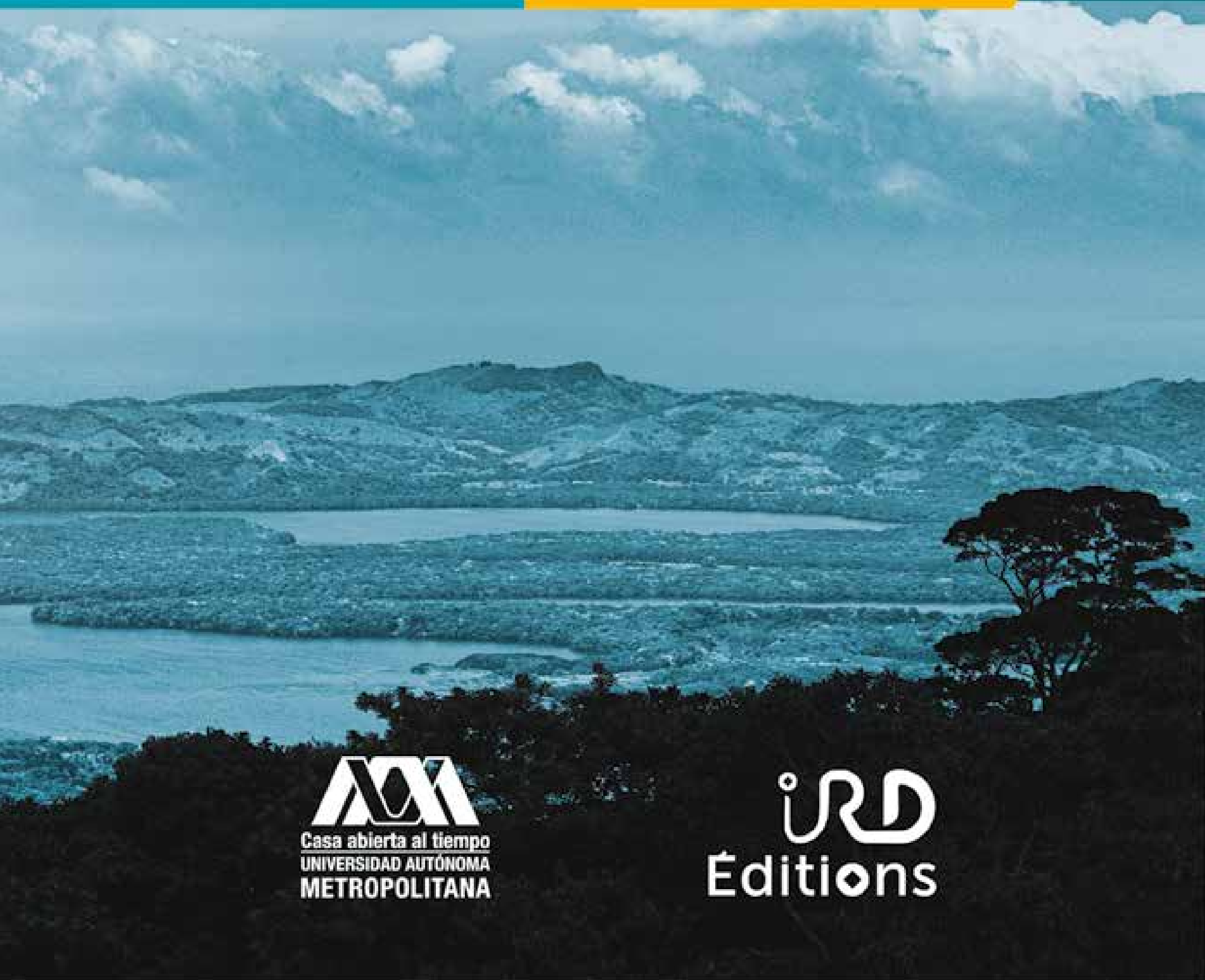
Ecology of the Sontecomapan Lagoon, Veracruz

María Elena Castellanos-Páez, Alfonso Esquivel Herrera
Javier Aldeco-Ramírez, Marc Pagano
(Coord.)



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Hecho en México

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Preface

Coastal lagoons are open systems that are influenced by their exteriors, both marine adjacent zones and runoffs as well as terrestrial surroundings with associated plant communities such as marshes, swamps and mangroves, as well as nearby human settlements.

One of the water bodies that have been least impacted is the Sontecomapan Lagoon, located in the State of Veracruz, in the Mexican Southeast. Here there are processes that are representative of unaffected tropical coastal systems, where mangrove, tide, freshwater runoffs and species diversity deserve a baseline study. The biogeochemical processes in clay and silt sediments, which generate nutrients susceptible of being exploited by the system's phytoplankton and the adjacent coastal sea, are the beginning of a chain of carbon transfer that has been little studied in tropical lagoons. Based on the interaction between phytoplankton and zooplankton, a wealth of species is generated that underlie fishing production from both, ecologically important species and those of commercial interest. The tide's effect allows chemical processes and their products to be exported to the adjacent coastal sea. With this, the often-underappreciated environmental service provided by coastal lagoons is remembered.

This book is an effort from Mexican and French scientists to rescue existing information about the Sontecomapan Lagoon System and what has been done so far in terms of research, and make it available to the scientific community.

The book consists of 12 chapters divided into 8 parts. The first part presents the physical and chemical characteristics of the lagoon water, the second part is an updated checklist of the lagoon's biota, the third part is a contribution on the bacterial community related to the nitrogen cycle. The fourth part is integrated by a study on the lagoon's phytoplankton and a research on the effect of mangrove litter extracts on the phytoplankton's growth and primary productivity. The fifth part includes a work on biomass, size, structure and trophic compartments of metazooplankton. The sixth part includes two investigations about fish; the first one is a review on species richness and the second one deals with the structure of the community and its relationship with the environment. The seventh part is subdivided into two sections; organic and inorganic contamination, and restoration and preservation; in the first one we find a work related with fecal contamination of anthropogenic origin and the enteropathogenic genera found in the lagoon water, and another work that addresses the water contamination by heavy metals; in the second section a study on mangrove's forest coverage change is included. The eighth part consists of the general discussion chapter.

The duration of the rainy and dry seasons in the region are related to the variability generated mainly by the temperatures in the Tropical Atlantic and the ENSO. While the duration of the northern winds season will depend on the oscillations of the circumpolar current (Jet Stream) and probably the influence of the North Atlantic Oscillation (NAO). Because of this, the authors use different temporalities based on their rainfall data and days with northern winds.

With this book we look to spread the knowledge generated through many years of work carried out by the authors and other researchers, and at the same time, encourage scientific research to carry on with the characterization of the Sontecomapan Lagoon, which is the basis needed for the creation of management plans for its sustainable use.

PART 1

Physicochemical characteristics

A multivariate approach to the spatial and temporal water characterization parameters at the tropical Sontecomapan Lagoon, Mexico

Alfonso Esquivel Herrera & Ruth Soto Castor

A multivariate approach to the spatial and temporal water characterization parameters at the tropical Sontecomapan Lagoon, Mexico

Alfonso Esquivel Herrera[*] & Ruth Soto Castor[**]

ABSTRACT: Sontecomapan is a coastal lagoon in the Gulf of Mexico Southeast which, altough small, due to its geomorphological characters has a complex behaviour. Four surveys were performed: in March and September 2009, and January and June 2010, which comprehend the three climatic seasons of the region, north wind-influenced, dry season and rainy season. Ten sampling points were selected for the measurement of physical, chemical and biological (chlorophyll *a*) parameters, which included the main environments found at this lagoon. These data were analysed through a multivariate approach. Sontecomapan was hypereutrophic based on total phosphorus (geometric means 0.202 to 0.386 mg L^{-1}), while it was oligotrophic for chlorophyll *a* (geometric means 0.40 to 2.08 mg m^{-3}). Comparison with chlorophyll *a* values reported for Sontecomapan since the early 1990's, show that the data presented here are the lowest for the period and this seems to be related to an increase in the flushing of plankton to the sea. An ordination by principal component analysis showed that for the period reported here, soluble reactive phosphorus and ammonium were negatively correlated or independent from salinity, so these nutrients originated either from streams or runoff. The ordination also confirmed that Sontecomapan can be divided into mesohaline, polyhaline and euhaline zones, with Arroyo La Palma as an independent point.

KEYWORDS: Mesotrophic, principal component analysis, coastal lagoon zonation, climatic seasons

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Introduction

Sontecomapan is coastal lagoon which, even if relatively small (891-952 ha) (Aké-Castillo *et al.*, 1995; CIMAR, 2011; Contreras, 1985), presents diverse environments due to its geomorphology and a correspondingly complex behaviour. It has a single and somewhat narrow entrance channel at its northeast end, so it can be considered as a *choked* lagoon (Duck & Da Silva, 2012). It is a brackish system, but its condition varies from marine, during the dry season, to freshwater at the end of its rainy season which result from the interplay between stream input or runoff and tidal influence (López-Portillo *et al.*, 2017).

As coastal aquatic systems where the exchange between epicontinental water masses and marine water occur, coastal lagoons are sites favouring environmental diversification development which provides habitat opportunities for diverse species, including several with economic importance (De la Lanza & Lozano, 1999). This is dependent on the conditions occurring at these shallow water bodies which can result in high biological productivity at all trophic levels but are, in turn, dependent on the lagoon's hydrological regime. The hydrological regime is the result of the interaction between the prevailing climatic conditions and the morphometric features of each lagoon, hence providing specific characteristics to each lagoon (González & Nodar, 1981).

One way to cope with this complexity is to describe coastal lagoons in terms of zones with a similar behaviour instead of doing it on a punctual scale, based on individual sampling sites. Thus, Yáñez-Arancibia (1987) agrees with Fairbridge's proposal of the following sectors for estuarine environments: a) the low or marine sector, connected to the open sea; b) the medium estuary, submitted to a strong mixing between seawater and freshwater; c) the upper or fluvial estuary, dominated by freshwater but sensitive to daily tidal effects. For Sontecomapan these sectors were recognized by Morán *et al.* (1995), who reported them as euhaline, polyhaline and mesohaline, respectively.

Even though, according to the former, the delimitation of these sectors could be performed solely on salinity, evident classification will be best accomplished if other relevant parameters are included by implementing tools such as multivariate analysis. As aforementioned, due to environmental components complexity (*i.e.*, water quality, hydrological parameters, and biological taxa) within the lagoon systems, multivariate statistical approaches have been widely applied to determine the relative influence of environmental factors and their interactions (Kim *et al.*, 2016; Gazzaz *et al.*, 2012; Tobiszewski *et al.*, 2010). This should be done on a temporal and spatial sense, since the extent of lagoon zones is dynamic and will differ from one season to another (De la Lanza *et al.*, 1998; Varona-Cordero & Gutiérrez-Mendieta, 2003).

The purpose of the present paper is to apply such a multivariate approach to Sontecomapan Lagoon which, due to its geomorphological features, has a complex behaviour and can be subdivided into sectors. This can also establish if there are changes in phytoplankton biomass and trophic status in an interannual scale. In recent years this lagoon has been, at the same time, considered as relatively unpolluted (Gallegos & Botello, 1986; Lot, 1971) but also as a man-disturbed ecosystem (Calva *et al.*, 2005; Ponce-Vélez *et al.*, 1994), so its characterization is important for management purposes, as those areas where water is stagnant would be more sensitive to pollution (Mitchell *et al.*, 2017; Newton & Mudge, 2003). This becomes even more important under the climatic change context.

Materials and Methods

Study area and sampling points

Sontecomapan is a Mexican coastal lagoon located between 18°30' N and 18°34' N and 94°47' and 95°11' W, with an area of about 891 ha. Its origin is volcanic, lava flows originally delimited its basin which was further modified by sediment transport through tides, streams and rivers (Lankford, 1977). This lagoon connects to the Gulf of Mexico through a 5.5 m deep, and 137 m wide mouth at its northern end, which then proceeds through a canal towards the Southwest to the main lagoon's body. The rest of the lagoon is shallower, with a mean depth of 1.5 m and is bordered by mangroves and wetlands which are part of the wildlife protection zone known as Reserva de la Biosfera de Los Tuxtlas (RBLT) (Calva *et al.*, 2005). The lagoon is fed by several rivers and streams, located mainly at its southern and south-eastern portion, though one of its main tributaries, La Palma stream, connects to the canal's west side (Fig. 1).

This region's climate comprises three seasons: north wind season (characterised by strong cold winds, either dry or humid, from the North), dry, and rainy season (Contreras, 1985; López-Portillo *et al.*, 2017). During the rainy season (late June to October), the lagoon receives a continuous freshwater inflow from small rivers and runoff. In contrast, at the peak of the dry season (March to early June), it shows marine salinities. During the Nortes season (November to February), the lagoon displays intermediate conditions, thus being brackish (Aké-Castillo & Vázquez, 2008).

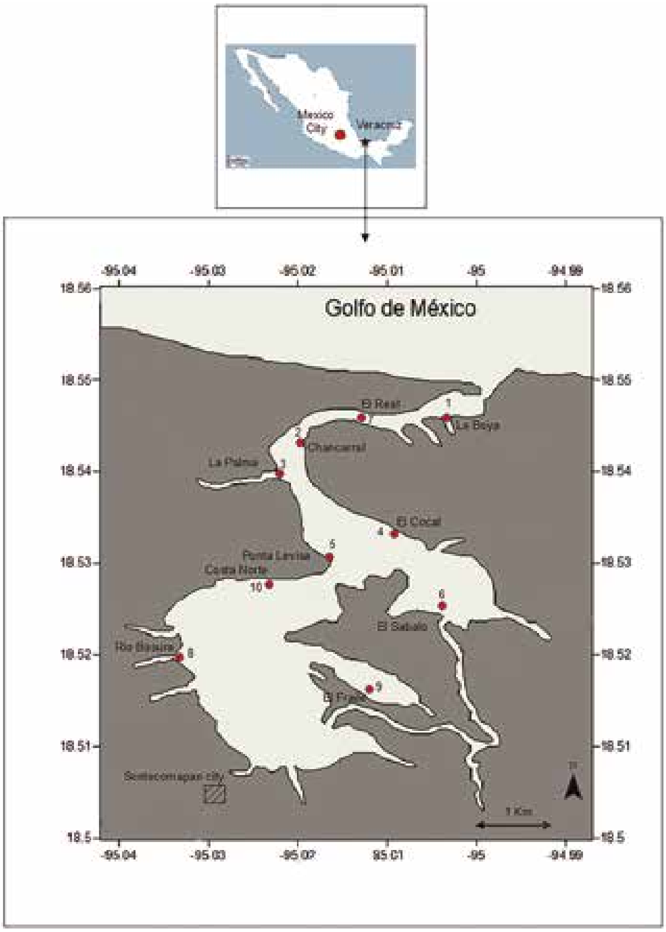

Figure 1. Sontecomapan lagoon. Sampling points. (Modified by Hervé, 2014).

