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FISH ASSEMBLAGES IN THREE ARID MANGROVE SYSTEMS OF THE GULF OF CALIFORNIA: COMPARING OBSERVATIONS FROM 1980 AND 2010

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ABSTRACT

We evaluated the fish community structure in three mangrove systems from La Paz Bay, southwestern Gulf of California, Mexico, during two annual cycles separated by 30 yrs (1980-1981 and 2009-2010). The three mangrove system have suffered different degrees of anthropogenic impact that range from relatively pristine (Balandra) to minor impacts from development (Zacatecas), and to highly modified (Enfermería). A robust comparison between periods was attained by field sampling and identification of fishes using a museum collection. Species richness, density, biomass, Shannon diversity, Pielou evenness, and average taxonomic distinctness (ATD) were computed using data collected during each period (12 monthly samples). During both periods, a few species dominated fish abundance in the three systems [Eucinostomus dowii (Gill, 1863), Diapterus brevirostris (Sauvage, 1879), and Mugil curema (Valenciennes, 1836)]. Enfermería showed the most substantial changes in ecological indices; there were significant differences in mean monthly richness, density, and evenness over time. MDS and ANOSIM analyses revealed no differences in assemblage structure; however, SIMPER analysis indicated greater similarity in the assemblage structure overtime in Enfermería (38.06%) compared with Zacatecas (33.49%) and Balandra (9.71%). ATD values were relatively consistent between periods at Balandra and Zacatecas. However, a few samples collected at Enfermería had ATD values that indicated that the dominant species were closely related. This is likely due to the extensive habitat modification the system has suffered. Our study emphasizes the importance of long-term studies for understanding the changes in community structure in mangrove systems that are caused by habitat alteration.

Many coastal human communities benefit from the ecosystem services provided by mangrove forests, including coastal protection from erosion and hurricanes, provision of nursery grounds for commercial fish species, and carbon storage (Bouillon et al. 2008, Feller et al. 2010). Global estimates indicate a mean decrease in the coverage of mangrove forests on the order of 2% yr⁻¹ between 1970 and 2000 that has been attributed to coastal development, logging, and changes in land use patterns, such as for aquaculture development (Duarte et al. 2005, Donato et al. 2011, Irving et al. 2011). Such an elevated rate of habitat loss has a negative impact on mangrove ecosystems (Nelleman et al. 2009).

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 the 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 (Ramírez-García and Lot 1994).

Historical trends in mangrove coverage in the GC are unreliable. Mendoza et al. (1984) estimated that between 1973 and 1981, 22% of the area covered by mangrove forest in the southwestern GC suffered some sort of disturbance due to human influence. In contrast, Hak et al. (2008) analyzed satellite images and estimated an increase of >30% in the areal coverage of mangrove forest between 1986 and 2001 for southwestern GC. For La Paz Bay, located in the southwestern GC, there are reports of the combined negative effects of hurricanes and deforestation (Lechuga-Devéze 2009). The lack of consistency in the reports of temporal trends of mangrove forest coverage results at least in part from important information gaps concerning the temporal dynamics of mangrove ecosystems for many regions of the world, including the Baja California Peninsula (León de la Luz et al. 2011).

Mangrove systems are recognized as important sources of coastal fisheries productivity (Manson et al. 2005, Nagelkerken 2007), although there is evidence that the trees themselves do not provide an important carbon source to local food webs (Lugendo et al. 2007, Bouillon et al. 2008). 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). Worldwide, 30% of all commercial fishes are considered mangrove-dependent (Naylor et al. 2000). However, the relationship between mangrove habitat loss and fish productivity has remained elusive. The evidence for a negative relationship is still indirect and mostly circumstantial, and a major impediment to a thorough evaluation of the nature of their mangrove dependency has been due to limited research (see Blaber 2007). More thorough and interdisciplinary research is needed to better understand mangrove ecosystems and their dynamics (Dahdough-Guebas and Koedam 2008).

In the Mexican Pacific, studies of fish assemblages associated with mangrove ecosystems are scarce and local in scale, although the predominance of juvenile and sub-adult fishes suggests 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, Amezcua-Linares et al. 1987, Flores-Verdugo et al. 1990). Aburto-Oropeza et al. (2008) correlated coastal fish landings with the size of nearby mangrove systems in Gulf of California and found a positive direct relationship. However, they were unable to shed light on present or past trends in the relationship between fish abundance and mangrove coverage due to the lack of historical data. 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). 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.

In the present study, we capitalized on a thorough characterization of the fish community structure of three mangrove systems in La Paz Bay, Mexico, that was conducted in 1980–1981. The three mangrove systems have since suffered different degrees of anthropogenic habitat modification over time. We evaluated whether these systems show evidence of long-term changes in fish community structure by

comparing the data collected in the early 1980s with a survey performed almost 30 yrs later (2009–2010) using virtually identical methods. This study gave us the oppor-

yrs later (2009–2010) using virtually identical methods. This study gave us the opportunity to conduct a rare, long-term analysis of the relationship between mangroveassociated fish communities as a function of the degree of anthropogenic impact due to habitat modification.

