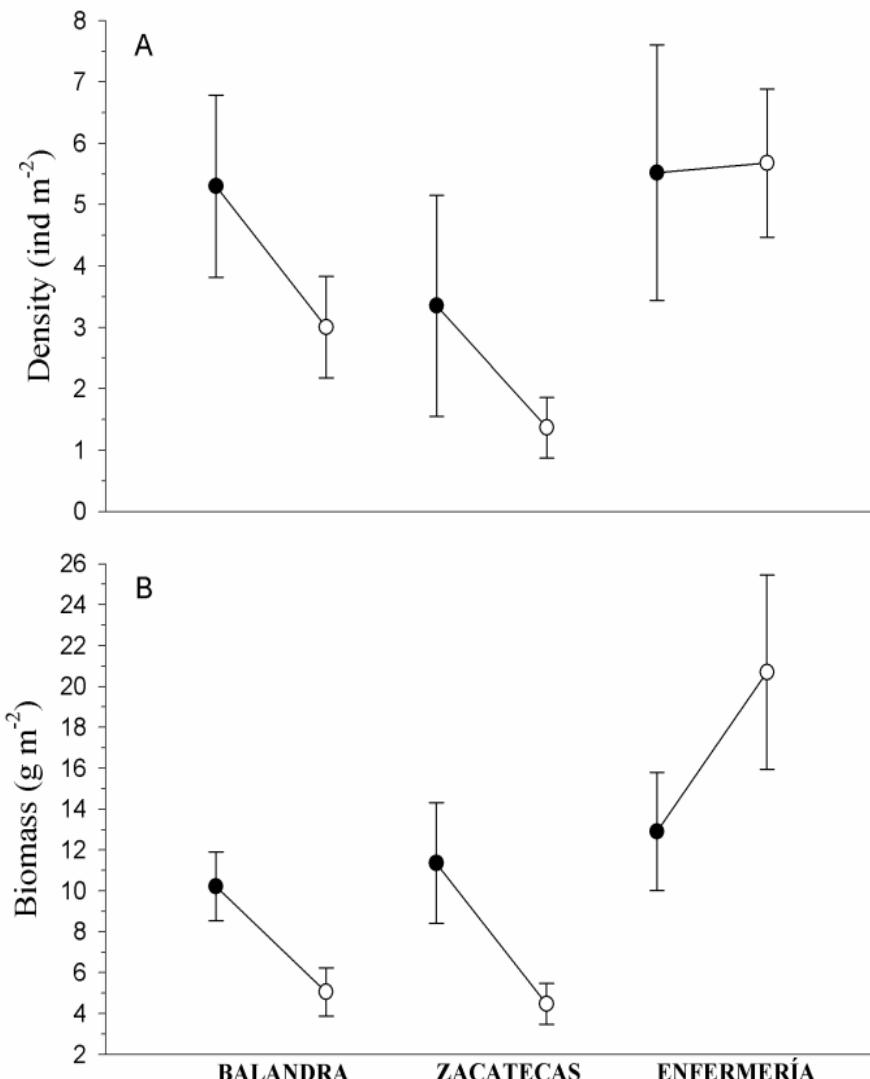


Figure 3. Comparison A) density (ind m^{-2}) and B) biomass (g m^{-2}) in three mangrove forest estuaries in La Paz Bay, Mexico. Closed symbols are 1980–1981 and open symbols are 2009–2010. Mean (SE) are presented.

Table 1. Mean density (D, ind m⁻²), Biomass (B, g m⁻²) and standard length (SL, mm), for fish species collected in 1980–1981 and 2009–2010 in three mangroves systems in La Paz Bay, southwestern Gulf of California, Mexico. Bold numbers correspond to species with higher density or biomass

Family/specie	1980-1981									2009-2010								
	Balandra			Zacatecas			Enfermeria			Balandra			Zacatecas			Enfermeria		
	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL
Albulidae																		
<i>Albula esuncula</i>	0.02	0.07	58	<0.01	<0.01	74				0.01	0.05	72						
Carangidae																		
<i>Caranx caninus</i>										<0.01	0.01	61						
<i>Caranx sexfasciatus</i>				<0.01	0.03	101												
<i>Oligoplites altus</i>													<0.01	<0.01	34			
<i>Oligoplites fulgens</i>										<0.01	0.01	49						
<i>Oligoplites saurus</i>	<0.01	<0.01	77	<0.01	<0.01	99	<0.01	<0.01	27				<0.01	0.01	57			
Centropomidae																		
<i>Centropomus nigrescens</i>										<0.01	<0.01	37						
<i>Centropomus viridis</i>										<0.01	0.01	61	<0.01	<0.01	74	0.02	0.84	133
Chanidae																		
<i>Chanos</i>										<0.01	0.05	83	<0.01	<0.01	54	0.01	0.04	79
Clupeidae																		
<i>Etrumeus teres</i>										<0.01	<0.01	37						
<i>Harengula thrissina</i>										<0.01	<0.01	49						
<i>Lile nigrofasciata</i>							0.01	0.02	61									
<i>Lile stolifera</i>							0.01	0.05	73	0.01	0.04	58				0.20	0.43	50
Family/specie	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL
<i>Opisthonema libertate</i>	0.12	0.44	55	<0.01	0.02	95												
<i>Opisthonema medirastre</i>										0.01	0.03	50						
Diodontidae																		
<i>Diodon holocanthus</i>	<0.01	0.14	118							<0.01	0.14	164						
<i>Diodon hystrix</i>													<0.01	1.30	205			