The annual rainfall ranges between 3000 and 4000 mm. Lagoon water is considered as isothermal, varying from 22 °C in January to 26 °C in May. Generally, water turbidity is high, with a mean Secchi depth of 0.60 m. Water surface temperatures average 24 °C and there is no noticeable temperature gradient down the water column. The southern portion of the lagoon is mesohaline (salinity ranging from 5 to 18 PSU), the central portion is polyhaline (salinity ranging from 25 to 30 PSU) and the mouth is euhaline (salinity ranging from 30 to 40 PSU) (Morán *et al.*, 1995).

Samples were taken at ten stations (Fig. 1). The sampling station locations (Table 1) were selected as those representative of the diverse ecological and hydrological features; these have been set as interdisciplinary sampling points as in Benitez-Díaz-Mirón *et al.* (2014).

Table 1. Station name, number, abbreviation and geographical coordinates of sampling points

Sampling Point	Coordinates	Abbreviations
La Boya	N 18°33’02.6”	LB
1	W 94°59’26.8”	
El Chancarral	N 18°32’53.2”	CH§
2	W 95°00’52”	
La Palma	N 18°32’21”	LP
3	W 95°01’02.1”	
El Cocal	N 18°32’21”	CO
4	W 95°00’24.8”	
Punta Levisa	N 18°32’10”	LE
5	W 95°00’42.3”	
El Sabalo	N 18°32’09.8”	SA
6	W 95°00’49.2”	
El Real	N 18°33’19.3”	ER
7	W 95°00’51.7”	
Rìo Basura	N 18°31’41.3”	BA
8	W 95°02’68”	
El Fraile	N 18°30’51.7”	FR
9	W 95°00’39.5”	
Costa Norte	N 18°32’49”	
10	W 95°01’20.7”	CN

Samples were taken in the dry and north wind seasons (March 2009 and January 2010, respectively) and in the rainy season (September 2009 and June 2010), covering the three climatic seasons for this region (López-Portillo *et al.*, 2017).

Physical and chemical water analysis

Physical and chemical measurements were taken at each sampling station using a Van Dorn sampler 0.3 m below the surface and 0.3 m above the bottom. YSI multiparametric probes were used for temperature and salinity; dissolved oxygen concentrations were analysed using the Winkler method (Strickland & Parsons, 1972). Whatman GF/F glass-fibre filters were used for the spectrophotometric determination of chlorophyll *a*, through the Jeffrey & Humphrey's trichromatic method (1975). The filtrate was used to determine the nutrient concentrations (nitrate, nitrite, total ammonia, soluble reactive phosphorus [SRP], and total phosphorus [TP] after persulphate digestion) with a HACH Odyssey DR/2000 spectrophotometer, using the methods described by Strickland & Parsons (1972) and Aminot & Chaussepied (1983). Ammonium (NH_4^+) concentrations were computed from total ammonia through Emerson *et al.*'s equation using water temperature and pH (US-EPA, 2013).

Statistical analysis

Plots of the 95% confidence intervals for the mean were obtained for each parameter, classifying them by survey and sampling point and by survey and sampling depth (surface or bottom); when these confidence intervals overlap, at least partially, then they are statistically equal. Shapiro-Wilk's test for normality was applied to data to assess if a parametric ANOVA or a non-parametric Kruskal-Wallis should be applied to each variable. ANOVA was performed to environmental variables to determine if significant variations occurred due to sampling depth, sampling site or survey, and their combinations. Tukey's HSD (Honest Significant Difference) was applied as a post-hoc test whenever significant differences occurred, for determining which cases were responsible, this was performed on a survey and sampling point basis, and on a survey and sampling level basis. Environmental data ($NH4^+$, NO2, NO3, SRP, TP, chlorophyll *a*, dissolved O_2, and salinity) were transformed to *Z* (Legendre & Legendre, 1984) for their cluster analysis based on Euclidean distance and Ward's aggregation algorithm for clustering by sampling sites and survey; values (X) were transformed to $\log_{10}(X+1)$ prior to their ordination through principal component analysis, in order to analyse the links between environmental conditions (Pielou, 1984). For all these analysis, Statistica for Windows 7 (StatSoft, Tulsa, OK) was employed.

Results

Normality tests and ANOVA

Shapiro-Wilk's normality test showed that only dissolved oxygen had a normal distribution (Table 2). Thus, the Kruskal-Wallis ANOVA was applied to the rest of the parameters and found significant differences due to survey, and in the case of total phosphorus also by sampling site (Table 3). For dissolved oxygen, the parametric ANOVA found differences due to survey, to sampling point and also to interaction between survey and sampling point (Table 4). None of them found differences due to sampling level (surface/bottom), proving that the water column was mixed and not stratified at any of the sampling points or surveys, as confirmed in Table 6.

Table 2. Shapiro-Wilk's test for normality

Variable	Shapiro-Wilk's W	P
Water temperature	0.92045	0.00010
Salinity	0.90900	0.00003
NH_3	0.79669	<0.00001
$^-NO_2$	0.76349	<0.00001
$^-NO_3$	0.44799	<0.00001
SRP	0.61526	<0.00001
TP	0.65598	<0.00001
O_2	0.97928	0.21998
chlorophyll a	0.50966	<0.00001

Table 3. Non-parametric Kruskal-Wallis ANOVA

Parameters	Source of variation	Kruskal-Wallis' H	p
Water temperature	Survey	46.1423	<0.0001
	Sampling site	7.8371	0.5506
	Surface/Bottom	0.3013	0.5831
Salinity	Survey	46.0585	<0.0001
	Sampling site	12.3598	0.1938
	Surface/Bottom	2.5074	0.1133
NH_4^+	Survey	42.5043	<0.0001
	Sampling site	10.8434	0.2866
	Surface/Bottom	0.1887	0.6640
$^-NO_2$	Survey	9.3835	0.0246
	Sampling site	7.7688	0.5576
	Surface/Bottom	2.7154	0.0994
$^-NO_3$	Survey	51.3111	<0.0001
	Sampling site	7.2948	0.6065
	Surface/Bottom	0.1120	0.7379

Table 3. Non-parametric Kruskal-Wallis ANOVA (conti.)

Parameters	Source of variation	Kruskal-Wallis' H	p
SRP	Survey	33.8244	<0.0001
	Sampling site	8.0384	0.5303
	Surface/Bottom	0.3279	0.5669
TP	Survey	15.2475	0.0016
	Sampling site	22.1218	0.0085
	Surface/Bottom	0.1837	0.6682
chlorophyll a	Survey	23.5803	<0.0001
	Sampling site	4.0653	0.9071
	Surface/Bottom	1.2855	0.2569

Table 4. Parametric ANOVA for dissolved oxygen

Source of Variation	SS	D.F.	MS	F	p
Survey	37.049	3	12.350	3.946	0.0115
Surface/Bottom- Survey	3.805	3	1.268	0.405	0.7497
Surface/Bottom	8.327	1	8.327	2.632	0.1100
Sampling point	68.409	9	7.601	2.402	0.0213
Surface/Bottom- Sampling point	7.905	9	0.878	0.278	0.9784
Survey- Sampling point	108.673	27	4.025	2.666	0.0024

Temporal and spatial distribution of parameters

Figure 2 presents the 95% confidence intervals for the mean of each of the water parameters classified by survey and sampling point; Figure 3 classifies them by survey and sampling level. In general terms, mismatch between confidence intervals points to a significant difference; the data responsible for significant differences are shown in the tables 5 and 6, according to Tukey's post-hoc analysis, the first column contains the differing data, and the second column shows to which other data they exhibits a pairwise difference. Temperature did not differ between surface and bottom, the highest temperatures occurred during June 2010, when temperature was higher at most of the sampling points, attaining 33 °C at Sábalo, except at Levisa and El Real, where the highest temperatures were during September 2009. For salinity (Fig. 2a, Fig. 3a) the graphs show that the minima were found during September (rainy season), when even at La Boya salinity was less than 5 PSU, on the other hand, the highest salinities were in March (dry season), when salinities from 30 to 35 PSU occurred at La Boya, El Chancarral and El Real. Tukey's test showed that differences also occurred among the sampling point at Basura and the points at La Barra and Chancarral, during the dry season (Table 5).

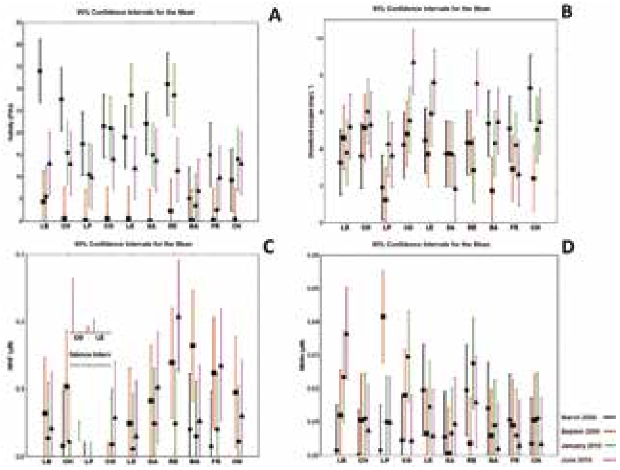

Figure 2A. 95% intervals for the mean by sampling point and survey for:
A) Salinity B) Dissolved oxygen
C) Ammonium D) Nitrite.

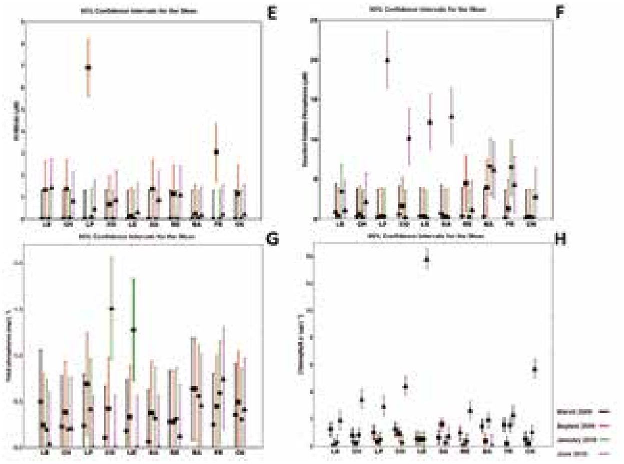

Figure 2B. (continuation) 95% intervals for the mean by sampling point and survey for:
E) Nitrate F) SRP
G) TP H) Chlorophyll *a*.
BA: Basura, CH: Chancarral, CO: Cocal, CN: Costa Norte, ER: El Real, FR: Fraile, LB: La Barra, LE: Levisa, LP: La Palma, SA: Sábalo.

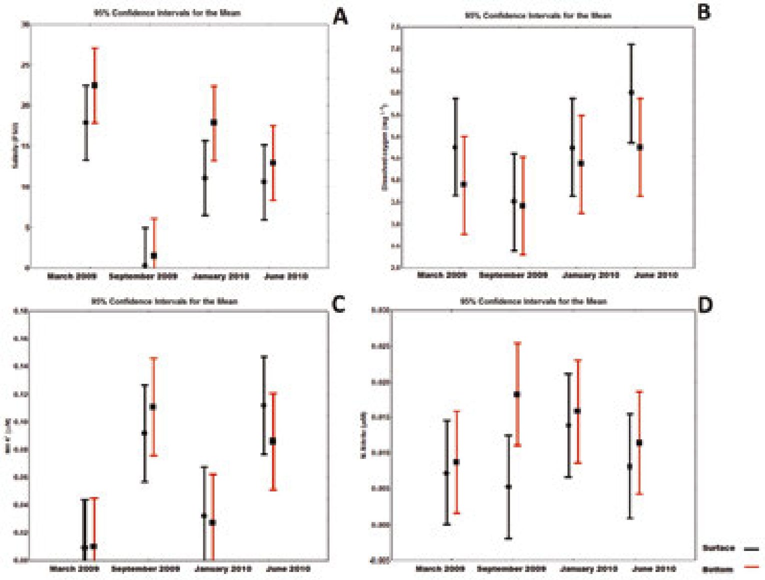

Figure 3A. 95%-intervals for the mean by sampling level and survey for:

A) Salinity	B) Dissolved oxygen
C) Ammonium	D) Nitrite

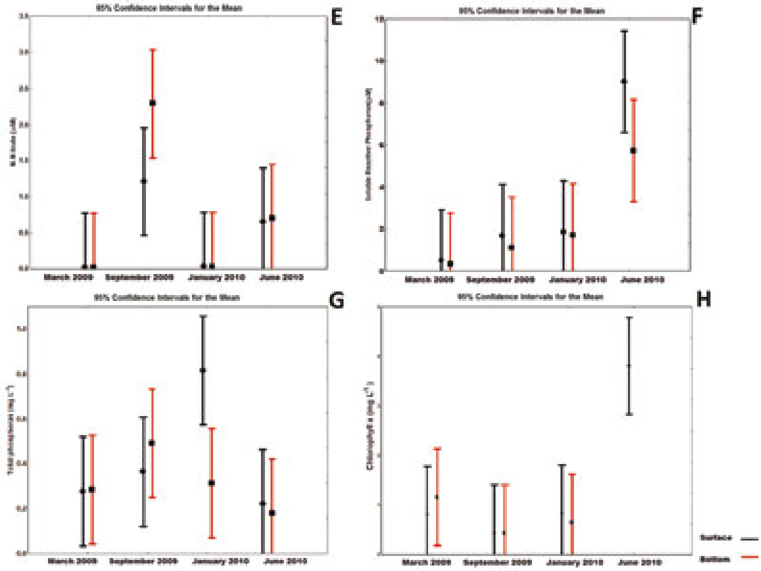

Figure 3B. (continuation) 95% intervals for the mean by sampling level and survey for:

E) Nitrate	F) SRP
G) TP	H) Chlorophyll *a*.

Table 5. Data for which significant differences were detected through Tukey's HSD ($p<0.05$). Paired comparisons by survey and sampling point

The prefix indicates the survey (M, March 2009; S, September 2009; J, January 2010; JU, June 2010), the suffix stands for the sampling point (BA, Basura; CH, Chancarral; CO, Cocal; CN, Costa Norte; ER, El Real; FR, Fraile; LB, La Barra; LE, Levisa; LP, La Palma; SA, Sábalo).