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 1985). The area is considered arid and rainfall 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). 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). We focused on the three mangrove habitats for which historical information on fish communities exists. Red, black, and white mangrove species [*Rhizophora mangle* Linnaeus, *Avicennia germinans* Linnaeus (Linnaeus), and *Laguncularia racemosa* (Linnaeus)] are present at the three sites, which are separated by tens of kilometers (Fig. 1).

Balandra.—Balandra is a protected cove covering 52 ha of which 22 ha are occupied by mangrove forest. 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. The residence time of water within Balandra has been calculated as 1.5 d. This is considered to be the most pristine mangrove system in the region beacause it has suffered little anthropogenic habitat modification (Mendoza-Salgado et al. 2011).

Zacatecas.—This mangrove system 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 100 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 residence time of water is in the order of 1–6 d, depending on the stage of the tidal cycle (Mendoza-Salgado et al. 2011).

Enfermería.—This system 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. All these impacts have contributed to the system's deterioration (see review in Mendoza-Salgado et al. 2011). These modifications caused changes in the dynamics of water exchange, leading to lower current speeds and a relatively high residence time (20-26 d, Mendoza-Salgado et al. 2011), which has led to increased sediment accumulation. The present-day conditions are not conducive to the establishment of new mangrove seedlings, contributing to the decay in the long-term condition of the mangrove forest and additional losses of aquatic habitat (Fig. 2).



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) Enfermería, sampling site, and the major construction impacting the system (FW1: freeway built in 1964, FW2: freeway built in 2005, BR: shrimp-farming project built between 1994 and 2000).

Sampling Methods

Historical Samples.—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 Enfermería, respectively; Fig. 1).

Fish samples collected by Maeda-Martínez (1981) were 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.

Present-Day Samples.—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's (1981). Each captured individual was weighed, measured (SL), and identified. Specimens were incorporated into CIBNOR's Ichthyology Collection.

Data Analysis.—Because more than one tow was performed at Balandra and Zacatecas in 1980–1981, we calculated the mean value for a given parameter for a given month to allow for



Figure 2. Photographs of the Enfermería mangrove system, La Paz Bay, southwestern Gulf of California, Mexico, taken in 1982 and 2002, reflecting the effects of anthropogenic habitat modification. Photographs by Renato Mendoza (CIBNOR).

a simple comparison among mangrove systems. A sampling unit is thus defined as a single data point from each mangrove system each month. Here we focus on evaluating long-term changes in the average conditions of each mangrove system over the course of an annual cycle, and we will examine seasonality elsewhere.

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. Because of differences between the historical and present-day sampling in the area covered by each tow due to the seine length (ca. 97.5 m² in 1980–1981 and ca. 65 m² in 2009–2010), 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) and consider these indices 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) 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 path in the tree are set from 0 to 100. Additionally, ATD 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 importance 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. One-way analysis of similarities (ANOSIM) was performed for each study area to identify differences in community structure between periods. The last step was the application of a similarity percentage (SIMPER) test to determine which species contributed more to defining the assemblage of community; these tests were run with the software PRIMER ver. 6.0 (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 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 have a relative abundance higher than the corresponding arithmetic mean and lower than average frequency of occurrence and; (4) rare species, whose values of relative abundance and relative frequency of occurrence are lower than their respective arithmetic means.

Finally, we generated a size distribution of the entire fish community at each mangrove system during each period to evaluate whether the size structure of the fish community has changed over time and as a function of habitat modification. Size-frequency distributions were generated using 10-mm size classes. The percent difference in the relative abundance of each size class was also calculated to aid in the characterization of changes in size distribution over time. Differences in length-frequency distributions were tested with a Kolmogorov-Smirnov two sample test (Zar 1999).

Results

We recorded a total of 39,123 individuals for the 1980–1981 historical collection and 16,740 for the field sampling carried out during 2009–2010 (Table 1). 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 both periods were: *Eucinostomus dowii* (see Table 1 for species authorities), *Mugil curema, Diapterus brevirostrisn,* and *Eucinostomus currani* at Balandra; *E. dowii, D. brevirostris, E. currani,* and *M. curema* at Zacatecas; and *D. brevirostris, Gerres simillimus, E. dowii,* and *M. curema* at Enfermería (Table 1). The species that contributed the most biomass to the catch during both periods were: *Hyporhamphus naos, M. curema,* and *E. dowii* for Balandra; *Pomadasys branickii, M. curema, E. dowii,* and *D. brevirostris* for Zacatecas; and *D. brevirostris, G. simillimus,* and *M. curema* for Enfermería. In general, the species with higher densities and biomass were from the mangrove dependent families, Gerreidae and Mugilidae.

At Balandra, there were no significant differences between periods in species richness (t = -0.61, P = 0.54), Pielou evenness (t = 0.59, P = 0.56), and Shannon-Wiener diversity (t = 0.29, P = 0.77; Fig. 3). Richness was consistently higher in Balandra during both sampling periods compared to the other mangrove systems. At Zacatecas, there were significant differences between 1980–1981 and 2009–2010 in species richness (t = 2.81, P = 0.01), but not in evenness (t = 0.36, P = 0.72) or diversity (t = 1.81, P = 0.08). Enfermería exhibited the most drastic change in its community structure over time with significant changes evident in species richness, evenness, and diversity (t = 3.12, P < 0.01; t = 4.16, P < 0.01; and t = 4.77, P < 0.01; respectively). Evenness values were comparable between all mangrove systems and periods, except for the dramatically lower values found for Enfermería for the 2009–2010 sampling period. Shannon-Wiener diversity was relatively high in Balandra during both periods. However, for Zacatecas and Enfermería, diversity was lower during the 2009–2010 period than in the historical data set.