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	1980-1981									2009-2010								
	Balandra			Zacatecas			Enfermeria			Balandra			Zacatecas			Enfermeria		
Family/specie	D	B	SL	D	B	SL	D	B	SL									
Eleotridae																		
<i>Dormitator latifrons</i>	<0.01	<0.01	21				0.03	0.02	29				<0.01	<0.01	44			
<i>Erotelis armiger</i>																		
Elopidae																		
<i>Elops affinis</i>				<0.01	<0.01	64				<0.01	<0.01	57				0.00	0.06	79
Engraulidae																		
<i>Anchoa ischana</i>	1.14	1.13	53	0.05	0.14	64	<0.01	0.03	89	0.09	0.14	52				0.00	0.05	106
Engraulidae																		
<i>Anchoa nasus</i>		0.03								0.01	0.18	112						
<i>Anchoa</i> sp.										0.02	0.01	33	<0.01	<0.01	38			
Fistulariidae																		
<i>Fistularia corneta</i>										<0.01	0.01	291						
Gerreidae																		
<i>Diapterus brevirostris</i>	0.18	0.24	34	0.33	0.47	36	1.88	4.60	40	0.30	0.54	35	0.33	0.39	35	4.42	17.35	43
<i>Eucinostomus currani</i>	0.11	0.13	32	0.33	0.31	28	0.33	0.42	33	0.37	0.31	27	0.13	0.06	26	0.04	0.17	46
<i>Eucinostomus dowii</i>	2.35	1.51	27	0.72	1.53	41	0.89	1.51	39	0.61	0.52	25	0.67	1.15	33	0.05	0.04	30
<i>Eucinostomus entomelas</i>										0.02	0.25	78	0.03	0.39	77			
<i>Eucinostomus gracilis</i>										<0.01	<0.01	28	<0.01	<0.01	17			
<i>Eucinostomus</i> sp.										0.55	0.08	19						
<i>Eugerres axillaris</i>	<0.01	<0.01	56	0.01	0.08	62	0.02	0.21	77							0.03	0.23	63
<i>Eugerres lineatus</i>										<0.01	0.03	60						
<i>Gerres simillimus</i>	0.03	0.05	30	<0.01	0.02	68	1.67	2.45	30	0.55	0.19	25	0.01	0.01	44	0.16	0.19	32
Gobiidae																		
<i>Bathygobius ramosus</i>										<0.01	0.01	60						
<i>Ctenogobius sagittula</i>	0.03	0.04	43	0.07	0.16	57	0.03	0.09	56	0.05	0.05	38	0.01	0.01	58	0.03	0.10	61
<i>Gobiosoma chiquita</i>	0.02	0.01	27				0.01	0.00	24	0.01	0.01	28						
<i>Quietula y-cauda</i>	0.42	0.13	27	<0.01	<0.01	26	0.23	0.07	28									
Haemulidae																		
<i>Conodon serrifer</i>	<0.01	0.01	91	<0.01	0.01	77												

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Table 1. (Continued)

Family/specie	1980-1981												2009-2010													
	Balandra			Zacatecas			Enfermeria			Balandra			Zacatecas			Enfermeria										
	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL		
<i>Haemulon flaviguttatum</i>	0.03	0.45	80							<0.01	0.01	60														
<i>Haemulon maculicauda</i>	<0.01	0.01	100																							
<i>Haemulon scudderii</i>										<0.01	0.07	56														
<i>Haemulon sexfasciatum</i>										<0.01	<0.01	43	<0.01	0.01	42											
<i>Haemulon steindachneri</i>										<0.01	0.08	87														
<i>Haemulopsis axillaris</i>										0.02	0.11	62	<0.01	0.02	100	0.02	0.03	46								
<i>Haemulopsis leuciscus</i>													<0.01	<0.01	39											
<i>Microlepidotus inornatus</i>	<0.01	0.03	113																							
<i>Orthopristis chalceus</i>	0.01	0.01	29										<0.01	0.01	85											
<i>Orthopristis reddingi</i>	<0.01	<0.01	47																							
<i>Pomadasys branickii</i>	<0.01	<0.01	44	1.23	2.75	44				<0.01	<0.01	54	0.02	0.25	84											
Haemulidae																										
<i>Pomadasys macracanthus</i>													<0.01	0.04	104	<0.01	0.07	97								
<i>Pomadasys panamensis</i>				0.02	0.05	40																				
<i>Xenichthys xanti</i>				<0.01	<0.01	70																				
Hemiramphidae																										
<i>Hyporhamphus naos</i>	0.23	2.67	117	0.03	1.05	169	0.08	1.03	109	0.01	0.21	117	<0.01	0.18	185											
<i>Hyporhamphus rosae</i>										<0.01	<0.01	120														
Kyphosidae																										
<i>Kyphosus analogus</i>	<0.01	0.02	104																							
Labrisomidae																										
<i>Exernes asper</i>	<0.01	<0.01	30				<0.01	<0.01	29	<0.01	<0.01	34														
<i>Paraclinus mexicanus</i>										<0.01	<0.01	34														
Lutjanidae																										
<i>Hoplopagrus guentherii</i>	<0.01	0.01	67																							
<i>Lutjanus aratus</i>	<0.01	0.02	94	<0.01	0.10	108				<0.01	0.06	125														
<i>Lutjanus argentiventralis</i>	0.01	0.02	36	0.02	0.27	67	0.03	0.31	62	0.02	0.09	52	<0.01	0.03	76	<0.01	0.07	115								
<i>Lutjanus colorado</i>	<0.01	0.02	82	<0.01	0.00	28	0.01	0.25	74																	