Chlorophyll	
JU-CH	ALL, EXCEPT M-LB, M-CP, M-BA, MFR, S-SA
JU-LE	ALL
JU-ER	M-CH, M-LE, M-SA, M-CN, S-LB, S-CH, S-LP, S-LE, S-ER, S-BA, S-FR, S-CN, J-LB, J-CH, J-LP, J-CO, J-LE, J-SA, J-ER, JU-LE
JU-CN	ALL, EXCEPT JU-CO
JU-FR	S-LB, S-CH, S-LP, S-LE, S-SA, S-ER, SBA, S-FR, S-CN, J-LB, J-LP, J-CO, J-SA, J-ER, JU-CO, JU-LE, JU-BA
M-FR	S-LB, S-ER, S-SA
S-LB	S-SA
O_2	
M-CN	M-LP, S-LP, S-BA
JU-CO	M-LB, M-LP, S-LP, S-BA, S-FR, S-CN, J-ER
JU-LE	M-LP, S-LP, S-BA, S-CN
JU-ER	M-LP, S-LP, S-BA, S-CN, JU-SA
JU-SA	M-CN
JU-FR	JU-CO
SRP	
JU-LP	ALL
JU-LE	ALL, EXCEPT S-ER, S-BA, J-LB, J-BA, J-FR, JU-CH, JU-LP, JU-CO
JU-SA	ALL, EXCEPT S-ER, S-BA, J-LB, J-BA, J-FR, JU-LP, JU-CO, JU-LE
JU-ER	JU-LP, JU-LE, JU-SA
NO_3	
S-LP	ALL
NO_2	
S-LP	M-LB, M-CH, M-LP, S-SA
SALINITY	
M-BA	M-LB, M-CH, M-ER, J-LE, J-ER
M-LB	M-BA, M-CN, ALL SEPTEMBER 2009, J-LB, J-LP, J-BA, J-FR, JU-LB, JU-CH, JU-LP, JU-LE, JU-ER, JU-BA, JU-CN

Table 5. (Continuation)

Chlorophyll	
M-CH	M-BA, S-LB, ALL SEPTEMBER 2009, J-BA, J-FR
M-CO	S-CH, S-LP, S-CO, S-LE, S-SA, S-BA, S-FR, S-CN
M-SA	S-CH, S-LP, S-CO, S-LE, S-SA, S-BA, S-FR, S-CN, J-BA, J-FR, JU-LP, JU-FR, JU-BA
M-CN	M-LB, M-ER
S-LB	M-LB, M-CH, M-ER, J-LE, J-ER
S-CH	M-LB, M-CH, M-CO, M-SA, M-ER, J-LE, J-ER
S-CO	M-LB, M-CH, M-CO, M-SA, M-ER, J-LE, J-ER
S-CN	M-LB, M-CH, M-CO, M-SA, M-ER, J-LE, J-ER
S-LP	M-LB, M-CH, M-CO, M-SA, M-ER, J-CO, J-LE, J-ER
J-LB	M-LB, M-CH, M-CO, M-SA, M-ER, J-LE, J-ER
J-LE	J-BA, J-FR, JU-BA
J-ER	J-BA, J-FR, JU-BA

Table 6. Data for which significant differences were detected through Tukey's HSD ($p<0.05$). Paired comparisons by survey and sampling level.

The prefix indicates the survey (M, March 2009; S, September 2009; J, January 2010; JU, June 2010), the suffix stands for the sampling level (S, surface, B, bottom).

NH_4^+	
M-S	S-S, JU-S, S-B
S-S	M-S, M-B
JU-S	M-S, J-S, M-B, J-B
S-B	M-S, J-S, M-B, J-B
CHLOROPHYLL	
JU-S	ALL
O_2	
JU-S	S-S, S-B
TP	
J-S	M-S, JU-S, JU-B
SRP	
JU-S	M-S, S-S, J-S, M-B, S-B, J-B
M-B	JU-S, JU-B
SALINITY	

Table 6. (Continuation)

NH_4^+	
S-S	M-S, J-S, JU-S, M-B, J-B, JU-B
M-B	S-S, J-S, JU-S, S-F
S-B	M-S, M-B, J-B, JU-B
NO_3	
S-B	M-S, J-S, M-B, J-B

Table 7. Ranges of values for water parameters found at Sontecomapan lagoon.

Reference	Salinity	Dissolved Oxygen	Total Ammonia	Nitrite + Nitrate	Nitrate	Soluble Reactive Phosphorus	Total Phosphorus	chlorophyll a	Primary Production
Castro *et al.*,1985[a]	0 – 23 PSU				0 – 0.65 μM	0.03 – 4.02 μM			
Castro, 1986[b]	0 – 38 PSU		0.92 μM		0 – 2.5 μM	0 – 15.4 μM			
Castellanos, 2002[c]	7.8 – 25.5 PSU								
Contreras, 1991[d]			6.80 - 24.6 μM	2.01-14.71 μM			2.13 - 9.51 μM	4.78–76.16 mg m^{-3}	
Contreras, 1995[e]								5.20–76.10 mg m^{-3}	34 – 213 mgC m^{-3} h^{-1}
Morán, 1994[f]	0.27-37.55 PSU		2.1 - 53.4 μM	0.26-30.35 μM			1.85 – 9.29 μM	0 – 161.82 mg m^{-3}	
Suchil, 1990[g]	1.55–18.96 PSU		14.05-24.50 μM	5.17– 8.96 μM				5.28– 23.23 mg m^{-3}	34.37-213.71 mgC m^{-3} h^{-1}
Mondragón, 2005[h]	1 – 31 PSU	2.95-12.82 mg l^{-1}	0 – 0.2 mg l^{-1}	0 – 0.017 mg l^{-1}	0.1 – 2.8 mg l^{-1}	0.02 – 0.21 mg l^{-1}	0.02 – 0.24 mg l^{-1}	0.21–64.29 mg m^{-3}	11.76-378.12 mgC m^{-3} h^{-1}
This paper[1]	0 – 36 PSU	0 – 9 mg l^{-1}	0 – 0.354 mg l^{-1}	0 – 0.063 mg l^{-1}	0 – 10.8 mg l^{-1}	0.08 – 26.84 mg l^{-1}	0 – 2.29 mg l^{-1}	0 – 13.83 mg m^{-3}	

[a] Monthly surveys April 1983-March 1984; [b] Monthly surveys April 1983-March 1984; [c] Six surveys April 1995 – September 1996; [d] Not specified; [e] Monthly surveys; [f] Six surveys February-October 1992; [g] February, April, August, October 198; February & June 2005; [I] March & September 2009 January & June 2010

For dissolved oxygen, the strongest effect was for the interaction among survey and sampling point with El Cocal, Levisa and El Real, having the highest values during June (rainy season) (Fig. 2b, Fig. 3b). These same sampling points showed the highest chlorophyll *a* concentrations during the same survey, especially Levisa (Fig. 2h). Ammonium was at its highest during the rainy season surveys of September and June, particularly at El Chancarral and El Real (Fig. 2c, Fig. 3c). The highest Nitrite-N also occurred during the rainy season surveys, especially at La Palma in September, and La Barra in June (Fig.

2d, Fig. 3d). For Nitrate-N the highest concentrations occurred in September at La Palma, and a lesser but still high value at El Fraile, bottom concentrations were higher during this survey (Fig. 2e, Fig. 3e).

SRP presented the highest values during June at La Palma, El Cocal, Levisa and El Sábalo, at the middle portion of the lagoon (Fig. 2f, Fig. 3f, Table 5). TP was highest in surface samples from the January survey, at El Cocal and Levisa (Fig. 2g, Fig. 3g). In June chlorophyll *a* presented a value close to 14 μg L^{-1} at Levisa and other remarkable values ranging from 3 to 6 μg L^{-1} at Costa Norte, El Cocal, Chancarral and La Palma (Fig. 2h).

Multivariate Analysis

The dendrogram for sampling points at each survey is presented in Figure 4, which represents the average trend for the period described here. Four clusters appeared at an Euclidean distance of 10, which roughly corresponded to the four surveys. The greatest difference took place in June 2010 (except for st. 8), at the onset of rainy season, when the highest chlorophyll *a* and SRP concentrations were found. The second cluster was mostly for the the rainy season's peak (September, 2009). A third cluster was for the dry season (March, 2009), but only for the sampling points at the outer and middle portions. These same sampling points, but from the north winds season, formed a fourth cluster's sub-cluster; at the other sub-cluster the inner lagoon's sampling points intermingled for the north wind and dry seasons (Fig. 4).

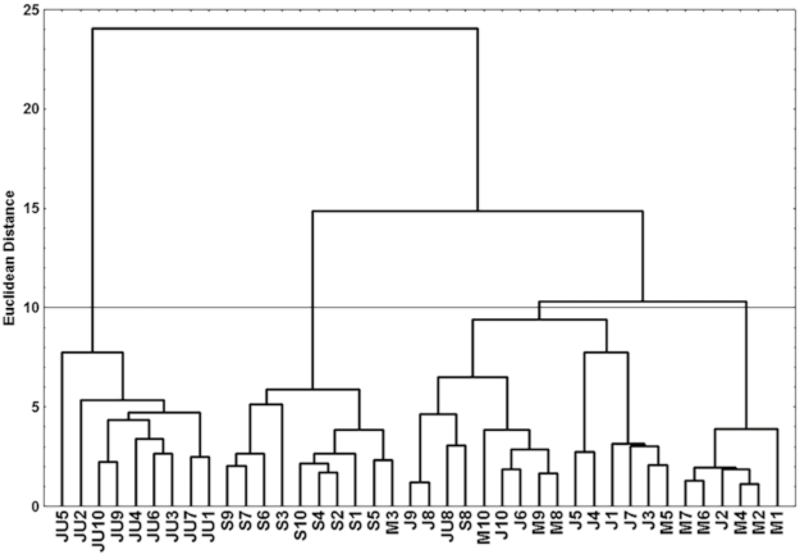

Figure 4. Dendrogram of the sampling points and surveys based on the *z*-values of the physical and chemical parameters

Ordination by principal component analysis (PCA) from water variables showed high salinity during the north wind season (January) and the early rainy season (June) at the negative part of the main axis. At the positive extreme ammonium from the dry season (March) appeared, TP from the dry and early rainy season and SRP from the north wind season (Fig. 5). Ammonia from the other surveys appeared at the negative part of the second axis along with oxygen at the early rainy season, SRP at the full rainy season, nitrate at the dry season and nitrite at the early rainy season. The positive part of this second axis relates to dissolved oxygen during the north wind season survey (January), chlorophyll at the full rainy season, TP at the north wind season and SRP at the early rainy season (Fig. 5).

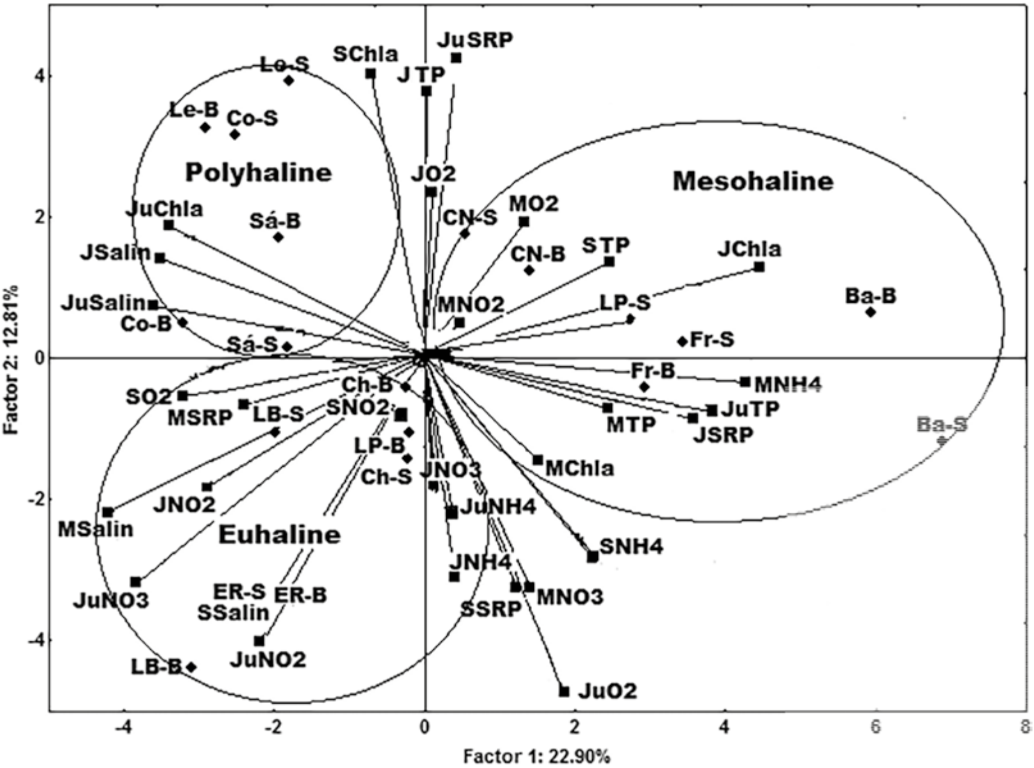

Figure 5. Principal component analysis. Biplot for water variables and sampling points. Sampling points are indicated with a small diamond. The suffix indicates surface (S) or bottom (B). Ba, Basura; Ch, Chancarral; Co, Cocal; CN, Costa Norte, ER, El Real; Fr, Fraile; LB, La Barra; Le, Levisa; LP; La, Palma; Sá, Sábalo. Variables, a small square at the end of a line. The prefix indicates the survey, M (March 2009), S (September 2009), J (January 2010), Ju (June 2010). Chla chlorophyll *a*, NH4 ammonium, NO2 nitrite, NO3 nitrate, SRP soluble reactive phosphate, TP total phosphorus, Salin salinity

Sampling points were clustered so that three zones could be distinguished: a mesohaline to oligohaline zone comprising Basura, Fraile and Costa Norte, at the positive part of the first ordination axis. The polyhaline zone comprised Levisa, Sábalo and Cocal, at the positive part of the second axis. The euhaline zone comprised La Barra, Chancarral and El Real, at the negative part of the second axis. La Palma stood as a group on its own, in fact the surface samples clustered to the mesohaline zone while the bottom samples clustered to the euhaline zone; this implies a stratification, contrary to what happened with the rest of the sampling points. This zonation is represented on a map at Figure 6.

Discussion

During these surveys, for all sampling points the water column in generally was well mixed, as confirmed by the statistical tests applied. In the horizontal axis, three sectors that correspond to those described by Yáñez-Arancibia (1987) were detected, and have also been found for other coastal lagoons, such as the Carretas-Pereyra, Chiapas system (Varona-Cordero & Gutiérrez-Mendieta, 2003), which had been previously described for Sontecomapan by Mondragón (2005) and Morán *et al.* (1995). Nevertheless, this is the average trend for the period covered by the present research, as there are seasonal variations all along these sectors such as the September survey, when due to heavy rainfall and runoff, salinity lower than 5 PSU was detected at La Boya, the nearest sampling point to sea communication, while euhaline salinities occurred at several points in the March survey. This trend in seasonal shifts along sectors has also been detected in other Mexican coastal lagoons, such as the Chantuto-Panzacola and Carretas-Pereyra systems in Chiapas (Varona-Cordero & Gutiérrez-Mendieta, 2003).