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 Enfermería, respectively; Fig. 4). Biomass did change significantly between periods for Balandra (t = 2.550, P = 0.017) and Zacatecas (t = 0.017)

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Carangidae																	
Caranx caninus Günther, 1867	I			,	ı	ī	ı	ī	ı	<0.01	61	Ц	ı	ı ı	ı	I	ī
Caranx sexfasciatus Quoy and Gaimard, 1825	,		v	0.01	101	К	ı	ī	ī	ī	ī	ī	ı	1	Ţ	I	ī
Oligoplites altus (Günther, 1868)	ī			ı	ı	ī	ı	ī	ī	I	ī	I	<0.01	34 R	ı	I	I
Oligoplites refulgens Gilbert and Starks, 1904	I			ı	ı	ī	ı	ī	ı	<0.01	49	К	ı	1	ı	I	ı
Oligoplites saurus (Bloch and Schneider, 1801)	<0.01	1 1	~	0.01	66	ž	≤0.01	27	К	ī	ī	I.	<0.01	57 R	I	I	ī
Centropomidae																	
Centropomus nigrescens Günther, 1864	I			ı	ı	ī	ı	ī	ı	<0.01	37	К	ı	1	ı	I	ı
Centropomus viridis Lockington, 1877	ī			ī	ī	ī	I	ī	ī	<0.01	61	Ч	<0.01	74 R	0.02	133	Ц
Chanidae																	
Chanos chanos (Forsskål, 1775)	ī			ı	ı	ī	ı	ī	ī	<0.01	83	К	<0.01	54 R	0.01	79	К
Clupeidae																	
Etrumeus teres (DeKay, 1842)	I			,	ı	ī	ı	ī	ı	<0.01	37	R	ı	ı ı	ı	I	ī
Harengula thrissina (Jordan and Gilbert, 1882)	I			,	ı	1	ı	ī	ı	<0.01	49	К	ı	1	ı	I	ī
Lile nigrofasciata Castro-Aguirre, Ruiz-Campos and Balart, 2002	I			ī	ı	ī	0.01	61	К	ī	ī	ī	ı	1	ı	I	ī
Lile stolifera (Jordan and Gilbert, 1882)	I			,	ı	ī	0.01	73	К	0.01	58	R	ı	ı ı	0.20	50	Ц
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Diodon holocanthus Linnaeus, 1758	<0.01 1	18 1	~	,	ı	ī	ı	ī	ı	<0.01	164	R	ı	ı ı	ı	I	ī
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Dormitator latifrons (Richardson, 1844)	<0.01	21	К	ı	ī	ı.	0.03	29	К	ı	ī	1	ī	i.	ī	ı.	ī	ī
Erotelis armiger (Jordan and Richardson in Jordan, 1895)	ı	ī	ı	I	I	ī	ī	ı	ı	<0.01	44	К	ī	ī	ī	ī	ī	ī
Elopidae																		
Elops affinis Regan, 1909	ı		ī	<0.01	64	К	ī	ī	,	<0.01	57	R	,	,	ī	0.00	79	К
Eugraulidae																		
Anchoa ischana (Jordan and Gilbert, 1882)	1.14	53	D	0.05	64	К	<0.01	89	К	0.09	52	D	ı	ī	ı	0.00	106	К
Anchoa nasus (Kner and Steindachner, 1867)	ı			I	ı	ī	ī	,	ī	0.01	112	К	ī	,	ī	ī	ī	ī
Anchoa sp.	ī		ī	ı	ı	ī	ī	ı	ı	0.02	33	ы	<0.01	38	Ч	ī	ī	ī
Fistulariidae																		
Fistularia corneta Gilbert and Starks, 1904	ı	ī	ī	ı	I	ī	ī	ı	ı	<0.01	291	R	ī	ı.	ī	ī	ī	ī
Gerreidae																		
Diapterus brevirostris (Sauvage, 1879)	0.18	34	D	0.33	36	D	1.88	40	D	0.30	35	D	0.33	35	D	4.42	43	D
Eucinostomus currani Zahuranec in Yáñez-Arancibia, 1980	0.11	32	Ц	0.33	28	D	0.33	33	D	0.37	27	D	0.13	26	D	0.04	46	ĹŢ
Eucinostomus dowii (Gill, 1863)	2.35	27	D	0.72	41	D	0.89	39	D	0.61	25	D	0.67	33	D	0.05	30	[L]
Eucinostomus entomelas Zahuranec in Yáñez-Arancibia, 1980	ï		ī	ı	ı	ī	ī	ī	,	0.02	78	R	0.03	LL	Ч	ī	ī	ī
Eucinostomus gracilis (Gill, 1862)	ı	ī	I	I	I	ī	ī	ı	I	<0.01	28	ч	<0.01	17	Ч	ī	ī	ī
Eucinostomus sp.	ı	ī	ī	ı	I	ī	ī	ı	I	0.55	19	D	ī	ı.	ī	ī	ī	ī
Eugerres axillaris (Günther, 1864)	<0.01	56	К	0.01	62	Ч	0.02	LL	К	ı	ī	1	ī	i.	ī	0.03	63	Ч
Eugerres lineatus (Humboldt in Humboldt and Valenciennes, 1821)	ī		ī	ı	ı	ī	ī	ı	ı	<0.01	60	К	ī	ī	ī	ī	ī	ī
Gerres simillimus Reagan, 1907	0.03	30	Ц	<0.01	68	Ч	1.67	30	D	0.55	25	D	0.01	4	Ц	0.16	32	ĹŢ
Gobiidae																		
Bathygobius ramosus Ginsburg, 1947	ı	ī	ī	ı	ī	ı.	ī	ī	ı	<0.01	60	К	ī	i.	ī	ı.	ī	ī
Ctenogobius sagittula (Günther, 1861)	0.03	43	ц	0.07	57	ц	0.03	56	К	0.05	38	Ц	0.01	58	Ц	0.03	61	Ľ
Gobiosoma chiquita (Jenkins and Evermann, 1889)	0.02	27	Ц	ı	ī	ī	0.01	24	К	0.01	28	Ц	ı	ī	ī	ī	ī	ī
Quietula y-cauda (Jenkins and Evermann, 1889)	0.42	27	D	<0.01	26	К	0.23	28	Ц	ı	ı.	ī	ī	ı.	ī	ī	ı.	ī