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	1980-1981									2009-2010								
	Balandra			Zacatecas			Enfermeria			Balandra			Zacatecas			Enfermeria		
Family/specie	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL
<i>Lutjanus novemfasciatus</i>	<0.01	0.22	81	0.01	1.04	65	0.01	0.83	89	0.01	0.11	57	<0.01	<0.01	69	<0.01	0.03	92
Mugilidae																		
<i>Mugil cephalus</i>	<0.01	<0.01	79	0.25	0.44	28	0.11	0.02	28	0.01	0.06	40	<0.01	0.02	75	0.02	0.08	51
<i>Mugil curema</i>	0.47	1.77	50	0.27	2.71	63	0.16	0.95	46	0.26	1.27	42	0.13	0.29	37	0.61	0.60	35
Mullidae																		
<i>Pseudupeneus grandisquamis</i>										<0.01	<0.01	38						
Opichthidae																		
<i>Myrichthys tigrinus</i>													<0.01	0.09	603			
Paralichthyidae																		
<i>Citharichthys gilberti</i>										<0.01	0.04	186	<0.01	<0.01	55			
<i>Etropus crossotus</i>	<0.01	0.01	66	<0.01	<0.01	92				0.01	0.04	58	<0.01	0.01	81			
Poeciliidae																		
<i>Poecilia butleri</i>										<0.01	<0.01	19	<0.01	<0.01	26	0.08	0.08	28
Polynemidae																		
<i>Polydactylus approximans</i>										<0.01	<0.01	42						
Scaridae																		
<i>Scarus ghobban</i>	<0.01	0.04	66							<0.01	<0.01	34						
Sciaenidae																		
<i>Bairdiella icistia</i>	<0.01		139															
<i>Cynoscion phoxocephalus</i>										<0.01	<0.01	43						
<i>Cynoscion xanthulus</i>	<0.01	<0.01	63	<0.01	0.05	159	<0.01	0.01	58							<0.01	0.09	145
<i>Elattarchus archidium</i>																<0.01	0.10	155
<i>Umbrina wintersteeni</i>										<0.01	0.02	156						
Serranidae																		
<i>Paralabrax maculatofasciatus</i>	0.08	0.72	68	<0.01	0.07	107	0.01	0.02	53	0.01	0.06	56	0.02	0.14	67			
Syngnathidae																		
<i>Hippocampus ingens</i>													<0.01	<0.01	113			
<i>Syngnathus auliscus</i>	<0.01	<0.01	88				<0.01	0.00	93									

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Table 1. (Continued)

	1980-1981									2009-2010								
	Balandra			Zacatecas			Enfermeria			Balandra			Zacatecas			Enfermeria		
Family/specie	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL	D	B	SL
Synodontidae																		
<i>Synodus scituliceps</i>														<0.01	0.03	183		
<i>Synodus sechurae</i>										<0.01	0.01	108						
Tetraodontidae																		
<i>Sphoeroides annulatus</i>	0.01	0.28	65	<0.01	0.07	71				<0.01	<0.01	33	<0.01	0.01	35			
<i>Sphoeroides lobatus</i>										<0.01	<0.01	24	<0.01	<0.01	41			
Urolophidae																		
<i>Urobatis halleri</i>										<0.01	0.13	198						