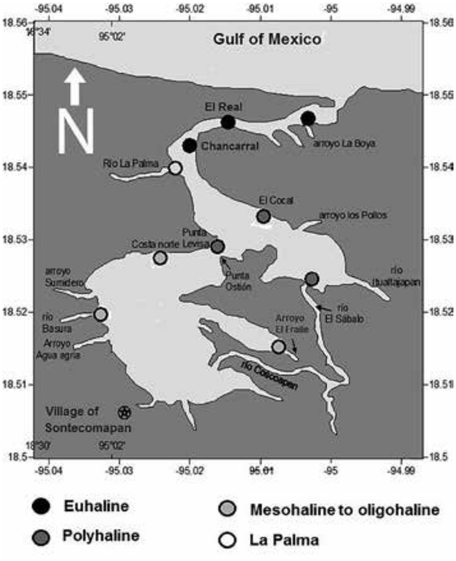

Figure 6. Map of the zones identified through the principal component analysis.

The stream at La Palma appeared different from the remaining sampling points. At this station, nitrite and nitrate attained their highest values during the September survey, probably arriving with the runoff from human settlements upstream. For SRP, peak values occurred during the June survey; during the rainy season offset, which also points to an upstream origin of SRP. What is remarkable is that phosphorus input was higher at the early rainy season while nitrogen input increased at the season peak. This differs from the trend detected by Aké-Castillo & Vázquez (2008), who applied canonical correspondence analysis (CCA) to their 2002 and 2003 Sontecomapan data and found that salinity positively correlated to SRP, and negatively to silicate, which implies that SRP was of marine origin, while silicate mainly came from rivers and streams; in the present case, SRP entered through stream La Palma as the PCA ordination showed either negative correlation (January) or lack of correlation (June) between SRP and salinity (Fig. 5).

At coastal lagoons nitrite comes from two sources, either through discharge to water from human activities, or through the oxidation of the ammonia that arises from excretion by ammonotelic organisms or from organic matter decay by ammonia-oxidizing bacteria. Nitrite can then follow one of two paths: it is either further oxidised to nitrate (nitrification) and becomes a nutrient for primary producers, or it enters the reductive path to be transformed into N_2 which is lost from the aquatic environment (denitrification), thus nitrite can be considered as a core chemical species in the nitrogen cycle (Ward, 1996; 2007). The path followed depends on the environment's oxidative or reductive conditions, and associated microbial assemblages as well. For the present case, it appears that part of the nitrate is derived from human sources and sediment associated to emergent macrophytes and mangrove at the lagoon's banks, an environment which is fit for ammonia oxidation. In Aké-Castillo & Vázquez' (2008) CCA ordination, a negative correlation appears between ammonium and nitrate, and ammonium locates close to the mangrove-influenced zones. In López Portillo *et al.* (2017) the principal component analysis also shows a negative correlation between nitrate or nitrite and ammonium at Sontecomapan. It is to be determined if these results mean that nitrite and nitrate originate from the ammonia oxidation released by organic matter decay at mangrove sediment.

Mondragón (2005) attempted a trophic characterisation of Sontecomapan by using Vollenwieder & Kerekes' (1980) probability plots for some parameters, where a relative trophic level is assigned to each value according to the probability for its occurrence, and applied them to two surveys, in February and June 2005. For February, he assigned a hypertrophic category to most of the sampling points, and the rest as eutrophic, based on TP values; while according to chlorophyll *a,* the sampling points fell into more categories, from oligotrophic to hypertrophic, and were centred in eutrophic. For June, TP values ranged from mesotrophic to hypertrophic, and the same occurred for chlorophyll *a*. In the present work, the TP values also fall in the range from mesotrophic to hypertrophic, while chlorophyll *a* ranges from oligotrophic to eutrophic. When comparing these results, the trend for the 2005 values was from eutrophic to hypertrophic, whereas the results reported here ranged from mesotrophic to eutrophic. However, by employing indexes devised for tropical or subtropical reservoirs (Salas & Martino, 1991; Fernandes-Cunha *et al.*, 2013), instead of Vollenwieder & Kerekes' (1980) index for temperate waters, for the considered period, the corresponding geometric means (0.202 to 0.386 mg L^{-1}) show that Sontecomapan was hypereutrophic based on TP, while it was oligotrophic for chlorophyll *a* (geometric means 0.40 to 2.08 mg m^{-3}). According to Salas & Martino (1991) and Fernandes-Cunha *et al.* (2013) the corresponding values for the eutrophic onset conditions are 0.078 mg L^{-1} for total phosphorus, and 27.2 mg m^{-3} for chlorophyll *a*. This points to the importance of not attempting the trophic characterization of a coastal lagoon on a sole variable basis.

In fact, a comparison of the chlorophyll *a* values reported for Sontecomapan since the early 90's, show that the 2009-2010 data are the lowest for the period (Table 5). Currently, we have no information that would allow us to determine if the system's productivity had diminished; nutrients were within the ranges reported for the same period, and SRP was at higher concentrations than in former surveys, so that primary production appears not to have been nutrient-limited. Another possible reason for the chlorophyll *a* and particulate organic matter (POM) diminishing was probably related to exceptional rainfall during this period, especially during the September survey, which probably caused a flushing out of suspended materials to the sea, as confirmed by salinities lesser than 5 PSU near the communication mouth. This agrees with the low zooplankton densities found by Benítez-Díaz-Mirón *et al.* (2014) in the same period, which the authors attribute to an increased water exchange, among other possible causes. On the other hand, Aké-Castillo & Vázquez (2008) and Muciño *et al.* (2011) performed surveys in 2002-2003 and 1999, respectively, and detected a phytoplankton flow in both ways, which depended on the tide and rainfall combined effect. Even if flushing protects coastal environments from pollution (Mitchell *et al.*, 2017; Newton & Mudge, 2003), whenever it becomes too intense, it might result in a loss of planktonic production because of its exportation to the sea.

Aké-Castillo & Vázquez (2008) found that folin phenol active substances (FPAS) released by mangrove leaf litter were positively correlated to the abundance of bloom-forming phytoplankton species, and that this occurred mainly in those zones with high mangrove influence during the dry season, correlating with high salinities. Our PCA ordination presents a contrasting situation, with lower salinities occurring at these sites; taking chlorophyll *a* as a phytoplankton density proxy, the highest densities were found at the rainy season onset (June) along the sampling points at the channel dividing line from the main lagoon body.

The point that Aké-Castillo & Vázquez (2008) remark the most from their results is that dominant phytoplankton species at Sontecomapan rank along a gradient stablished according to FPAS, as determined through CCA. This is important, as some of these are harmful algal bloom phytoplankton species (HAB) (Muciño *et al.*, 2011) and FPAS concentration values could be employed to predict the probabilities of HAB occurrence.

Concerning the use of multivariate methods for the characterization of aquatic ecosystems, they have been successfully employed, in some recent works, for the analysis of lotic, lentic, freshwater and coastal ecosystems, for the identification of spatial or seasonal patterns in their behaviour or else for the detection and assessment of human-induced perturbations extent. For example, Magyar *et al.* (2013) applied a multivariate approach to assess spatial changes at Neusiedler See, an Austrian lake. Wang *et al.* (2013), Duan *et al.* (2016), and Kim *et al.* (2016) used multivariate statistical techniques for determining water quality patterns in a river, a lake and a canal, while Wang *et al.* (2015) employed PCA for the characterization of the extent and sources of metal pollution in coastal areas. Liu *et al.* (2011) linked multivariate analysis to geostatistical methods while Tobiszewski *et al.* (2010) combined multivariate analysis with experts' opinion. Srichandan *et al.* (2015) employed this approach to the study of phytoplankton assemblages at a coastal lagoon. All these researchers agreed in considering these methods as an aid to the patterns and processes understanding in these systems, as well as a necessary step for the construction of predictive models. As an emerging use of multivariate methods, ordination of coastal lagoons has also been applied for identifying those areas related to the breeding and survival of vectors for diseases such as malaria, dengue and chikungunya (Sheela *et al*., 2015).

It is expected that the present characterization of Sontecomapan will serve as an aid for its proper management. Duck and Figuereido da Silva (2012) consider that there has been more of a coastal bodies mismanagement which has degraded rather than enhanced the ecosystem services derived from them. This has often resulted from a short sightedness, but also from insufficient knowledge of lagoon processes. At the moment, this becomes even more important, for it is necessary to forecast the effects of an eventual climate change.

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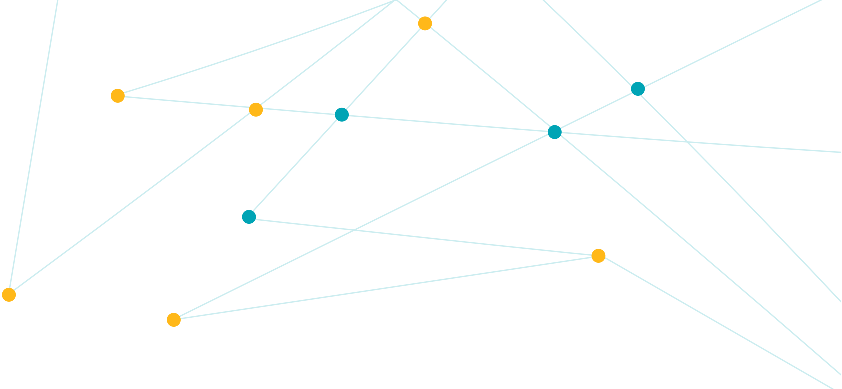

PART 2

Biota

Literature review and update checklist for the biota of Sontecomapan Lagoon

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ABSTRACT: This review aims to offer a published information synthesis in scientific papers and book chapters about the species of phytoplankton, plants, invertebrates and vertebrates which have been reported for the Sontecompan Lagoon. However, fish were not included in the present work except for those mentioned as hosts for parasites.[1] Here is the list of flora with 431 species. The list of invertebrates includes 201 species and the list of vertebrates contains 185 species.[2]

KEYWORDS: Phytoplankton, Plants, Invertebrates, Vertebrates.

Materials and Methods

Species checklists were done from sorting and analysing about 59 scientific articles and book chapters published between 1976 and 2015. The scientific names were updated

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[1] The complete list of fish of the Sontecopan Lagoon is included in other chapter of this book.

[2] The total number of vertebrate species will be higher considering the contributions of other chapters of this book related to fish.

according to specialized group taxonomic databases and for those groups where no specific databases were available, the updated names were retrieved from the Integrated Taxonomic Information System (ITIS, 2017). The online databases consulted include AlgaeBase (Guiry & Guiry, 2017), Rotifer World Catalog (Jersabek & Leitner, 2013), World of Copepods Database (Walter & Boxshall, 2017), World Register of Marine Species (WoRMS, 2017), Global Biodiversity Information Facility (GBIF, 2017), Worldwide Mollusc Species Database (Galli, 2016), the Informatic Unity for Biodiversity (UNIBIO, 2017), and The eBird/Clements checklist of birds of the world: v2016 (Clements *et al.*, 2016)

Results

1. Flora

Table 1. shows the list of all the vegetation species reported in the literature reviewed where 403 are phytoplankters,[3] 4 macroalgae, 1 seagrass, 3 mangrove, 7 species of coastal vegetation, 1 fern, 2 legumes, 1 liana, 2 bromeliads, and 7 orchids.

1.1. Phytoplankton

The first published paper about the Sontecomapan lagoon phytoplankton was that of Aké-Castillo *et al.* (1995), the authors studied the different species of *Skeletonema* which were obtained in bimonthly samples during 1991 and 1992. The authors reported the presence of *Skeletonema subsalsum*, *S. pseudocostatum* and *S. costatum* in the lagoon body and *Skeletonema tropicum* in the mouth. Only *S. pseudocostatum* appeared throughout the whole year; they also pointed out that salinity was the most important factor in the morphological variation of *S. pseudocostatum* and *S. costatum*. Other species found were *Bacteriastrum hyalinum*, *Chaetoceros affinis*, *C. peruvianus*, and *Dinophysis caudata*.

Guerra-Martínez & Lara-Villa (1996) reported *Ceratium furca* var *hircus* blooms, which is a typical marine dinoflagellate; they were registered in the dry seasons of 1991 and 1992 when salinity ranged from 5 to 30 ppt.

A list of 66 species of planktonic diatoms found until that time was given by Meave del Castillo & Lara-Villa (1997). The presence of *Skeletonema subsalsum* was found in samples obtained bimonthly from December, 1990 to November, 1992; this diatom was com-

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[3] In this contribution the authors summarized a list of 403 taxa and the number of species summarized in the Phytoplankton chapter of this book is 357, the difference could be because in this chapter different varieties of the same species were taken under consideration by the authors.

mon in samples taken in rainy and "Nortes" (June to February) seasons (Aké-Castillo & Meave del Castillo, 1997).

Blooms of *Ceratium furca* var *hircus* of an atypical form (larger in size and with shorter antapical horns) were found in May of 1991 and 1992, Guerra Martínez *et al.* (1997).

Aké-Castillo *et al.* (2000) detected six different forms of *Chaetoceros* with less than four setae per cell belonging to four different taxa: *Chaetoceros minimus*, *C. subtilis* var *abnormis* f. abnormis, *C. subtilis* var *abnormis* f. *simplex* and *C. throndsenii* var *trisetosa*.

Later on, 5 taxa of *Chaetoceros* (*Chaetoceros minimus*, *C. subtilis* var *abnormis* f. *abnormis*, *C. subtilis* var *abnormis* f. *simplex*, *C. throndsenii* var *throndsenia*, and *C. throndsenii* var *trisetosa*) featuring less than 4 setae, were reported by Aké-Castillo *et al.* (2004). All five species were found in the dry season and only *C. subtilis* var *abnormis* f. *simplex* was also found in the rainy season. All the species occurred at salinities ranging from 3 to 7.4 ppt, and none of them was present at salinities ranging from 9 to 38 ppt.

In a study from October 2002 to October 2003 (Aké-Castillo & Vázquez 2008), the authors reported 179 phytoplankton taxa and established their relationship with some variables such as the chemical compounds originated by the mangrove decomposition, nutrients, salinity and temperature. The composition, distribution and abundance of the thecate dinoflagellates of Sontecomapan Lagoon were determined in October of 1999 and March 2000; 34 were identified (Figueroa-Torres *et al.*, 2009).