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Haemulidae																		
Conodon serrifer Jordan and Gilbert, 1882	<0.01	91	ц	<0.01	LL	К	ī		I	1	I	I	I	I	ī	I	ī	
Haemulon flaviguttatum Gill, 1862	0.03	80	Ц	I	ı	I	ī	1	<0.0>	1 6	0 R	I	I	I	I	ī	ı	
Haemulon maculicauda (Gill, 1862)	<0.01	100	R	I	ı	ı	ī	1	I		I	I	1	I	ı	1	ı.	
Haemulon scudderii Gill, 1862	T	ı	1	I	ı	ı	ī	1	<0.0>	1 5	6 R	I	ı	I	ı	ľ	ı	
Haemulon sexfasciatum Gill, 1862	I	ı	ī	I	ı	I	ī	1	<0.0>	1 4	3 R	<0.01	4	Ц	I	ī	ı	
Haemulon steindachneri (Jordan and Gilbert, 1882)	I	ı	ī	ī	ı	ı	ī	1	<0.0>	1 8	7 R	ı	I	ī	I	I	ī	
Haemulopsis axillaris (Steindachner, 1869)	ı	ī	ī	ı	ī	ı	ī		0.0	2	2 F	<0.01	100	ž	0.02	46	Я	
Haemulopsis leuciscus (Günther, 1864)	I	ī	1	I	ī	ı	ī		I	I	I	<0.01	39	ĸ	ī	I	ī	
Microlepidotus inornatus Gill, 1862	<0.01	113	Я	I	ı	I	ī	1	I	1	I	I	I	I	I	ī	ı	
Orthopristis chalceus (Günther, 1864)	0.01	29	Я	ı	ī	ı	ī		I	1	I	<0.01	85	2	I	T	ī	
Orthopristis reddingi Jordan and Richardson in Jordan, 1895	<0.01	47	Я	I	ı	ı	ī	1	I	1	I	I	ı	I	ı	ľ	ı	
Pomadasys branickii (Steindachner, 1879)	<0.01	4	ГL	1.23	4	D	ı	1	<0.0>	1 5	4 R	0.02	8	<u>Г</u> .	ı	ı.	ı	
Pomadasys macracanthus (Günther, 1864)	I	ī	ī	ı	ī	ı	ī	1	I	I	I	<0.01	104	2	<0.0>	97	Ч	
Pomadasys panamensis (Steindachner, 1876)	I	ī	ī	0.02	40	К	ī	1	I	I	I	I	I	I	I	I	ī	
Xenichthys xanti Gill, 1863	ı	ī	1	<0.01	70	К	ı		I	1	I	ı	1	1	I	1	ı.	
Hemiramphidae																		
Hyporhamphus naos Banford and Collette, 2001	0.23	117	D	0.03	169	Ц	0.08	109 F	0.0	1 11	7 F	<0.01	185	ĸ	ľ	,	ī	
Hyporhamphus rosae (Jordan and Gilbert, 1880)	ı			ı	ī	ī	ī		<0.0>	1 12	0 R	ı	I	1	ı	1	ī	
Kyphosidae																		
Kyphosus analogus (Gill, 1862)	<0.01	104	R	ı	ı	ı	ī		ı	I	I	I	I	I	ı	ī	ı	
Labrisomidae																		
Exerpes asper (Jenkins and Evermann, 1889)	<0.01	30	R	ı	ī	v I	¢0.01	29 R	<0.0>	1 3	4 R	ı	I	1	ı	1	ī	
Paraclinus mexicanus (Gilbert, 1904)	I	ı		ı	ı	ı	ī	1	<0.0>	1 3	4 R	·	'		ı	'	·	