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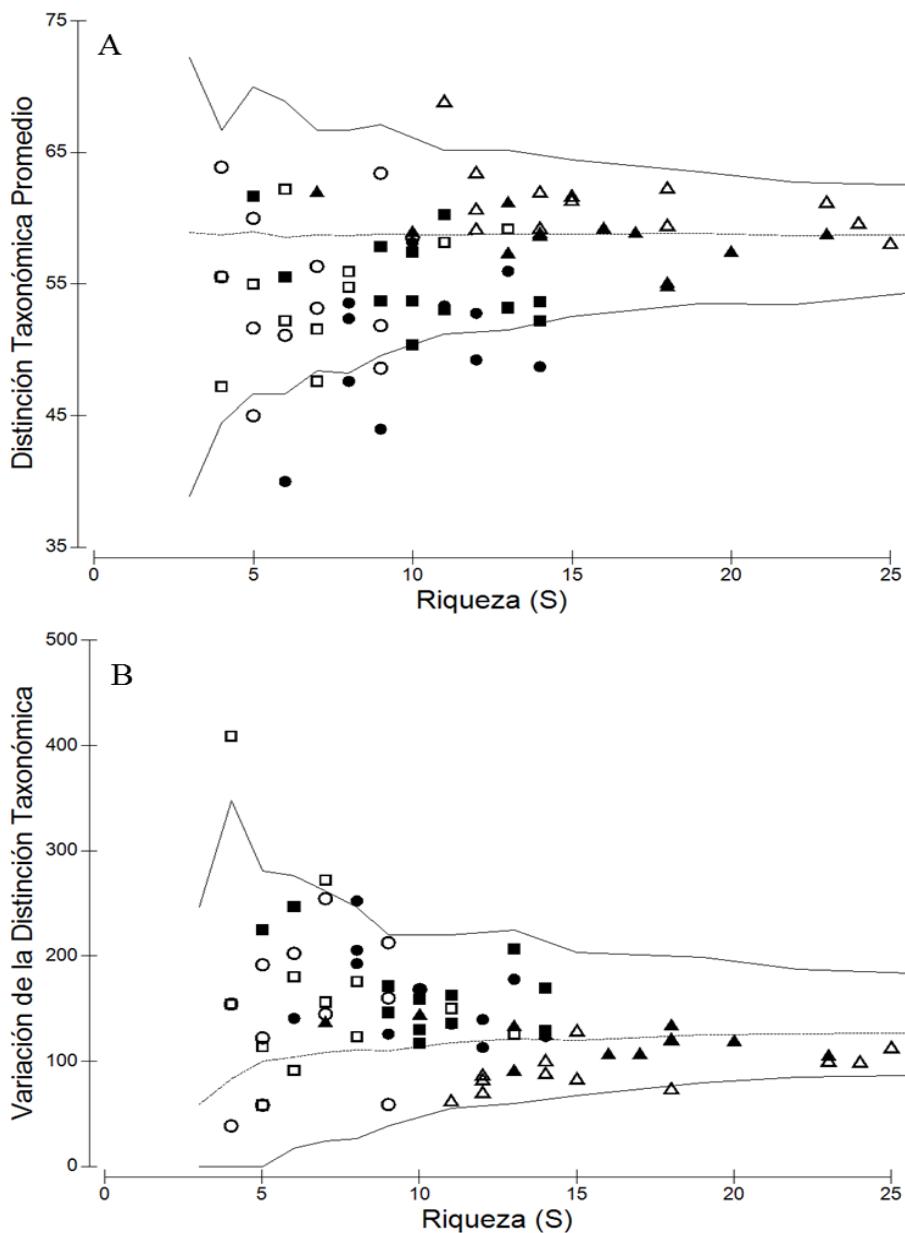


Figure 4. Funnel plot of A) the average taxonomic distinctness (ATD) and B) variation taxonomic distinctness (VarTD) in three estuary of La Paz Bay. Closed symbols are 1980–1981 and open symbols are 2009–2010. Triangles = Balandra, squares = Zacatecas, and circles = Enfermeria.

The result of the Olmstead-Tukey classification scheme showed that there was a high proportion of rare species, followed by a low proportion of frequent species in all mangrove systems in both periods of study. Only one species (*Mugil cephalus*) was classified as occasional (Zacatecas 2009–2010).

In Balandra, there were six dominants species in 1980, while in 2010 there were seven species (Table 2). In Zacatecas, five species were dominant in 1980, while four species were

dominant in 2010 (Table 3). In Enfermería, there were four dominant species in 1980 compared to only two in 2010 (Table 3).

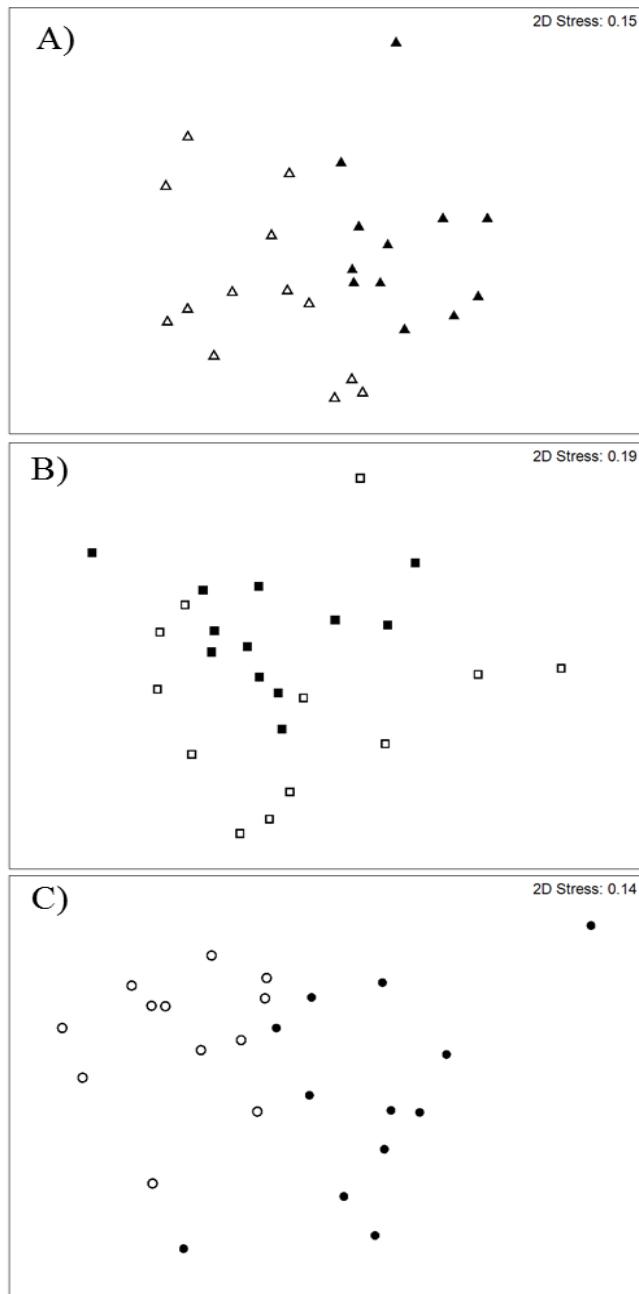


Figure 5. Two-dimensional multi-dimensional scaling plot including all fish samples for each in A) Balandra, B) Zacatecas and C) Enfermería. Closed symbols are 1980–1981 and open symbols are 2009–2010. Triangles = Balandra, squares = Zacatecas, and circles = Enfermería.