Muciño-Márquez *et al.* (2011a) identified 177 phytoplankton species in samples obtained from a nycthemeral sampling carried out at a station located at the mouth of the lagoon.

A list of 27 phytoplankton species wich form toxic algal blooms (HABs) were identified in a nycthemeral sampling done from October 27 to 28,1999 by Muciño-Márquez *et al.* (2011b).

The distribution and abundance of species for the genus *Prorocentrum* was determined from a nycthemeral sampling carried out at a station located at the mouth of the lagoon. Five species were found: *Prorocentrum compressum*, *P. gracile*, *P. micans*, *P. mexicanum* and *P. robustum* (Muciño-Márquez *et al.*, 2011c).

Aké-Castillo & Vázquez (2011) determined a new variety of *Peridinium quinquecorne* which was called *Peridinium quinquecorne* var *trispiniferum* since *P. quinquecorne* characteristically bears 4 conspicuous spines on the hypotheca and the organisms found in Sontecomapan Lagoon, during samplings made in 1999, 2001, 2003 and 2007. This featured was consistently observed in the four years.

Aké-Castillo (2015) reported the morphological and autoecological description of *Thalassiosira cedarkeyensis* found in Sontecomapan Lagoon. Other species mentioned in this work were: *Prorocentrum cordatum* (Ostenfeld) Dodge (= *P. minimum* (Pavillard) J. Schiller) and *Ceratium furca* var *hircus* (Schöder) Margalef ex Sournia.

1.2. Macroalgae and seagrasses

González-Gutiérrez (1977) carried out an annual study on the distribution and growth dynamics of *Ruppia maritima* based on monthly samplings from April 1974. The author found that the *R. maritima* patches occupied different lagoon sites and showed a remarkable asynchrony in their development phases.

The presence of metals in the alga *Cladophoris membranacea* (now *Cladophora membranacea*) and in the marine phanerogam *Ruppia maritima* was determined by González-Fierro *et al.* (1994).

Collado-Vides & West (1996) reported the presence of *Bostrychia calliptera* in a study about mangrove-associated microflora. This occurrence was a new record of the species in the Gulf of Mexico. The studied material was collected on the roots of *Rhizophora mangle* and the pneumatophores of *Avicennia germinans* in the Sontecomapan Lagoon.

In 2008 Calva & Torres-Alvarado reported the bioaccumulation of polycyclic aromatic hydrocarbons (PAHs) in two species of algae (*Ulva intestinalis* and *Bostrychia pinnata*) and the seagrass (*Ruppia maritima*) collected in different sampling points of the lagoon; PAHs levels in *R. maritima* were higher than those determined in the macroalgae. The authors pointed out that one of the reasons for this is that the PAHs are lipophilic and the *Ruppia maritima* has more fatty acids in its structure than the macroalgae.

1.3. Mangrove, trees and coastal vegetation

Medina *et al.* (1995) studied the mineral content in foliar tissues of mangrove and other coastal vegetation of Sontecomapan Lagoon. They worked with the halophilic species *Rhizophora mangle* and *Laguncularia racemosa*, as well as the facultative halophyte *Acrostichum aureum* and the non-halophyte *Pachira aquatica*. The results revealed that salinity had a significative influence on the minerals incorporation on the studied species. On the other hand, Ángeles-Álvarez (1997a) pointed out that the highest *Pachira aquatica* individuals can reach the 25 meters (way over the reported maximum height of 18 m for this species), and they were found in the spring called “La Fosa de los Enanos”; likewise, the *Rhizophora mangle* individuals in the lagoon can reach a height of up to 30 m, making them the tallest in the Gulf of Mexico coast (Ángeles-Álvarez, 1997b).

Carmona-Díaz *et al.* (2004) mentioned the mangrove presence of *Rhizophora mangle*, *Avicennia germinans* and *Laguncularia racemosa*, *Cycnoches ventricosum* orchids, *Chysis bractescens*, *Brassavola nodosa*, *Encyclia alata*, *E. belizensis*, *Epidendrum nocturnum* and *E. ciliare*, bromeliad *Tillandsia concolor* and other trees such as *Tetrorchidium rotundatum* and *Andira galeottiana* in the management plan for the Sontecomapan mangroves.

In a study related with the composition of aliphatic hydrocarbons in sediments from Sontecomapan Lagoon Calva *et al.*, 2005 pointed out the presence of the *Eichhornia crassipes* aquatic lily and the *Thypa dominguensis* tulip at the mouth of the Basura and Sumidero rivers and the existence of areas with *Ruppia maritima*, *Rhizophora mangle*, *Laguncularia racemosa* and *Avicennia germinans*.

Aké-Castillo *et al.* (2006) published the results of a study on litter production and decomposition of *Rhizophora mangle* which was carried out from October of 2002 to October of 2003. The authors found that in the mangrove decomposition, after 60 days there were important differences between climatic seasons and that the factors affecting leaf decay were different between seasons.

Capparidastrum tuxtlense (Capparaceae) was reported as new species by Cornejo & Iltis (2010). The authors mentioned that it is an endemic species restricted to the low eastern portion of the San Martin volcano in the State of Veracruz. The habitat and distribution of *C. tuxtlense* is near sea level ans it was found in Playa Escondida, the Sontecomapan Lagoon mouth and on the Basura riverside.

A paper was published in 2012 about the evaluation of asexual propagation by aerial shoots on the mangrove species of the Sontecomapan Lagoon, *Rhizophora mangle*, *Laguncularia racemosa* and *Avicennia germinans* (Hernández-Carmona *et al.* 2012) where *Laguncularia racemosa* gave the best results on the number and timing for roots production, therefore the authors recommended it for reforesting if the asexual propagation by aerial shoots is to be used.

Sousa & Sotuyo (2012) carried out a phylogenetic analysis of the genus *Muellera*. Two of the studied species were *Muellera frutescens* and *Muellera unifoliolata*. Some aditional materials used in the study were collected from different places of the Sontecomapan Lagoon. *M. frutescens* was obtained in Punta Ostión, Punta Levisa, a site about 6 Km from the pier, Isla Coscoapan and Rancho Levisa (at the edge of the lagoon), whereas the origin of *M. unifoliolata* was the lagoon bar.

Hernández-Carmona *et al.* (2014) published an evaluation about the structure, distribution and abundance of the liana or climbing plant *Rhabdadenia biflora*. The results showed that *R. biflora* preferred *Rhizophora mangle* as host species instead of *Avicennia germinans* and *Laguncularia racemosa*. The authors also indicated the swamp fern *Acrostichum aureum* and the shrub *Inga* sp. as hosts.

Carmona-Díaz *et al.* (2014) reported that the epiphytic bromeliad of mangrove *Aechmea bracteata* did not show preference for any of the mangrove species present at the studied area.