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Table 1. Continued.																		
				1980	-1981								2009-	-2010	_			
	Bala	andra		Enfe	rmeria		Zac	atecas		Bal	andra		Enfe	rmeri	a	Zac	atecas	
Family/species	D	SL	0-T	D	SL	0-T	D	SL	- L-O	D	SL	0-T	D	SL	0-T	D	SL (L-
Lutjanidae																		
Hoplopagrus guentherii Gill, 1862	<0.01	67	Я	ı	ī	ī	ı	I	ı	ī	ī	ī	ī	I	ī	I	ı	
Lutjanus aratus (Günther, 1864)	<0.01	94	Ч	<0.01	108	К	I	I	ī	<0.01	125	Я	ī	ī	ī	I	ı	
Lutjanus argentiventris (Peters, 1869)	0.01	36	ГĻ	0.02	67	Ц	0.03	62	Ц	0.02	52	Ľ	<0.01	76	Ľ	<0.01	115	Ч
Lutjanus colorado Jordan and Gilbert, 1882	<0.01	82	Я	<0.01	28	К	0.01	74	К	ı	ī	ı	ı	ī	ī	<0.01	112	К
Lutjanus novemfasciatus Gill, 1862	<0.01	81	Ľ	0.01	65	Ц	0.01	89	Ĺ	0.01	57	Ľ	<0.01	69	К	<0.01	92	К
Mugilidae																		
Mugil cephalus Linnaeus, 1758	<0.01	79	Я	0.25	28	0	0.11	28	К	0.01	40	Ц	<0.01	75	Ч	0.02	51	К
Mugil curema Valenciennes in Cuvier and Valenciennes, 1836	0.47	50	D	0.27	63	D	0.16	46	ц	0.26	42	D	0.13	37	D	0.61	35	D
Mullidae																		
Pseudupeneus grandisquamis (Gill, 1863)	ı	ī	ī	ı	T	ī	ī	ī	ī	<0.01	38	Я	ī	ī	ī	ī	I	i.
Opichthidae																		
Myrichthys tigrinus Girard, 1859	ı	ı	ı	ı	ī	ī	ı	I	ı	ī	ī	ī	<0.01	603	Ч	I	ı	
Paralichthyidae																		
Citharichthys gilberti Jenkins and Evermann, 1889	ı	ı.	ı	ı	ī	ı.	ī	ı	1	<0.01	186	К	<0.01	55	К	ı	ı	
Etropus crossotus Jordan and Gilbert, 1882	<0.01	99	Я	<0.01	92	Ч	ı	I	ī	0.01	58	К	<0.01	81	К	I	ı	ī
Poeciliidae																		
Poecilia butleri Jordan, 1889	ı	ī	ı	ı	ī	ī	I	I	ī	<0.01	19	Я	<0.01	26	Ч	0.08	28	[I]
Polynemidae																		
Polydactylus approximans (Lay and Bennett, 1839)	ı	ı	ı	ı	ī	ī	ı	ı	ı	<0.01	42	Я	ī	I	ī	I	ı	ī
Scaridae																		
Scarus ghobban Forsskål, 1775	<0.01	99	Я	ı	,	ī	ī	ı	ı	<0.01	34	К	ī	ī	ī	ı	ī	

			198	0-1981							2009	-2010				
	Bala	ndra	Enf	ermeria		Zaca	tecas		alanc	Ira	Enfe	ermeri	а	Zac	cateca	s
Family/species	D	SL 0-1	D	SL (- L-(D	SL O-T		SI	. 0-T	D	SL	D-T-0	D	SL	0-T
Sciaenidae																
Bairdiella icistia (Jordan and Gilbert, 1882)	<0.01	139 R	I	T	ī	ı	1	T	1	ı	ı	ī	ı	ī	ı	ī
Cynoscion phoxocephalus Jordan and Gilbert, 1882	I		ı	I				<0.0>	1 4	3 R	·	ī	ı	ī	ı	ī
Cynoscion xanthulus Jordan and Gilbert, 1882	<0.01	63 R	<0.01	159	R <	0.01	58 R	I	I	ı	ı	ī	ī	<0.01	145	К
Elattarchus archidium (Jordan and Gilbert, 1882)	I	1	I	I	ī	ı	1	I	I	ı	ı	ī	ī	<0.01	155	К
Umbrina wintersteeni Walker and Radford, 1992	I	1	I	T	ī	ı	1	<0.0>	1 15	6 R	ı	ī	ı	ī	ı	ī
Serranidae																
Paralabrax maculatofasciatus (Steindachner, 1868)	0.08	68 F	<0.01	107	R	0.01	53 R	0.0	1 5	6 F	0.02	67	Ц	ī	ı	ī
Syngnathidae																
Hippocampus histrix Kaup, 1856	ı	1	I	ī	1	ī	- 1	T	1	ī	<0.01	113	Я	ī	ī	ī
Syngnathus auliscus (Swain, 1882)	<0.01	88 R	I	I	V	0.01	93 R	I	I	I	ı	ī	ı	ī	ı	ī
Synodontidae																
Synodus scituliceps Jordan and Gilbert, 1882	I	1	I	T	ī	ı	1	T	I	ı	<0.01	183	Я	ī	ı	ī
Synodus sechurae Hildebrand, 1946	I	1	I	I	ī	ı	1	<0.0>	1 10	8 R	ı	ī	ı	ī	I	ī
Tetraodontidae																
Sphoeroides annulatus (Jenyns, 1842)	0.01	65 F	<0.01	71	R	ī	1	<0.0>	1 3	3 R	<0.01	35	R	ı.	ī	ī
Sphoeroides lobatus (Steindachner, 1870)	I	1	ı	ı		ī		<0.0>	1	4 R	<0.01	41	К	,	ı	ī
Urolophidae																
Urobatis halleri (Cooper, 1863)		т т	ı	ı	ī	ī	1	<0.0>	1 19	8 8	ī	ı.	ī	ı.	ī	ī