Table 2. Classification of the fish community in Balandra for both periods, according to their frequency and abundance

Dominants	Frequents	Rares	
<i>1980</i>			
<i>Mugil curema</i>	<i>Ctenogobius sagittula</i>	<i>Albula esuncula</i>	<i>Bairdiella icistia</i>
<i>Quietula y-cauda</i>	<i>Eucinostomus currani</i>	<i>Scarus ghobban</i>	<i>Conodon serrifer</i>
<i>Hyporhamphus naos</i>	<i>Paralabrax maculatofasciatus</i>	<i>Diodon holocanthus</i>	<i>Cynoscion xanthulus</i>
<i>Eucinostomus dowii</i>	<i>Gobiosoma chiquita</i>	<i>Etropus crossotus</i>	<i>Eugerres axillaris</i>
<i>Anchoa ischana</i>	<i>Sphoeroides annulatus</i>	<i>Hoplopagrus guentherii</i>	<i>Haemulon maculicauda</i>
<i>Diapterus brevirostris</i>	<i>Opisthonema libertate</i>	<i>Lutjanus aratus</i>	<i>Kyphosus analogus</i>
	<i>Lutjanus novemfasciatus</i>	<i>Orthopristis chalceus</i>	<i>Mugil cephalus</i>
	<i>Haemulon flaviguttatum</i>	<i>Lutjanus colorado</i>	<i>Orthopristis reddingi</i>
	<i>Gerres simillimus</i>	<i>Microlepidotus inornatus</i>	<i>Dormitator latifrons</i>
	<i>Lutjanus argentiventris</i>	<i>Exernes asper</i>	<i>Syngnathus auliscus</i>
	<i>Pomadasys branickii</i>	<i>Oligoplites saurus</i>	
<i>2010</i>			
<i>Eucinostomus dowii</i>	<i>Ctenogobius sagittula</i>	<i>Anchoa</i> sp.	<i>Lutjanus aratus</i>
<i>Gerres simillimus</i>	<i>Lutjanus argentiventris</i>	<i>Eucinostomus entomelas</i>	<i>Paraclinus mexicanus</i>
<i>Eucinostomus</i> sp.	<i>Haemulopsis axillaris</i>	<i>Opisthonema medirastre</i>	<i>Diodon holocanthus</i>
<i>Eucinostomus currani</i>	<i>Gobiosoma chiquita</i>	<i>Lile stolifera</i>	<i>Elops affinis</i>
<i>Diapterus brevirostris</i>	<i>Hyporhamphus naos</i>	<i>Anchoa nasus</i>	<i>Fistularia corneta</i>
<i>Mugil curema</i>	<i>Lutjanus novemfasciatus</i>	<i>Etropus crossotus</i>	<i>Haemulon scudderii</i>
<i>Anchoa ischana</i>	<i>Paralabrax maculatofasciatus</i>	<i>Chanos chanos</i>	<i>Poecilia butleri</i>
	<i>Mugil cephalus</i>	<i>Oligoplites fulgens</i>	<i>Scarus ghobban</i>
	<i>Albula esuncula</i>	<i>Erotelis armiger</i>	<i>Sphoeroides annulatus</i>
	<i>Caranx caninus</i>	<i>Eucinostomus gracilis</i>	<i>Synodus sechurae</i>
		<i>Exernes asper</i>	<i>Centropomus nigrescens</i>
		<i>Haemulon flaviguttatum</i>	<i>Citharichthys gilberti</i>
		<i>Haemulon steindachneri</i>	<i>Etrumeus teres</i>
		<i>Cynoscion phoxocephalus</i>	<i>Eugerres lineatus</i>
		<i>Haemulon sexfasciatum</i>	<i>Polydactylus approximans</i>
		<i>Centropomus viridis</i>	<i>Pomadasys branickii</i>
		<i>Sphoeroides lobatus</i>	<i>Umbrina wintersteeni</i>
		<i>Bathygobius ramosus</i>	<i>Urobatis halleri</i>
		<i>Hyporhamphus rosae</i>	<i>P. grandisquamis*</i>
		<i>Harengula thrissina</i>	

Pseudupeneus grandisquamis*.