Table 1. Flora species reported for the Sontecomapan Lagoon, Veracruz

	Genus	Species	variety	References
Phytoplankton				
	Achnanthes cf. curvirostrum J. Brun, 1895			Aké-Castillo & Vázquez (2008)
	Achnanthes exigua Grunow, 1880	*exigua* (Grunow, 1880)		Aké-Castillo & Vázquez (2008)
	Achnanthes	*lanceolata* (Brébisson ex Kützing) (Grunow, 1880)		Muciño-Márquez *et al.* (2011a)
	Achnanthes	*sublaevis* (Hustedt, 1937)		Aké-Castillo & Vázquez (2008)
	Achnanthes	sp.		Aké-Castillo & Vázquez (2008)
	Achradina	sp.		Muciño-Márquez *et al.* (2011a)
	Actinastrum	*hantzschii* (Lagerheim, 1882)		Muciño-Márquez *et al.* (2011a)
	Akashiwo	*sanguinea* (K.Hirasaka) (G.Hansen & Ø.Moestrup, 2000)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Amphidinium	*sphenoides* (Wulff, 1916)		Aké-Castillo & Vázquez (2008)
	Amphidinium	sp. 1 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Amphipleura	sp. 2 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Amphiprora	*paludosa* (W. Smith, 1853)		Muciño-Márquez *et al.* (2011a)
	Amphora	*angusta*	*eulensteinii* (Grunow) Cleve	Muciño-Márquez *et al.* (2011a)
	Amphora	*clara* (A. Schmidt, 1875)		Muciño-Márquez *et al.* (2011a)
	Amphora	*ovalis*	*pediculus* Kützing, 1844	Muciño-Márquez *et al.* (2011a)
	Amphora	*proteus* (Gregory, 1857)		Muciño-Márquez *et al.* (2011a)
	Amphora	sp.		Aké-Castillo & Vázquez (2008)
	Amphora	sp.		Muciño-Márquez *et al.* (2011a)
	Asterionellopsis	*glacialis* (Castracane) (Round, 1990)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Aulodiscus	sp.		Muciño-Márquez *et al.* (2011a)
	Azpeitia	*nodulifera* (A.W. F. Schmidt) (G. A. Fryxell & P. A. Sims, 1986)		Aké-Castillo & Vázquez (2008)
	Bacillaria	*paxillifera* (O.F.Müller) (T. Marsson, 1901)		Aké-Castillo & Vázquez (2008)
	Bacillaria	*paxillifer* (O. F. Müll.) Hendy		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Bacteriastrum	*elegans* (Pavillard)		Aké-Castillo & Vázquez (2008)
	Bacteriastrum	*elongatum* (Cleve, 1897)		Aké-Castillo & Vázquez (2008)
	Bacteriastrum	*delicatulum* (Cleve, 1897)		Muciño-Márquez *et al.* (2011a)
	Bacteriastrum	*hyalinum* (Lauder, 1864)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo *et al.* (1995)
	Bacteriastrum	*hyalinum*	*princeps* (Castrachane) Ikari	Muciño-Márquez *et al.* (2011a)
	Bacteriastrum	sp.		Muciño-Márquez *et al.* (2011a)
	Bellerochea	sp.		Meave del Castillo & Lara-Villa (1997)
	Biddulphia	*biddulphiana* (J. E. Smith) (Boyer, 1900)		Muciño-Márquez *et al.* (2011a)
	Biddulphia	*pulchella* (S. F. Gray, 1821)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Biddulphia	sp.		Aké-Castillo & Vázquez (2008)
	Biddulphia	sp.		Muciño-Márquez *et al.* (2011a)
	Biddulphia	sp. 1 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Biddulphia	sp. 2 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Caloneis	sp.		Aké-Castillo & Vázquez (2008)
	Campylodiscus	*echeneis* (Ehrenberg ex Kützing, 1844)		Meave del Castillo & Lara-Villa (1997)
	Campylodiscus	sp.		Meave del Castillo & Lara-Villa (1997)
	Cerataulina	*bergonii* (H.Peragallo) (Schütt, 1896)		Meave del Castillo & Lara-Villa (1997)
	Ceratium	*breve*	*parallelum* (Schmidt) Jörgensen	Figueroa-Torres *et al.* (2009)
	Ceratium	*deflexum* (Kofoid) (Jörgensen)		Figueroa-Torres *et al.* (2009)
	Ceratium	*furca*	*furca* (Ehrenberg) Claparède et Lachmann	Figueroa-Torres *et al.* (2009)
	Ceratium	*furca*	*hircus* (Schröder) Margalef ex Sournia, 1973	Guerra-Martínez & Lara-Villa (1996); Guerra-Martínez & Meave del Castillo (1997); Aké-Castillo & Vázquez (2008); Figueroa-Torres *et al.* (2009); Aké-Castillo (2015)
	Ceratium	*furca* (Ehrenberg) (Claparède & Lachmann, 1859)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Ceratium	*fusus* (Ehrenberg) (Dujardin, 1841)		Figueroa-Torres *et al.* (2009); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Ceratium	*horridum*	*buceros* (Zacharias) Sournia	Figueroa-Torres *et al.* (2009)
	Ceratium	*lunula* (Schimper ex Karsten) (Jörgensen)		Figueroa-Torres *et al.* (2009)
	Ceratium	*massiliense*	*f. armatum* (Karsten) Jörgensen	Figueroa-Torres *et al.* (2009)
	Ceratium	teres Kofoid		Figueroa-Torres *et al.* (2009)
	Ceratium	*tripos* (O. F. Müller) (Nitzsch, 1817)		Muciño-Márquez *et al.* (2011a)
	Ceratium	*tripos*	*atlanticum* (Ostenfeld) Paulsen	Figueroa-Torres *et al.* (2009)
	Ceratium	sp.		Muciño-Márquez *et al.* (2011a)
	Chaetoceros	*affinis* (Lauder, 1864)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo *et al.* (1995); Aké-Castillo & Vázquez (2008)
	Chaetoceros	*atlanticus* (Cleve, 1873)		Muciño-Márquez *et al.* (2011a)
	Chaetoceros	*debilis* (Cleve, 1894)		Muciño-Márquez *et al.* (2011a)
	Chaetoceros	sp.		Muciño-Márquez *et al.* (2011a)
	Chaetoceros	cf. *atlanticum*		Muciño-Márquez *et al.* (2011b)
	Chaetoceros	*compressus* (Lauder, 1864)		Meave del Castillo & Lara-Villa (1997)
	Chaetoceros	*curvisetus* (Cleve, 1889)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Chaetoceros	*danicus* (Cleve, 1889)		Aké-Castillo & Vázquez (2008)
	Chaetoceros	cf. *debilis*		Muciño-Márquez *et al.* (2011b)
	Chaetoceros	*denticulatus* Lauder		Meave del Castillo & Lara-Villa (1997)
	Chaetoceros	*diversus* (Cleve, 1873)		Aké-Castillo & Vázquez (2008)
	Chaetoceros	*heterovalvatus* Prosckina-Lavrenko		Aké-Castillo & Vázquez (2008)
	Chaetoceros	*holsaticus* (F.Schütt, 1895)		Aké-Castillo & Vázquez (2008)
	Chaetoceros	*laciniosus* (F.Schütt, 1895)		Aké-Castillo & Vázquez (2008)
	Chaetoceros	*lorenzianus* (Grunow, 1863)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Chaetoceros	*minimus* (Levander) (D. Marino, G. Giuffre, M. Montresor & A. Zingone, 1991)		Aké-Castillo *et al.* (2000); Aké-Castillo *et al.* (2004)
	Chaetoceros	*mulleri*	subsalsum (Lemmermann) Johansen et Rushforth	Aké-Castillo & Vázquez (2008)
	Chaetoceros	*peruvianus* (Brightwell, 1856)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo *et al.* (1995); Aké-Castillo & Vázquez (2008)
	Chaetoceros	*simplex* (Ostenfeld, 1902)		Aké-Castillo & Vázquez (2008)
	Chaetoceros	*subtilis*	*abnormis f. abnormis* Prosckina-Lavrenko, 1961	Aké-Castillo *et al.* (2000); Aké-Castillo *et al.* (2004); Aké-Castillo & Vázquez (2008)
	Chaetoceros	*subtilis*	*abnormis f. simplex* Prosckina-Lavrenko, 1961	Aké-Castillo *et al.* (2000); Aké-Castillo *et al.* (2004); Aké-Castillo & Vázquez (2008)
	Chaetoceros	*throndsenii*	*throndsenia* (Marino, Montresor & Zingone) Marino, Montresor & Zingone	Aké-Castillo *et al.* (2004); Aké-Castillo & Vázquez (2008)
	Chaetoceros	*throndsenii*	trisetosa Zingone, 1991	Aké-Castillo *et al.* (2000); Aké-Castillo *et al.* (2004); Aké-Castillo & Vázquez (2008)
	Chaetoceros	aff. *muelleri* Lemmermann		Meave del Castillo & Lara-Villa (1997)
	Chaetoceros	aff. *subtilis* Cleve		Meave del Castillo & Lara-Villa (1997)
	Chatonella	sp.		Muciño-Márquez *et al.* (2011a)
	Chlamydomonas	sp.		Aké-Castillo & Vázquez (2008)
	Chroccocus	sp.		Aké-Castillo & Vázquez (2008)
	Closterium	sp.		Aké-Castillo & Vázquez (2008)
	Cocconeis	*disculoides* (Hustedt, 1955)		Muciño-Márquez *et al.* (2011a)
	Cocconeis	*placentula*	*placentula* Ehrenberg	Muciño-Márquez *et al.* (2011a)
	Cocconeis	*scutellum* (Ehrenberg, 1838)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Cocconeis.	sp.		Muciño-Márquez *et al.* (2011a)
	Corethron	*hystrix* (Hensen, 1887)		Meave del Castillo & Lara-Villa (1997)
	Coscinodiscus	*centralis* (Ehrenberg, 1844)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Coscinodiscus	*concinnus* (W.Smith, 1856)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Coscinodiscus	*curvatulus* Grunow ex (A. Schmidt, 1878)		Muciño-Márquez *et al.* (2011a)
	Coscinodiscus	*granii* (Gough, 1905)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Coscinodiscus	*radiatus* (Ehrenberg, 1840)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Coscinodiscus	*perforatus*	*pavillardi* (Forti) Hustedt, 1928	Meave del Castillo & Lara-Villa (1997)
	Coscinodiscus	sp.		Muciño-Márquez *et al.* (2011a)
	Cosmarium	sp. 1 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Cosmarium	sp. 2 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Cuneolus	aff. *skvortzowii* (Nikolaev) Medlin		Meave del Castillo & Lara-Villa (1997)
	Cyclotella	cf. *austriaca* (M. Peragallo) Hustedt		Aké-Castillo & Vázquez (2008)
	Cyclotella	*cryptica* (Reimann, Lewin & Guillard)		Aké-Castillo & Vázquez (2008)
	Cyclotella	*meneghiniana* (Kützing, 1844)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Cyclotella	*striata* (Kützing) (Grunow, 1880)		Aké-Castillo & Vázquez (2008)
	Cyclotella	sp. 1 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Cyclotella	sp. 2 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Cylindrotheca	*closterium* (Ehrenberg) (Reimann & J.C.Lewin, 1964)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Cymatopleura	sp.		Aké-Castillo & Vázquez (2008)
	Cymatopleura	*solea* (Brébisson) (W. Smith, 1851)		Muciño-Márquez *et al.* (2011a)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Cymatosira	sp.		Aké-Castillo & Vázquez (2008)
	Cymbella	sp.		Aké-Castillo & Vázquez (2008)
	Cymbella	*minuta*	*pseudogracilis* (Cholnoky) Reimer, 1975	Muciño-Márquez *et al.* (2011a)
	Denticula	*elegans* (Kützing, 1844)		Muciño-Márquez *et al.* (2011a)
	Detonula	*pumila* (Castracane) (Gran, 1900)		Meave del Castillo & Lara-Villa (1997)
	Diatomea	sp.		Muciño-Márquez *et al.* (2011a)
	Dictyocha	*fibula* (Ehrenberg, 1839)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Dictyocha	sp.		Muciño-Márquez *et al.* (2011a)
	Dinophysis	*caudata* (Saville-Kent)		Aké-Castillo *et al.* (1995); Aké-Castillo & Vázquez (2008); Figueroa-Torres *et al.* (2009); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Dinophysis	*rotundata* (Claparède et Lachmann)		Figueroa-Torres *et al.* (2009)
	Dinophysis	sp.		Muciño-Márquez *et al.* (2011a)
	Diploneis	*bombus* (Ehrenberg) (Ehrenberg, 1853)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Diploneis	*ovalis* (Hilse) (Cleve, 1891)		Muciño-Márquez *et al.* (2011a)
	Diploneis	sp.		Meave del Castillo & Lara-Villa (1997)
	Diploneis	sp.		Aké-Castillo & Vázquez (2008)
	Diploneis	sp.		Muciño-Márquez *et al.* (2011a)
	Diplopsalis	sp.		Muciño-Márquez *et al.* (2011a)
	Ditylum	*brightwellii* (T.West) (Grunow, 1885)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Entomoneis	*alata* (Ehrenberg) (Ehrenberg, 1845)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Eucampia	*zodiacus* (Ehrenberg)		Aké-Castillo & Vázquez (2008)
	Euglena	*polymorpha* (P. A. Dangeard, 1902)		Muciño-Márquez *et al.* (2011a)
	Euglena	*stellata* (Mainx, 1926)		Muciño-Márquez *et al.* (2011a)
	Euglena	cf. *stellata*		Muciño-Márquez *et al.* (2011b)
	Euglena	*viridis* (O. F. Müller) (Ehrenberg, 1830)		Muciño-Márquez *et al.* (2011a)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Euglena	cf *viridis*		Muciño-Márquez *et al.* (2011b)
	Euglena	sp.		Muciño-Márquez *et al.* (2011a)
	Euglena	sp. 1 *sensu* Muciño-(Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Euglena	sp. 2 *sensu* Muciño-(Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Fallacia	*pygmaea* (Kützing) (A. J. Stickle & D. G. Mann, 1990)		Muciño-Márquez *et al.* (2011a)
	Fragilaria	*capucina*	*capucina*	Muciño-Márquez *et al.* (2011a)
	Fragilaria	*construens* (Ehrenberg) (Grunow, 1862)		Muciño-Márquez *et al.* (2011a)
	Fragilaria	*construens* f. construens (Ehrenberg)		Aké-Castillo & Vázquez (2008)
	Fragilaria	*exigua* (Grunow, 1878)		Muciño-Márquez *et al.* (2011a)
	Fragilaria	aff. *fasciculata*		Meave del Castillo & Lara-Villa (1997)
	Fragilaria	*parasitica*	*parasitica*	Muciño-Márquez *et al.* (2011a)
	Fragilaria	*pinnata* (Ehrenberg, 1843)		Muciño-Márquez *et al.* (2011a)
	Fragilaria	*tenuicollis* (Heib., 1863)		Muciño-Márquez *et al.* (2011a)
	Fragilaria	*ulna*	*acus* (Kützing) Lange-Bertalot, 1980	Muciño-Márquez *et al.* (2011a)
	Fragilaria	*ulna*	*goulardii* (Brébisson) Lange-Bertalot, 1980	Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Fragilaria	*ulna*	*ulna* (Nitzsch) Lange-Bertalot	Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Fragilaria	*ulna*	*ungeriana* (Grun.) Lange-Bertalot	Meave del Castillo & Lara-Villa (1997)
	Fragilaria	sp.		Aké-Castillo & Vázquez (2008)
	Fragilaria	sp. 1 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Fragilaria	sp. 2 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Fragilaria	sp.		Muciño-Márquez *et al.* (2011a)
	Fragilaria	sp. 1 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Fragilaria	sp. 2 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Fragilaria	sp. 3 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Fragilaria	sp. 4 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Glenodinium	*gymnodinium* (Penard, 1891)		Muciño-Márquez *et al.* (2011a)
	Glenodinium	*danicum* (Paulsen, 1907)		Muciño-Márquez *et al.* (2011a)
	Glenodinium	sp.		Muciño-Márquez *et al.* (2011a)
	Gomphoneis	*herculeana*	*robusta* (Grunow) Cleve, 1894	Muciño-Márquez *et al.* (2011a)
	Gomphoneis	*olivacea* (Lyngbye) (P. A. Dawson, 1974)		Aké-Castillo & Vázquez (2008)
	Gomphonema	*parvulum* (Kützing) (Kützing, 1849)		Aké-Castillo & Vázquez (2008)
	Goniodoma	*sphaericum* (Murray & Whitting, 1899)		Muciño-Márquez *et al.* (2011a)
	Gonyaulax	*digitalis* (Pouchet) Kofoid		Figueroa-Torres *et al.* (2009); Aké-Castillo & Vázquez (2008)
	Gonyaulax	*polygramma* Stein		Figueroa-Torres *et al.* (2009)
	Gonyaulax	*spinifera* (Claparède et Lachmann) Diesing		Figueroa-Torres *et al.* (2009)
	Gonyaulax	*turbynei* (Murray & Whitting, 1899)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Guinardia	*flaccida* (Castracane) (H. Peragallo, 1892)		Aké-Castillo & Vázquez (2008)
	Guinardia	*striata* (Stolterfoth) (Hasle, 1996)		Aké-Castillo & Vázquez (2008)
	Gymnodinium	sp.		Aké-Castillo & Vázquez (2008)
	Gymnodinium	sp.		Muciño-Márquez *et al.* (2011a)
	Gymnodinium	sp. 1 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Gyrodinium	*fusiforme* (Kofoid & Swezy, 1921)		Muciño-Márquez *et al.* (2011a)
	Gyrodinium	*spirale* (Bergh) (Kofoid & Swezy, 1921)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Gyrosigma	*attenuatum* (Kützing) (Rabenhorst, 1853)		Muciño-Márquez *et al.* (2011a)
	Gyrosigma	*balticum* (Ehrenberg) (Rabenhorst, 1853)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Gyrosigma	*distortum* (W.Smith) (Griffith & Henfrey, 1856)		Meave del Castillo & Lara-Villa (1997)
	Gyrosigma	*fasciola* (Ehrenberg) (J. W. Griffith & Henfrey, 1856)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Gyrosigma	*robustum* (Grunow) (Cleve, 1894)		Aké-Castillo & Vázquez (2008)
	Gyrosigma	cf. *terryanum* (H. Peragallo) Cleve		Aké-Castillo & Vázquez (2008)
	Gyrosigma	sp.		Aké-Castillo & Vázquez (2008)
	Gyrosigma	sp. 1 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Gyrosigma	sp. 2 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Hantzschia	*amphioxys* (Ehrenberg) (Grunow, 1880)		Muciño-Márquez *et al.* (2011a)
	Hantzschia	sp.		Meave del Castillo & Lara-Villa (1997)
	Hemiaulus	*sinensis* (Greville, 1865)		Aké-Castillo & Vázquez (2008)
	Heterocapsa	sp.		Aké-Castillo & Vázquez (2008)
	Hyalodiscus	sp. 1 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Hyalodiscus	sp. 3 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Hyalodiscus	sp.		Aké-Castillo & Vázquez (2008)
	Johannesbaptistia	*pellucida* (Dickie) (W. R.Taylor & Drouet, 1938)		Muciño-Márquez *et al.* (2011a)
	Komvophoron	sp.		Aké-Castillo & Vázquez (2008)
	Lauderia	*borealis* (Gran, 1900)		Meave del Castillo & Lara-Villa (1997)
	Lepocinclis	*acus* (Ehrenberg)		Aké-Castillo & Vázquez (2008)