Figure 3. Comparison of fish community structure indices calculated 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.

2.207, P = 0.038), but not for Enfermería (t = -1.402, P = 0.175). The highest mean monthly density and biomass were found at Enfermería in the present-day collection (5.68 ± 1.21 ind m⁻² and 20.70 ± 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 (Fig. 5). ATD values that fall outside of the 95%



Figure 4. Fish biomass (g m⁻²) and density (ind m⁻²) in three mangrove forest estuaries in La Paz Bay, Mexico. Closed symbols are 1980–1981 and open symbols are 2009–2010. Triangles = Balandra, squares = Zacatecas, and circles = Enfermería. Mean (SE) are presented.

probability limits are markedly more dissimilar on a taxonomic level. Most of the samples (monthly ATDs) fell within the 95% probability limits, except for some samples from the historical collection from Zacatecas (January) and Enfermería (June, July, September, December, and January), and a few from 2009 to 2010 (October for Zacatecas and June and January for Enfermería). Interestingly, for Enfermería, 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.

The MDS analysis showed that the fish community collected in Balandra in 2009–2010 was distinct from the same system's historical data set as well as the other mangrove systems during both study periods (Fig. 6). The value of the stress of management was 0.19, indicating an adequate goodness of fit. However, analysis of similarity (ANOSIM) indicated no significant differences in the fish community structure between periods for Balandra (R = 0.97, P = 0.10), Zacatecas (R = 0.10, P = 0.10), or Enfermería (R = 0.47, P = 0.20).

SIMPER analysis indicated that 10 species defined the structure of the assemblage in 1980–1981 and 2009–2010 in Balandra, with a very low percentage of similarity between periods (9.71%). For both Zacatecas and Enfermería, there were seven key species in 1980–1981 and five species in 2009–2010, with the highest percentage of similarity between periods for Zacatecas (33.49%) and Enfermería (38.06%, Table 2).

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 1980–1981 (*M. curema, Quietula y-cauda, H. naos, E. dowii, Anchoa ischana,* and *D. brevirostris*), while in 2009–2010 there were seven species (*E.*



Figure 5. Funnel plot of the average taxonomic distinctness (D+) in three estuary of La Paz Bay. Closed symbols are 1980–1981 and open symbols are 2009–2010. Triangles = Balandra, squares = Zacatecas, and circles = Enfermería.



Figure 6. Two-dimensional multi-dimensional scaling plot including all fish samples for each month from three mangrove forest estuaries in La Paz, Mexico. Closed symbols are 1980–1981 and open symbols are 2009–2010. Triangles = Balandra, squares = Zacatecas, and circles = Enfermería.

dowii, G. simillimus, Eucinostomus spp., E. currani, D. brevirostris, M. curema, and A. ischana). In Zacatecas, five species were dominant (E. dowii, M. curema, E. currani, P. branickii, and D. brevirostris) in 1980–1981, while four species were dominant in 2009–2010 (E. dowii, D. brevirostris, E. currani, and M. curema). In Enfermería, there were only two dominant species in 2009–2010 (D. brevirostris and M. curema) compared to four in 1980–1981 (D. brevirostris, G. simillimus, E. dowii, and E. currani; Table 1).

There were significant differences between periods in the length-frequency distribution of fishes in each mangrove (P < 0.001, in all cases). The most abundant size class in the historical data set for Balandra was 11–25 mm SL. In the present-day data, there was an increase in the relative abundance of the 16–25 mm size class (Fig. 7A). Present-day conditions at Zacatecas also revealed a tendency toward dominance of smaller animals resulting primarily from a reduction in the relative frequency of the 40–45 mm size class (Fig. 7B). Enfermería was the only site that showed the opposite trend, with a decrease in the relative abundance of individuals <45 mm and a relative increase in the abundance of individuals from the larger size classes (Fig. 7C).

DISCUSSION

Long-term data sets of fish community structure are limited mostly to cases for which fishery-dependent data are available (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 dramatic changes in community composition. They suggested that within a region, populations may respond differently to climate change as well as to local environmental factors and interspecific ecological interactions. 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 density, biomass, and diversity as well as spatial differences in evenness and taxonomic distinctness measures. 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 the assemblage species composition, but noted that the changes in abundance and length-frequency distributions were not consistent among taxonomic groups. Larger species with slow-maturation exhibited the greatest changes in distribution, which they attributed to the influence of both commercial fishing and regional climates change. In contrast, few studies have examined changes in the community structure of coastal mangrove systems over the time scale of decades using a targeted and consistent sampling design.