Table 3. Classification of the fish community in Zacatecas for both periods, according to their frequency and abundance

Dominants	Frequents	Rares	
1980			
<i>Eucinostomus dowii</i>	<i>Ctenogobius sagittula</i>	<i>Sphoeroides annulatus</i>	<i>Opisthonema libertate</i>
<i>Mugil curema</i>	<i>Lutjanus argentiventris</i>	<i>Quietula y-cauda</i>	<i>Caranx sexfasciatus</i>
<i>Eucinostomus currani</i>	<i>Hyporhamphus naos</i>	<i>Pomadasys panamensis</i>	<i>Gerres simillimus</i>
<i>Pomadasys branickii</i>	<i>Lutjanus novemfasciatus</i>	<i>Paralabrax maculatofasciatus</i>	<i>Cynoscion xanthulus</i>
<i>Diapterus brevirostris</i>	<i>Eugerres axillaris</i>	<i>Lutjanus aratus</i>	<i>Elops affinis</i>
		<i>Lutjanus colorado</i>	<i>Etropus crossotus</i>
		<i>Albula esuncula</i>	<i>Oligoplites saurus</i>
		<i>Conodon serrifer</i>	<i>Xenichthys xanti</i>
		<i>Anchoa ischana</i>	
2010			
<i>Eucinostomus dowii</i>	<i>Paralabrax maculatofasciatus</i>	<i>Eucinostomus entomelas</i>	<i>Centropomus viridis</i>
<i>Diapterus brevirostris</i>	<i>Pomadasys branickii</i>	<i>Poecilia butleri</i>	<i>Citharichthys gilberti</i>
<i>Eucinostomus currani</i>	<i>Gerres simillimus</i>	<i>Chanos chanos</i>	<i>Etropus crossotus</i>
<i>Mugil curema</i>	<i>Ctenogobius sagittula</i>	<i>Mugil cephalus</i>	<i>Haemulopsis axillaris</i>
	<i>Haemulon sexfasciatum</i>	<i>Oligoplites altus</i>	<i>Hippocampus ingens</i>
	<i>Lutjanus argentiventris</i>	<i>Oligoplites saurus</i>	<i>Hyporhamphus naos</i>
Dominants	Frequents	Rares	
		<i>Anchoa</i> sp.	<i>Lutjanus novemfasciatus</i>
		<i>Diodon hystrix</i>	<i>Myrichthys tigrinus</i>
		<i>Haemulopsis leuciscus</i>	<i>Orthopristis chalceus</i>
		<i>Sphoeroides annulatus</i>	<i>Sphoeroides lobatus</i>
		<i>Eucinostomus gracilis</i>	<i>Synodus scituliceps</i>
		<i>Pomadasys macracanthus</i>	

Table 4. Classification of the fish community in Enfermeria for both periods, according to their frequency and abundance

Dominants	Frequents	Rares	
1980			
<i>Diapterus brevirostris</i>	<i>Quietula y-cauda</i>	<i>Mugil cephalus</i>	<i>Gobiosoma chiquita</i>
<i>Gerres simillimus</i>	<i>Mugil curema</i>	<i>Ctenogobius sagittula</i>	<i>Lile nigrofasciata</i>
<i>Eucinostomus dowii</i>	<i>Hyporhamphus naos</i>	<i>Dormitator latifrons</i>	<i>Cynoscion xanthulus</i>
<i>Eucinostomus currani</i>	<i>Lutjanus argentiventris</i>	<i>Eugerres axillaris</i>	<i>Exerpes asper</i>
	<i>Lutjanus novemfasciatus</i>	<i>Lutjanus colorado</i>	<i>Anchoa ischana</i>
		<i>Paralabrax maculatofasciatus</i>	<i>Oligoplites saurus</i>
		<i>Lile stolifera</i>	<i>Syngnathus aulicus</i>
2010			
<i>Diapterus brevirostris</i>	<i>Lile stolifera</i>	<i>Eugerres axillaris</i>	<i>Cynoscion xanthulus</i>
<i>Mugil curema</i>	<i>Gerres simillimus</i>	<i>Mugil cephalus</i>	<i>Elattarchus archidium</i>
	<i>Poecilia butleri</i>	<i>Haemulopsis axillaris</i>	<i>Lutjanus argentiventris</i>
	<i>Eucinostomus dowii</i>	<i>Chanos chanos</i>	<i>Lutjanus colorado</i>
	<i>Eucinostomus currani</i>	<i>Elops affinis</i>	<i>Lutjanus novemfasciatus</i>
	<i>Ctenogobius sagittula</i>	<i>Anchoa ischana</i>	
	<i>Centropomus viridis</i>	<i>Pomadasys macracanthus</i>	

DISCUSSION

The study by Maeda (1981) and the availability of a complete collection by Autonoma Universidad de Nuevo León made possible to compare the fish community structure from 30 years ago in three mangrove systems (Balandra, Zacatecas and Enfermeria) with current conditions. This highlights the importance of a good state of conservation and management of biological collections to analyze the evolution of ecosystem changes over time. This possibility is relevant today, given rapid changes that are happening in many coastal ecosystems in the world and the administration of land use and natural resources are demanding the detection of these changes and their assessment in relation to natural and anthropogenic factors.

This chapter is the first to evaluate and compare the fishes structure and composition of mangrove system from La Paz Bay in the southwestern Gulf of California with a sampling interval of 30 yrs and also complements existing studies of descriptions of the fish fauna (Abitia-Cárdenas et al. 1994, Balart et al. 1995, Castro-Aguirre and Balart 1997) and aspects

of the ecology and biology of some species of the region (Leija-Tristan et al. 1992, Balart et al. 1997).