	Leptocylindrus	*danicus* (Cleve, 1889)		Aké-Castillo & Vázquez (2008)
	Licmophora	sp.		Aké-Castillo & Vázquez (2008)
	Lithodesmium	*undulatum* (Ehrenberg, 1839)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Luticula	*mutica* (Kützing) D. G. Mann		Aké-Castillo & Vázquez (2008)
	Luticula	sp.		Aké-Castillo & Vázquez (2008)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Lyrella	*impercepta* (Hustedt) Moreno		Aké-Castillo & Vázquez (2008)
	Lyrella	sp.		Aké-Castillo & Vázquez (2008)
	Mastogloia	*pusilla*	*lineatus*	Muciño-Márquez *et al.* (2011a)
	Melosira	*nummuloides* (C. Agardh, 1824)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Merismopedia	*tenuissima* Lemmermann		Aké-Castillo & Vázquez (2008)
	Merismopedia	*elegans* (A.Braun ex Kützing, 1849)		Muciño-Márquez *et al.* (2011a)
	Merismopedia	*glauca* (Ehr.) Naegeli		Muciño-Márquez *et al.* (2011a)
	Merismopedia	sp.		Muciño-Márquez *et al.* (2011a)
	Minidiscus	*comicus* (Takano, 1981)		Aké-Castillo & Vázquez (2008)
	Mougeotia	sp.		Aké-Castillo & Vázquez (2008)
	Navicula	*americana* (Ehrenberg, 1843)		Muciño-Márquez *et al.* (2011a)
	Navicula	*cryptocephala* (Kützing, 1844)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Navicula	*cryptocephala*	*subsalina* Hustedt, 1925	Muciño-Márquez *et al.* (2011a)
	Navicula	*cuspidata* (Kutzing) Kutzing, 1844		Muciño-Márquez *et al.* (2011a)
	Navicula	*disparilis*	*stigmata*	Muciño-Márquez *et al.* (2011a)
	Navicula	*gastrum* (Ehrenberg) (Kützing, 1844)		Muciño-Márquez *et al.* (2011a)
	Navicula	*gottlandica* Grunow, 1880		Muciño-Márquez *et al.* (2011a)
	Navicula	cf. humerosa		Muciño-Márquez *et al.* (2011a)
	Navicula	*inconspicua* (Gregory, 1857)		Muciño-Márquez *et al.* (2011a)
	Navicula	*minnewaukonensis* (Elmore, 1921)		Muciño-Márquez *et al.* (2011a)
	Navicula	*ludloviana* (Schmidt, 1876)		Muciño-Márquez *et al.* (2011a)
	Navicula	*mutica* (Kützing, 1844)		Muciño-Márquez *et al.* (2011a)
	Navicula	*placentula* Ehrenberg) (Kützing, 1844)		Muciño-Márquez *et al.* (2011a)
	Navicula	*platyventris* Meister		Muciño-Márquez *et al.* (2011a)
	Navicula	*poconoensis* (Patrick, 1945)		Muciño-Márquez *et al.* (2011a)
	Navicula	*radiosa*	*tenella* (Brébisson) Cleve & Möller, 1879	Muciño-Márquez *et al.* (2011a)
	Navicula	*rhynchocephala* (Kützing, 1844)		Muciño-Márquez *et al.* (2011a)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Navicula	*salinarum*	*intermedia* (Grunow) Cleve, 1895	Muciño-Márquez *et al.* (2011a)
	Navicula	*soehrensis* (Krasske, 1923)		Aké-Castillo & Vázquez (2008)
	Navicula	*pelliculosa* (Kützing) (Hilse, 1863)		Aké-Castillo & Vázquez (2008)
	Navicula	cf. *pennata* (Schmidt in Schmidt)		Aké-Castillo & Vázquez (2008)
	Navicula	cf. *peregrina* (Ehrenberg) (Kützing, 1844)		Aké-Castillo & Vázquez (2008)
	Navicula	cf. *takoradiensis* Hendey		Aké-Castillo & Vázquez (2008)
	Navicula	*tuscula* (Ehrenberg, 1840)		Muciño-Márquez *et al.* (2011a)
	Navicula	sp. 1 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Navicula	sp. 2 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Navicula	sp.		Muciño-Márquez *et al.* (2011a)
	Navicula	sp. 1 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Navicula	sp. 2 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Neidium	cf. *iridis* (Ehr.) Cleve		Aké-Castillo & Vázquez (2008)
	Neidium	sp.		Aké-Castillo & Vázquez (2008)
	Neocalyptrella	*robusta* (G.Norman ex Ralfs) (Hernández-Becerril & Meave del Castillo, 1997)		Aké-Castillo & Vázquez (2008)
	Neostreptotheca	*subindica* (von Stosch, 1977)		Meave del Castillo & Lara-Villa (1997)
	Nitzschia	*amphibia* (Grunow, 1862)		Aké-Castillo & Vázquez (2008)
	Nitzschia	*constricta* (Kützing) (Ralfs, 1861)		Aké-Castillo & Vázquez (2008)
	Nitzschia	*frustulum* (Kützing) (Grunow, 1880)		Aké-Castillo & Vázquez (2008)
	Nitzschia	*hantzschiana* (Rabenhorst, 1860)		Aké-Castillo & Vázquez (2008)
	Nitzschia	*vidovichii* (Grunow)		Muciño-Márquez *et al.* (2011a)
	Nitzschia	sp.		Muciño-Márquez *et al.* (2011a)
	Nitzschia	*longissima* (Brébisson) (Ralfs, 1861)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011b)
	Nitzschia	*ovalis* (H. J. Arnott, 1880)		Aké-Castillo & Vázquez (2008)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Nitzschia	*palea* (Kützing) (W. Smith, 1856)		Aké-Castillo & Vázquez (2008)
	Nitzschia	*sigma* (Kützing) (W. Smith, 1853)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Nitzschia	*triblyonella* (Hantzsch, 1860)		Aké-Castillo & Vázquez (2008)
	Nitzschia	cf. *linearis* (W. Smith)		Aké-Castillo & Vázquez (2008)
	Nitzschia	sp.		Meave del Castillo & Lara-Villa (1997)
	Nitzschia	sp.		Aké-Castillo & Vázquez (2008)
	Nitzschia	*angularis* (W. Smith, 1853)		Muciño-Márquez *et al.* (2011a)
	Nitzschia	*closterium* (Ehrenberg) (W. Smith, 1853)		Muciño-Márquez *et al.* (2011a)
	Nitzschia	*compressa* (Bailey) (C. S. Boyer, 1916)		Muciño-Márquez *et al.* (2011a)
	Noctiluca	*scintillans* (Macartney) (Kofoid & Swezy, 1921)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Nostoc	sp.		Aké-Castillo & Vázquez (2008)
	Ocystis	sp.		Aké-Castillo & Vázquez (2008)
	Odontella	*longicruris* (Greville) (M. A. Hoban, 1983)		Aké-Castillo & Vázquez (2008)
	Odontella	*mobiliensis* (J.W.Bailey) (Grunow, 1884)		Meave del Castillo & Lara-Villa (1997); Muciño-Márquez *et al.* (2011a)
	Odontella	*regia* (Schultze) (Simonsen, 1974)		Meave del Castillo & Lara-Villa (1997)
	Odontella	sp.		Meave del Castillo & Lara-Villa (1997)
	Olisthodiscus	*luteus* (N. Carter, 1937)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Opephora	*martiyi* (Héribaud)		Aké-Castillo & Vázquez (2008)
	Operculodinium	*israelianum* (M. Rossignol) (D. Wall, 1967)		Muciño-Márquez *et al.* (2011a)
	Ornithocercus	*magnificus* Stein		Figueroa-Torres *et al.* (2009)
	Oscillatoria	*nigroviridis* T. & G.		Aké-Castillo & Vázquez (2008)
	Oscillatoria	cf. *limosa* Agardh ex Gomant		Aké-Castillo & Vázquez (2008)
	Oscillatoria	sp.		Aké-Castillo & Vázquez (2008)
	Oscillatoria	sp.		Muciño-Márquez *et al.* (2011a)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Oscillatoria	sp. 1 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Ostreopsis	sp.		Muciño-Márquez *et al.* (2011a)
	Oxytoxum	*globosum* Schiller		Muciño-Márquez *et al.* (2011a)
	Oxytoxum	*mediterraneum* Schiller		Muciño-Márquez *et al.* (2011a)
	Oxytoxum	*sceptrum* (Stein) Schöder		Figueroa-Torres *et al.* (2009)
	Oxytoxum	sp.		Aké-Castillo & Vázquez (2008)
	Pandorina	*morum* (O. F. Müller) (Bory de Saint-Vincent, 1824)		Muciño-Márquez *et al.* (2011a)
	Paralia	*sulcata* (Ehrenberg) (Cleve, 1873)		Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a)
	Peridinium	aff. *quinquecorne*		Aké-Castillo & Vázquez (2008)
	Peridinium	sp. 1 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Peridinium	sp. 2 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Peridinium	*quinquecorne*	*trispiniferum*	Aké-Castillo & Vázquez (2011)
	Petrodyction	*gemma* (Ehr.) (D. G. Mann)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Petroneis	*humerosa* (Brébisson ex W.Smith) (A. J. Stickle & D.G.Mann, 1990)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Phacus	*caudatus* (Hübner, 1886)		Aké-Castillo & Vázquez (2008)
	Phormidium	*retzii* (C. Agardh) (Kützing, 1843)		Muciño-Márquez *et al.* (2011a)
	Phytoflagellate	sp. 1 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Phytoflagellate	sp. 2 *sensu* (Aké-Castillo & Vázquez, 2008)		Aké-Castillo & Vázquez (2008)
	Pinnularia	*lata* (Brébisson) (W. Smith, 1853)		Muciño-Márquez *et al.* (2011a)
	Pinnularia	*mesolepta* (Ehrenberg) (W. Smith, 1853)		Muciño-Márquez *et al.* (2011a)
	Pinnularia	*nobilis* (Ehrenberg) (Ehrenberg, 1843)		Muciño-Márquez *et al.* (2011a)
	Pinnularia	sp.		Aké-Castillo & Vázquez (2008)
	Pinnularia	sp.		Muciño-Márquez *et al.* (2011a)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Pinnularia	sp. 1 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Pinnularia	sp. 2 *sensu* (Muciño-Márquez *et al.*, 2011a)		Muciño-Márquez *et al.* (2011a)
	Plagiotropis	sp.		Aké-Castillo & Vázquez (2008)
	Planktolyngbya	sp.		Aké-Castillo & Vázquez (2008)
	Pleuromonas	sp.		Muciño-Márquez *et al.* (2011a)
	Pleurosigma	*acuminatum* (Kützing) (W. Smith, 1852)		Aké-Castillo & Vázquez (2008)
	Pleurosigma	*angulatum* (Queckett) (W. Smith, 1852)		Aké-Castillo & Vázquez (2008)
	Pleurosigma	*fasciola*	*closterioides* (Grunow) Peragallo	Meave del Castillo & Lara-Villa (1997)
	Pleurosigma	*formosum* (W. Smith, 1852)		Aké-Castillo & Vázquez (2008)
	Pleurosigma	*rigidum*	*incurvata* Brun in H.Peragallo 1891	Muciño-Márquez *et al.* (2011a)
	Pleurosigma	sp. 1 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Pleurosigma	sp. 2 *sensu* (Meave del Castillo & Lara-Villa, 1997)		Meave del Castillo & Lara-Villa (1997)
	Pleurosira	*laevis* (Ehrenberg) (Compère, 1982)		Aké-Castillo & Vázquez (2008)
	Pleurosira	sp.		Meave del Castillo & Lara-Villa (1997)
	Podolampas	*bipes* Stein		Figueroa-Torres *et al.* (2009)
	Podolampas	*palmipes* Stein		Figueroa-Torres *et al.* (2009)
	Podolampas	*reticulata* (Kofoid, 1907)		Muciño-Márquez *et al.* (2011a)
	Proboscia	*alata* (Brightwell) (Sundström, 1986)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Proboscia	sp.		Muciño-Márquez *et al.* (2011a)
	Prorocentrum	*compressum* (J.W.Bailey) Abé		Figueroa-Torres *et al.* (2009); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b); Muciño-Márquez *et al.* (2011c)
	Prorocentrum	*cordatum* (Ostenfeld) J. (D. Dodge, 1975)		Aké-Castillo & Vázquez (2008); Aké-Castillo (2015)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Prorocentrum	*gracile* (Schütt, 1895)		Aké-Castillo & Vázquez (2008); Figueroa-Torres *et al.* (2009); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b); Muciño-Márquez *et al.* (2011c)
	Prorocentrum	*lima* (Ehrenberg) (F. Stein, 1878)		Aké-Castillo & Vázquez (2008)
	Prorocentrum	*mexicanum* (Osorio-Tafall 1942)		Muciño-Márquez *et al.* (2011c)
	Prorocentrum	*micans* (Ehrenberg, 1834)		Aké-Castillo & Vázquez (2008); Figueroa-Torres *et al.* (2009); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b); Muciño-Márquez *et al.* (2011c)
	Prorocentrum	*robustum* (Osorio-Tafall, 1942)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011c)
	Prorocentrum	sp.		Muciño-Márquez *et al.* (2011a)
	Protoperidinium	*brochii* (Kofoid et Swezy) Balech		Figueroa-Torres *et al.* (2009)
	Protoperidinium	*claudicans* Paulsen		Figueroa-Torres *et al.* (2009)
	Protoperidinium	*conicum* (Gran) Balech		Figueroa-Torres *et al.* (2009)
	Protoperidinium	*corniculum* Kofoid et Michener		Figueroa-Torres *et al.* (2009)
	Protoperidinium	*crassipes* (Kofoid) (Balech, 1974)		Aké-Castillo & Vázquez (2008); Figueroa-Torres *et al.* (2009)
	Protoperidinium	*depressum* (Bailey) (Balech, 1974)		Figueroa-Torres *et al.* (2009); Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Protoperidinium	*divergens* Ehrenberg		Figueroa-Torres *et al.* (2009)
	Protoperidinium	*oceanicum* (Vanhöffen) Balech		Figueroa-Torres *et al.* (2009)
	Protoperidinium	*ovatum* (Pouchet, 1883)		Muciño-Márquez *et al.* (2011a)
	Protoperidinium	*pentagonum* (Gran) Balech		Figueroa-Torres *et al.* (2009)
	Protoperidinium	*punctulatum* (Paulsen) (Balech, 1974)		Muciño-Márquez *et al.* (2011a)
	Protoperidinium	sp.		Aké-Castillo & Vázquez (2008)
	Psammodictyon	*constrictum* (Gregory) (D. G.Mann, 1990)		Aké-Castillo & Vázquez (2008)
	Psammodictyon	*panduriforme* (W.Gregory) (D. G. Mann, 1990)		Muciño-Márquez *et al.* (2011a)
	Pseudonitzschia	*pungens* (Grunow ex P.T. Cleve) (Hasle, 1993)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Pseudonitzschia	*pungens*	atlantica Cleve, 1897	Muciño-Márquez *et al.* (2011a)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Pseudonitzschia	*seriata* (P. T. Cleve) (H. Peragallo in H. & M. Peragallo, 1900)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Pseudonitzschia	sp.		Aké-Castillo & Vázquez (2008)
	Pseudosolenia	*calcaravis* (Schultze, 1858)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Pyramimonas	sp.		Aké-Castillo & Vázquez (2008)
	Pyrocystis	*lunula* (Schütt) (Schütt, 1896)		Muciño-Márquez *et al.* (2011a)
	Pyrophacus	*horologium* (Stein, 1883)		Figueroa-Torres *et al.* (2009); Muciño-Márquez *et al.* (2011a)
	Pyrophacus	*steinii* (Schiller) Wall y Dale		Figueroa-Torres *et al.* (2009)
	Raphidonema	*nivale* (Lagerheim, 1892)		Muciño-Márquez *et al.* (2011a)
	Rhizosolenia	*acuminata* (H. Peragallo) (H. Peragallo, 1907)		Muciño-Márquez *et al.* (2011a)
	Rhizosolenia	*bergonii* (H. Peragallo, 1892)		Muciño-Márquez *et al.* (2011a)
	Rhizosolenia	*imbricata* (Brightwell, 1858)		Aké-Castillo & Vázquez (2008)
	Rhizosolenia	*robusta* (G. Norman ex Ralfs, 1861)		Meave del Castillo & Lara-Villa (1997)
	Rhizosolenia	*setigera* (Brightwell, 1858)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Rhizosolenia	sp.		Aké-Castillo & Vázquez (2008)
	Rhopalodia	sp.		Muciño-Márquez *et al.* (2011a)
	Scenedesmus	*armatus* (Chodat) (Chodat, 1913)		Aké-Castillo & Vázquez (2008)
	Scenedesmus	*dispar* (Brébisson, 1868)		Aké-Castillo & Vázquez (2008)
	Scenedesmus	*acuminatus* (Lagerheim) (Chodat, 1902)		Muciño-Márquez *et al.* (2011a)
	Scenedesmus	*arcuatus* (Lemmermann) (Lemmermann, 1899)		Muciño-Márquez *et al.* (2011a)
	Scenedesmus	*falcatus* (Chodat, 1894)		Muciño-Márquez *et al.* (2011a)
	Scenedesmus	cf. *pecsensis* Uherk		Aké-Castillo & Vázquez (2008)
	Scenedesmus	*quadricauda* (Turpin) (Brébisson, 1835)		Muciño-Márquez *et al.* (2011a)
	Scenedesmus	*quadricauda*	*westii* G.M.Smith, 1916	Muciño-Márquez *et al.* (2011a)
	Scrippsiella	*trochoidea* (Stein) (Loeblich III, 1976)		Muciño-Márquez *et al.* (2011a); Muciño-Márquez *et al.* (2011b)
	Scrippsiella	sp.		Aké-Castillo & Vázquez (2008)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Selenastrum	*westii* (G. M. Smith, 1920)		Muciño-Márquez *et al.* (2011a)
	Sellophora	*pupula* (Kützing) Mereschkowsky		Aké-Castillo & Vázquez (2008)
	Skeletonema	*costatum* (Greville) (Cleve, 1873)		Aké-Castillo *et al.* (1995); Aké-Castillo & Meave del Castillo (1997); Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Skeletonema	*pseudocostatum* (L. K. Medlin, 1991)		Aké-Castillo *et al.* (1995); Aké-Castillo & Vázquez (2008)
	Skeletonema	*subsalsum* (Cleve-Euler) (Bethge, 1928)		Aké-Castillo *et al.* (1995); Aké-Castillo & Meave del Castillo (1997); Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Spirulina	*nordstedtii* (Gomont, 1892)		Muciño-Márquez *et al.* (2011a)
	Staurastrum	*leptocladum* (Nordstedt, 1870)		Muciño-Márquez *et al.* (2011a)
	Surirella	*biseriata* (Brébisson, 1835)		Aké-Castillo & Vázquez (2008)
	Surirella	*fastuosa* (Ehrenberg) (Ehrenberg, 1843)		Aké-Castillo & Vázquez (2008)
	Surirella	*fastuosa*	*cuneata* O.Witt, 1873	Meave del Castillo & Lara-Villa (1997)
	Surirella	*febigerii* (F. W. Lewis, 1861)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Surirella	*linearis* (W. Smith, 1853)		Aké-Castillo & Vázquez (2008)
	Surirella	*robusta* Ehrenberg		Aké-Castillo & Vázquez (2008)
	Surirella	*Striatula* (Turpin, 1828)		Muciño-Márquez *et al.* (2011a)
	Synechococcus	sp.		Aké-Castillo & Vázquez (2008)
	Synedra	*goulardii* Brébisson ex (Cleve & Grunow, 1880)		Muciño-Márquez *et al.* (2011a)
	Synedra	*ulna* (Nitzsch) (Ehrenberg, 1832)		Muciño-Márquez *et al.* (2011a)
	Synedra	sp.		Meave del Castillo & Lara-Villa (1997)
	Terpsinoë	*musica* (Ehrenberg, 1843)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)
	Thalassionema	*nitzschioides* (Grunow) (Mereschkowsky, 1902)		Meave del Castillo & Lara-Villa (1997); Aké-Castillo & Vázquez (2008)