We compared the fish community structure of three mangrove systems from La Paz Bay in the southwestern Gulf of California with a sampling interval of 30 yrs. Although our comparison of fish community structure was based on 12 monthly samples collected during two 1-yr periods (1980–1981 and 2009–2010), the sampling strategy was systematic and comparable. 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, Enfermería, exhibited significant changes in richness, evenness, and diversity (but not in density and biomass) between periods, the most pristine system, Balandra,

1980–1981 and 2009–2010.					
Species	Abundance (mean)	Similarity (mean)	Similarity (sim/SD)	Contribution (%)	Cumulative (%)
Balandra 1980–1981, mean similarity: 42.38					
Eucinostomus dowii	1.05	8.30	1.26	19.58	19.58
Quietula y-cauda	0.55	6.53	1.90	15.42	35.00
Mugil curema	0.54	6.15	1.73	14.52	49.52
Hyporhamphus naos	0.42	5.84	2.04	13.79	63.31
Anchoa ischana	0.62	3.14	0.66	7.41	70.72
Eucinostomus currani	0.27	3.11	0.91	7.33	78.05
Paralabrax maculatofasciatus	0.21	1.85	1.07	4.36	82.41
Ctenogobius sagittula	0.13	1.72	1.43	4.06	86.47
Gobiosoma chiquita	0.11	1.05	1.06	2.47	88.94
Sphoeroides annulatus	0.09	1.03	0.83	2.42	91.36
Balandra 2009–2010, mean similarity: 35.95					
Erotelis armiger	0.61	7.53	1.26	20.94	20.94
Kyphosus analogus	0.41	6.55	2.33	18.23	39.17
Elops affinis	0.44	4.90	2.18	13.62	52.79
Ctenogobius sagittula	0.34	4.07	1.17	11.33	64.12
Eucinostomus currani	0.44	3.00	0.33	8.34	72.46
Chanos chanos	0.18	2.56	1.06	7.11	79.57
Hippocampus sp.	0.10	1.26	1.00	3.50	83.08
Eugerres axillaris	0.39	1.15	0.36	3.20	86.28
Hyporhamphus naos	0.09	1.15	0.92	3.19	89.47
Mugil cephalus	0.07	0.73	0.65	2.03	91.50
Balandra 1980–1981 vs 2009–2010, mean similarity: 9.71					

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Species	Abundance (mean)	Similarity (mean)	Similarity (sim/SU)	Contribution (%)	Cumulative (%)
Zacatecas 1980–1981, mean similarity: 42.68					
Eucinostomus dowii	0.74	14.95	1.71	35.04	35.04
Mugil curema	0.45	10.48	2.30	24.56	59.60
Eucinostomus currani	0.39	4.66	1.11	10.93	70.52
Diapterus brevirostris	0.39	3.46	0.72	8.11	78.64
Pomadasys branickii	0.43	1.95	0.87	4.57	83.21
Ctenogobius sagittula	0.18	1.82	0.82	4.26	87.47
Lutjanus argentiventris	0.11	1.64	0.88	3.84	91.31
Zacatecas 2009–2010, mean similarity: 30.08					
Eucinostomus dowii	0.65	16.2	1.27	53.85	53.85
Diapterus brevirostris	0.36	5.61	0.89	18.66	72.51
Mugil curema	0.22	3.51	0.66	11.68	84.19
Paralabrax maculatofasciatus	0.08	1.15	0.37	3.82	88.01
Ctenogobius sagittula	0.05	0.81	0.38	2.70	90.71
Zacatecas 1980-1981 vs 2009-2010, mean similarity: 33.49					
Enfermería 1980–1981, mean similarity: 46.44					
Diapterus brevirostris	1.13	14.87	1.64	32.01	32.01
Gerres simillimus	0.95	8.87	1.51	19.10	51.11
Eucinostomus currani	0.50	8.27	1.73	17.82	68.93
Eucinostomus dowii	0.59	4.82	1.27	10.37	79.30
Quietula y-cauda	0.31	2.05	0.63	4.40	83.70
Lutjanus argentiventris	0.15	1.89	0.98	4.08	87.78
Mugil curema	0.25	1.75	0.72	3.77	91.56
Enfermería 2009–2010, mean similarity: 54.19					
Diapterus brevirostris	1.99	42.15	4.68	77.78	77.78
Gerres simillimus	0.26	2.86	0.89	5.28	83.06
Eucinostomus dowii	0.16	1.74	0.63	3.21	86.27
Mugil curema	0.36	1.30	0.39	2.40	88.68
Eucinostomus currani	0.13	1.25	0.48	2.31	90.98
Enfermería 1980–1981 vs 2009–2010. mean similarity: 38.06					

Table 2. Continued.

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Figure 7. Relative size-frequency distributions of the fish community at each of three mangrove forest estuaries—(A) Balandra, (B) Zacatecas, and (C) Enfermería—captured in 1980–1981 (black points, solid line) and 2009–2010 (white points, dashed line). The 85 mm standard length (SL) size class includes all fish >85 mm SL. Bars indicate the differences between the frequency distributions of the two periods (past–present).

only exibited 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. However, neither sampling period is considered anomalous in terms of interannual climatic or oceanographic variation. Since it has been well documented that El Niño/La Niña conditions lead to drastic changes in community structure of pelagic and coastal fish species in the region (Lluch-Cota et al. 1999), our results should be interpreted in the context of non-anomalous years.