In general, our results indicate the absence of a regional coherent trend in the changes in the fish community structure of the three mangrove systems. For example, while the highly impacted mangrove system, Enfermeria, exhibited significant changes in richness, evenness, and diversity between periods, the most pristine system, Balandra, only exhibited significant changes in biomass over time. These differences between neighboring mangrove systems suggest that large-scale processes such as changes in climate and rain patterns are negligible compared to more direct, local driving factors.

The Balandra system exhibited significant changes between periods only in total biomass. This mangrove system is presumably the least impacted of the three systems studied, because it is the furthest from an urban center, and there have been no changes in water circulation along the coast or in connection with the ocean. It is therefore considered relatively pristine. MDS analysis and ANOSIM suggests that were significant differences in the fish community structure between periods. The low similarity between periods was driven by differences in the species composition of the taxa that were rare. However, the dominant species remained consistent between sampling periods (*D. brevirostris*, *E. dowii*, *E. curranii*, and *M. curema*).

The Zacatecas system exhibited less dramatic but significant decreases in richness and biomass between periods, but no significant differences in assemblage structure, implying that there is a relatively low level of anthropogenic and natural impact in the habitat. This could be explained by the low number of visitors (it is out of the urban area) and that is considered critical habitat for nesting least tern (*Sterna antillarum*), a species considered endangered (NOM-059-2001 SEMARNAT 2002). Four species dominated the assemblage during both sampling periods: *E. dowii*, *D. brevirostris*, *M. curema*, and *Ctenogobius sagittula*. However, Zacatecas is a mangrove tidal channel that remains empties during low tide, and this condition has been associated with lower species richness in other similar systems (Faunce and Serafy 2006, Nagelkerken et al. 2008). This mangrove decreased richness and density by weight in 2010, compared to the historical collection, could be related to the constant dredging in the Ensenada de La Paz, which has presumably altered water circulation patterns n. Of the three mangrove systems, Zacatecas is the only one located within this protected lagoon (Figure 1). Unfortunately, there is no data to assess the relationship between dredging activity or possible changes in the natural mangrove geomorphology and structure of the fish community, and therefore, the interpretation presented here is tentative.

In the case of Enfermeria system, significant changes in richness, evenness, diversity and the fish community structure between periods were observed. This system has suffered the largest change over time in terms of the characteristics of the mangrove forest (Mendoza-Salgado et al. 2011, Santamaria-Gallegos et al. 2011.). It has had repeated impacts over time, starting with the construction of a highway between 1960 and 1964. Early 1980s, the mangrove suffered further modifications. Subsequently, a shrimp farming project was developed within the mangrove between 1994-2000. Over time, the restriction in connection between the mangroves and the bay has led to accumulation of sediments and high mortality of the mangroves (Mendoza-Salgado et al. 2011).

Although some of the ecological indexes showed a significant change, the base of assemblage's structure remained consistent. Only a few species, especially *D. brevirostris* and *M. curema*, were dominant and exhibited very high densities in the historical data that also

increased over time. In mangrove systems, a few species tend to dominate fish assemblages (Paperno and Brodie 2004, Jones and West 2005). Habitat alteration of mangroves tends to impact taxa with low resilience (James et al. 2008), which may explain why the fish community in the Enfermeria system was dominated by relatively few species. Such changes in diversity typically result from changes in the abundance and dominance of few species such as Gobiidae and Mugilidae (Ribeiro et al. 2008). In the present study, Enfermeria showed an increased dominance of Gerreidae.

The fish structure composition in Enfermeria system was dominated by species *Eucinostomus spp.* and *Mugil spp.*, which are taxonomically related. This could explain why some of the monthly values of ATD (five in 1980-1981 and two in 2009-2010) fell outside the confidence intervals of 95%. These two genera are known to be tolerant of high salinity and low dissolved oxygen conditions (Nagelkerken et al. 2001, 2002, 2008).

The results of ATD for Enfermeria could indicate that the community structure is not in equilibrium despite having been subject to a long-term disturbance. It has been shown that the response of the taxonomic indices to anthropogenic disturbances could reduce the taxonomic breadth, selecting closely related species, implying a high taxonomic homogeneity (Bevilacqua et al. 2011). Moreover, Bates et al. (2005) mention that the anthropogenic impacts generate very low values of ATD (as observed for Enfermeria, and that remove or eliminate top predators and remain tolerant species, which are closely related.

The characteristics of the landscape surrounding a mangrove forest estuary, including its relative position to the ocean, its bathymetry, dimensions of its connection to the coast, and its structural complexity (i.e., canopy cover, ramifications of the roots, inundation periods), have been shown to influence the resident fish community structure (Cocheret de la Morinière et al. 2004, Giarrizzo and Krumme 2007, Bosire et al. 2008). Anthropogenic modification of mangrove systems has been shown to lead to loss of aquatic habitat, shallower depths, and changes in physiochemical parameters, which generates a stressful environment for many fish species (Granek and Ruttenberg 2008). Reduced tidal exchange leads to decreased habitat quality, which in turn, lowers the influx of planktonic larvae and juveniles of mangrove dependent species (Layman et al. 2004). In addition, loss of canopy cover and defoliation increases the penetration of light, which, coupled with the loss of the structural complexity of the mangrove root system, results in a decrease in the available shelter against predators and the loss in diversity and density of species (Laegdsgaard and Johnson 2001, Manson et al. 2005, Nagelkerken et al. 2008).