The Balandra system exibited significant changes between periods only in total biomass. This system is the farthest from an urban center, circulation along the coast has not suffered alteration, and its connection with the ocean is not restricted. It is therefore considered relatively pristine. Although our MDS analysis suggests that in 2009–2010 the fish community structure of Balandra differed from the historical data set and the other two mangrove systems, the ANOSIM did not reveal significant differences. However, SIMPER analysis showed that the similarity between the two periods was only 9.71%. 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. currani,* and *Mugil curema*). This presence of a few individuals of a variable group of taxa between periods may be driven by processes associated with recruitment of fishes to the mangrove (Young et al. 1997, Griffiths and West 1999, Poizat et al. 2004), and should be the subject of future studies.

The Zacatecas system exibited less dramatic but significant decreases in richness and biomass between periods. MDS and ANOSIM analyses indicated no significant differences in assemblage structure, although SIMPER indicated some degree of similarity between periods (33.49%). Four species dominated the assemblage during both sampling periods: E. dowii, D. brevirostris, M. curema, and Ctenogobius sagittula. The Zacatecas mangrove has not suffered direct impacts from anthropogenic habitat modification, largely because it is considered key habitat for the Least Tern (Sterna antillarum) and is thus held under special protection (NOM-059-2001, SEMARNAT 2002). However, its tidal channel almost entirely empties during low tide, a phenomenon typically associated with lower fish species richness in mangrove systems (Faunce and Serafy 2006, Nagelkerken et al. 2008). The decrease in richness and biomass documented in 2009-2010 compared with 1980-1981 may also be related to alterations in the circulation patterns near the mangrove inside the Ensenada de La Paz coastal lagoon associated with several years of dredging (of the three mangrove systems, Zacatecas is the only one located inside this protected lagoon). Unfortunately, data to evaluate the relationship between dredging activity or potential natural changes in the mangrove's geomorphology and fish community structure are lacking, thus there is no evidence to support this hypothesis.

Finally, the Enfermería system has suffered the largest change over time in terms of the characteristics of the mangrove forest (Mendoza-Salgado et al. 2011, Santamaría-Gallegos et al. 2011). Even though construction of the main freeway was completed in 1964, following the early 1980s the site suffered further modifications. A shrimp-farming project was developed between 1994 and 2000, and an additional section of the freeway was built along the eastern side in 2005, further restricting the connection with the bay (see Mendoza-Salgado et al. 2011). 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).

Unlike the significant differences detected in richness, evenness, and diversity of the fish assemblage of the Enfermería system, MDS and ANOSIM analyses showed no significant differences between periods. However, the SIMPER analysis indicated greater similarity in the assemblage structure through time (38.06%) compared with Balandra and Zacatecas (9.71% and 33.49%, respectively). Therefore, 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 dominant fish assemblages (Paperno and Brodie 2004, Jones and West 2005). Habitat alteration of mangroves tends to impact taxa whose resilience is low (James et al. 2008), which

may explain why the fish community in the Enfermería 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 Gobidae and Mugilidae (Ribeiro et al. 2008). In the present study, Enfermería showed an increased dominance of Gerreidae.

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. 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 lower diversity we measured in the Enfermería mangrove compared to our other two estuary systems is thus consistent with the extensive habitat modification it has suffered over time.

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. The 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. ATD values were relatively consistent (within the 95% CI) between periods for most of the monthly data collected at Balandra and Zacatecas during both periods. However, 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 the ATD. During those samplings, the species composition was dominated by *Eucinostomus* spp. and *Mugil* spp., which are taxonomically related to each other. These two genera are known to be tolerant of environmental variability and extreme conditions such as high salinity and low concentrations of dissolved oxygen (Nagelkerken et al. 2001, 2002, 2008). The 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 ATD approach 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). Our results are consistent with this proposition; our interpretation of the data examined using both approaches was more informative and robust.

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, preferentially use soft muddy substratum as habitat (Nagelkerken et al. 2001, 2008), and have ample tolerance to environmental variability (Gonzalez-Acosta et al. 2005). The species that were classified as frequent are considered to be euryhaline or are known to be 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, such as species of the Kiphosidae and Eleotridae (Thompson and Forman 1987).

The comparison of length frequency distributions between 1980–1981 and 2009–2010 in Balandra and Zacatecas indicated that there was a slight shift toward smaller sizes over time. In contrast, there was a relative increase in the size of the fishes captured over time in the Enfermería estuary. The relatively larger individuals collected in 2009–2010 belonged predominantly to the Gerreidae and Mugilidae families. In all three systems, the presence of relatively small fishes is consistent with the nursery habitat function of mangroves and their use by small species (Laegdsgaard and Johnson 2001, Faunce and Serafy 2006, Nagelkerken et al. 2008).

Our study emphasizes the importance of long-term studies for understanding the changes in community structure in mangrove systems that are caused by habitat alteration. Our study 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, historical data on the status of mangrove systems can serve a better linkage between conservation and environmental management.

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