Layman et al. (2004) characterized the fish assemblages among estuaries with different degrees of fragmentation due to road construction on Andros Island, The Bahamas, and found that hydrologic restrictions between mangrove systems and the ocean have a negative influence on fish and invertebrate abundance. Other studies conducted in The Bahamas have shown that mangrove forests fragmented by road construction exhibit a decrease in habitat quality and an alteration of the structure of fish assemblages compared with nearby mangrove systems (Layman et al. 2004, Taylor et al. 2007, Rypel and Layman 2008). Loss of habitat quality has also been linked to a decrease in fish secondary production (Valentine-Rose et al. 2007, 2011). The consequence of alteration of habitat is the loss of biodiversity, which can cause drastic changes in the ecosystem function (Naeem et al. 1994, Valentine-Rose et al. 2011). The lowest diversity we measured was in the Enfermeria mangrove compared to the other two estuary systems, thus the results are consistent with the extensive habitat modification that it has suffered over time.

ATD is an index of the phylogenetic relationships between organisms within a biological assemblage, which we used as a measure of the distinctness in taxonomic levels among the species captured in each mangrove site. A few of the monthly samples collected in the Enfermería mangrove (five in 1980–1981 and two in 2009–2010) had ATD values that fell outside the 95% confidence intervals of ATD. During those samplings, the species composition was dominated by *Eucinostomus* spp. and *Mugil* spp., which are taxonomically related to each other. ATD results for the Enfermería mangrove may also be indicative of a non-equilibrium community structure after a long-term disturbance. Bates et al. (2005) stated that anthropogenic impacts lead to very low values of the ATD (as was observed for Enfermería) since they tend to remove or eliminate top predators and leave closely related, tolerant species. The approach using ATD has proved valuable for detecting anthropogenic impacts in studies based on diffuse historic data sets built from unknown sampling effort (Warwick and Clarke 1998, Clarke and Warwick 1999). However, other studies have recognized the need to consider a complementary set of different measures in the assessment of community-level biodiversity (Heino et al. 2005).

We found that the classical community structure indices of species richness, diversity, and evenness better captured differences between systems and periods than the more recent approach that considers taxonomic distinctness.

According to the classification obtained with the Olmstead-Tukey classification, the dominant species in the three estuaries during both periods belonged to the Gerreidae (*E. dowii* and *D. brevirostris*) and Mugilidae (*M. curema*) families, although in Enfermería only *D. brevirostris* was dominant. These species are typical residents of mangrove systems, they preferentially use soft muddy substratum as a habitat (Nagelkerken et al. 2001, 2008), and have ample tolerance to environmental variability, allowing them to sustain higher population densities and persist in time and space than other species with smaller ecological flexibility. In fact, they are considered ordinarily resident in these mangrove systems (Gonzalez-Acosta et al. 2005). The species that were classified as frequent are considered to be euryhaline and use this type of environment as a primary nursery habitat because the mangrove roots provides them protection against predators (Aburto-Oropeza et al. 2009), which makes them dependent on mangroves, such as members of the Gobidae and Lutjanidae families (Warburton 1978). Most of the species we captured were classified as rare, and were either transients or not dependent on mangroves and only use the mangrove system for feeding or breeding, these include species of the Kiphosidae and Eleotridae families (Thompson and Forman 1987).

More than 50% of the species captured in this study are important commercially in the region (Ramírez-Rodríguez 1997), suggesting the importance of these ecosystems to sustain the fishery in the coastal lagoon and the adjacent bay of La Paz. Estuaries are ecosystems that have an important primary functions for sustaining fish communities providing a nursery habitat, as well as for foraging and recruitment. This chapter provides evidence that the most modified mangrove system exhibited the most notable changes in fish community structure. Restoration of the Enfermería mangrove system is needed, and demands further interdisciplinary studies. Lastly, this study highlights the importance of historical data for determining the status of mangrove systems which can provide better linkage between conservation and environmental management.

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EDITORS' CONTACT INFORMATION

Dr. Rafael Riosmena Rodriguez, PhD,

Programa de Investigación en Botánica Marina,

Departamento de Biología Marina,

Universidad Autónoma de Baja California Sur

Km 5.5 carr al sur Col Mezquitito, La Paz,

Baja California Sur, 23080, México

Tel: +52 612 151 3651 or 52-6121401096

Email: riosmena@uabcs.mx

Dr. Adrian F. González-Acosta, PhD,

Programa en Ciencias Marinas and MSci.

Programa en Manejo de Recursos Marinoa,

Departamento de Pesquerías y Biología Marina,

CICIMAR-IPN, Av. Instituto Politécnico Nacional S/N,

Col. Playa Palo de Santa Rita, La Paz,

Baja California Sur, 23096, México

Tel: +52 61212253669

Email: aacosta@ipn.mx

Dr. Raquel Muñiz-Salazar,

Laboratorio de Epidemiología y Ecología Molecular,

Universidad Autónoma de Baja California

Blvd. Zertuche y Blvd. de los Lagos s/n. Fracc.

Valle Dorado. C.P. 22890. Ensenada, Baja California

Tel: +52-646-1750707 Ext. 65305

Email: ramusal@uabc.edu.mx

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