Table 1. (Continuation)

	Genus	Species	variety	References
Phytoplankton				
	Thalassionema	*nitzschioides*	*lanceolata* Grunow, 1928	Muciño-Márquez *et al.* (2011a)
	Thalassiosira	*cedarkeyensis* A.K.S.K. (Prasad, 1993)		Aké-Castillo & Vázquez (2008); Aké-Castillo (2015)
	Thalassiosira	*eccentrica* (Ehrenberg) (Cleve, 1904)		Aké-Castillo & Vázquez (2008)
	Thalassiosira	*lineata* (Jousé, 1968)		Aké-Castillo & Vázquez (2008)
	Thalassiosira	*oestrupii* (Ostenfeld) (Hasle, 1972)		Aké-Castillo & Vázquez (2008)
	Thalassiosira	*plicata* H. J. Schrader		Muciño-Márquez *et al.* (2011a)
	Thalassiosira	*simonsenii* Hasle & Fryxell		Aké-Castillo & Vázquez (2008)
	Thalassiosira	*decipiens* (Grunow) (E. G. Jørgensen, 1905)		Aké-Castillo & Vázquez (2008)
	Thalassiosira	*weissflogii* (Grunow) (G. Fryxell & Hasle, 1977)		Aké-Castillo & Vázquez (2008)
	Thalassiosira	sp.		Muciño-Márquez *et al.* (2011a)
	Thalassiothrix	*longissima* (Cleve & Grunow, 1880)		Aké-Castillo & Vázquez (2008)
	Trachelomonas	sp.		Muciño-Márquez *et al.* (2011a)
	Trachyneis	*aspera* (Ehrenberg) (Cleve, 1894)		Aké-Castillo & Vázquez (2008)
	Tryblionella	*cocconeiformis* (Grunow) (D. G. Mann, 1990)		Aké-Castillo & Vázquez (2008)
	Tryblionella	*parvula* (W. Smith) (T. Ohtsuka & Y. Fujita, 2001)		Aké-Castillo & Vázquez (2008)
	Tryblionella	*punctata* (W. Smith, 1853)		Aké-Castillo & Vázquez (2008)
	Tryblionella	cf. *calida* (Grunow) D. G. Mann		Aké-Castillo & Vázquez (2008)
	Ulothrix	sp.		Muciño-Márquez *et al.* (2011a)
	Zygabikodinium	*lenticulatum* (Loeblich Jr. & Loeblich III, 1970)		Figueroa-Torres *et al.* (2009); Muciño-Márquez *et al.* (2011a)
Macroalgae				
	Ulva	*intestinalis* (Linnaeus, 1753)		Calva & Torres-Alvarado (2008)
	Bostrychia	*pinnata* (J. Tanaka & Chihara 1984)		Calva & Torres-Alvarado (2008)
	Bostrychia	*calliptera* (Montagne) Montagne		Collado-Vides & West (1996)
	Cladophora (antes Cladophoris)	*membranacea* (Hofman Bang ex C. Agardh) Kützing		González-Fierro *et al.* (1994)

Table 1. (Continuation)

	Genus	Species	variety	References
Seagrass				
	Ruppia	*maritima* Linnaeus		González-Gutiérrez (1977); González-Fierro *et al.* (1994); Calva *et al.* (2005); Calva & Torres-Alvarado (2008)
Mangrove				
	Rhizophora	*mangle* L.		Medina *et al.* (1995); Ángeles-Álvarez (1997b); Barradas-García *et al.* (2004); Carmona-Díaz *et al.* (2004); Calva *et al.* (2005); Aké-Castillo *et al.* (2006); Hernández-Carmona *et al.* (2012); Hernández-Carmona *et al.* (2014)
	Laguncularia	*racemosa* (L.) C. F. Gaertn.		Medina *et al.* (1995); Barradas-García *et al.* (2004); Carmona-Díaz *et al.* (2004); Calva *et al.* (2005); Aké-Castillo *et al.* (2006); Hernández-Carmona *et al.* (2012); Hernández-Carmona *et al.* (2014)
	Avicennia	*germinans* (L.) L.		Barradas-García *et al.* (2004); Carmona-Díaz *et al.* (2004); Calva *et al.* (2005); Aké-Castillo *et al.* (2006); Hernández-Carmona *et al.* (2012); Hernández-Carmona *et al.* (2014)
Coastal vegetation				
	Tetrorchidium	*rotundatum* Standl.		Carmona-Díaz *et al.* (2004)
	Andira	*galeottiana* Standley, 1919		Carmona-Díaz *et al.* (2004)
	Eichhornia	*crassipes* (Mart.) Solms		Calva *et al.* (2005)
	Typha	*dominguensis* Pers.		Calva *et al.* (2005)
	Pachira	*aquatica* Aubl.		Ángeles-Álvarez (1997a); Medina *et al.* (1995); Aké-Castillo *et al.* (2006)
	Inga	sp.		Hernández-Carmona *et al.* (2014)
	Capparidastrum	*tuxtlense* Cornejo & Iltis		Cornejo & Iltis (2010)
Fern				
	Acrostichum	*aureum* L.		Medina *et al.* (1995); Aké-Castillo *et al.* (2006); Hernández *et al.* (2014)

Table 1. (Continuation)

	Genus	Species	variety	References
Legumes				
	Muellera	*frutescens* (Aubl.) Standl.		Sousa & Sotuyo (2012)
	Muellera	*unifoliolata* (Benth.) M. Sousa		Sousa & Sotuyo (2012)
Lianas or climbing plants				
	Rhabdadenia	*biflora* (Jacq.) Müll. Arg.		Hernández-Carmona *et al.* (2014)
Epiphytes				
Bromeliads				
	Aechmea	*bracteata* (Sw.) Griseb		Carmona-Díaz *et al.* (2014)
	Tillandsia	*concolor*		Carmona-Díaz *et al.* (2004)
Orchids				
	Cycnoches	*ventricosum* Bateman		Carmona-Díaz *et al.* (2004)
	Chysis	*bractescens* Lindl.		Carmona-Díaz *et al.* (2004)
	Brassavola	*nodosa* (L.) Lindl.		Carmona-Díaz *et al.* (2004)
	Encyclia	*alata* (Batem.) (Schlechter, 1673)		Carmona-Díaz *et al.* (2004)
	Encyclia	*belizensis* (Reich. F.) (Schlechter, 1674)		Carmona-Díaz *et al.* (2004)
	Epidendrum	*nocturnum* Jacq.		Carmona-Díaz *et al.* (2004)
	Epidendrum	*ciliare* L.		Carmona-Díaz *et al.* (2004)

2. Fauna

2.1. Invertebrates

Table 2 lists the invertebrate species that have been reported for the Sontecomapan Lagoon. There are a total of 201 species: 5 platyhelminthes, 6 trematodes, 2 nematods, 69 rotifers, 8 acantocephalans, 69 molluscs and 39 crustaceans (1 pentastomid, 3 cladocerans, 14 copepods, 1 rhizocephalan, 1 tanaidacean and 19 decapods), and 3 insects (odonata).

2.1.1. Worms (Platelminthes, Trematods, Nematods, Rotifers, Acanthocephalans)

Salgado-Maldonado (1976) recorded the presence of acanthocephalans *Dollfusentis chandleri* and described the new species *Dollfusentis bravoae*. Both were found in the intestine of the fish Pomadasys crocro obtained in Sontecoman Lagoon.

Salgado-Maldonado (1977) described the acanthocephalan *Caballerorhynchus lamothei*, being *Caballerorhynchus*, as a new genus and *lamothei* as a new species. *Caballerorhynchus lamothei* was found as parasite of the *Diapterus olisthostomus* fish.

A study on the reports of parasites found in wild animals from Los Tuxtlas region (Lamothe-Argumedo *et al.* 1997) included 14 species of parasites found in 10 different hosts in Sontecomapan Lagoon.

A new species of trematode (*Glypthelmins poncedeleoni* n. sp.) was described by Razo-Mendivil & León-Règagnon (2001). They obtained the trematode from the intestine of the *Leptodactylus melanonotus* frog. Later on, the species was renamed as *Rauschiella poncedeleoni* (Razo-Mendivil *et al.*, 2006).

Salgado-Maldonado & Cruz-Reyes (2002) carried out a study in different places at the States of Veracruz, Tabasco and Chiapas. They discovered a new acanthocephalan species: *Porrorchis nickoli*; this new species was found in 4 mammal species; 3 opossums (*Philander opossum*, *Didelphis virginiana* and *Didelphis marsupialis*) and 1 badger (*Nasua narica*). In Sontecompan Lagoon 2 *Didelphis virginiana* individuals were found with 2 acanthocephalans each.

Rotifer diversity was determined from March, 2000 to October, 2003. The samples were obtained at different depths of the water column and from the littoral vegetation. The authors reported 59 species from which 19 belong to the genus *Lecane* and 14 to the genus *Brachionus* (Castellanos-Páez *et al.*, 2005).

In a checklist of metazoan parasites of Mexican amphibians and reptiles (Paredes-León *et al.*, 2008), the authors referenced the *Rauschiella poncedeleoni* parasite found in the intestine of the *Leptodactylus melanonotus* frog (Razo-Mendivil & León-Règagnon, 2001) and the pentastomid parasite *Porocephalus crotali* obtained from the lungs of the *Bothrops asper* snake (Peláez & Juliá, 1983).

In the checklist for the Acanthocephala in wildlife vertebrates of Mexico, García-Prieto *et al.* (2010) mentioned the acanthocephalan species: a) *Caballerorhynchus lamothei* (Salgado-Maldonado, 1977) and their hosts in Sontecomapan Lagoon *Centropomus undecimalis*, *Eugerres plumieri* (Salgado-Maldonado, 1980); *Diapterus auratus* (Salgado-Maldonado, 1977) and *Discapseudes holthuisi* (Escobar-Briones *et al.* 1999); b) *Dollfusentis bravoae* encountered in the intestine of the *Pomadasys crocro* fish (Salgado-Maldonado, 1976); c) *Dollfusentis chandleri* in *Diapterus auratus* and *Eugerres plumieri* (Pérez-Ponce de León *et al.* 1996); d) *Floridosentis mugilis* in the intestine of *Mugil cephalus* (Salgado-Maldonado & Barquín, 1978); e) *Porrorchis nickoli* in the intestine of the *Didelphis virginiana* mammal (Salgado-Maldonado & Cruz-Reyes, 2002); f) *Gorgorhynchus medius* in the *Cathorops melanopus* fish (Salgado-Maldonado, 1978) and g) *Serrasentis sagittifer* in *Bagre marinus* (Salgado-Maldonado, 1978).

A phylogenetic study of *Floridosentis* spp. using molecular tools was performed by Rosas-Valdez *et al.* (2012) where they reported the appearance of the acanthocephalan *Floridosentis mugilis* in *Mugil cephalus* from the Sontecomapan Lagoon.

In 2013 a taxonomical integral study (molecular analysis, morphology, host association and geographic distribution) of the trematode *Clinostomum complanatum* was carried out. The authors worked with fish and birds that feed on fish at different sites of the Mexican Republic. They described a new species: *Clinostomum tataxumui*, present in the *Ardea herodias* brown heron and in the *Dormitator maculatus* fish of the Sontecomapan Lagoon, among other studied ecosystems where its existence was also reported (Sereno-Uribe *et al.*, 2013).

Benítez-Díaz Mirón *et al.* (2014) studied the spatio-temporal variations of the zooplankton community at Sontecomapan Lagoon. They reported 21 species of rotifers, 3 species of cladocerans and 14 species of copepods.

A new species of Acanthocephala *Neoechinorhynchus* (*Neoechinorhynchus*) *mexicoensis* was described by Pinacho-Pinacho *et al.* (2014); it was found in the intestine of the *Dormitator maculatus* fish (Bloch, 1792) collected at different sites of the Gulf of Mexico.

2.1.2. Mollusks

Rangel Ruiz (1987) studied the morphology of the *Pomacea flagellata* snail, the specimens examined were collected from Laguna Escondida, El Zacatal, and Sontecomapan Lagoon.

The presence of metals in the *Crassostrea rhizophorae* and *Brachidontes exustus* benthic organisms of Sontecomapan Lagoon was determined by González-Fierro *et al.* (1994).

García-Cubas & Reguero (1995) found 60 mollusk species pertaining to the Gastropoda and Bivalvia classes in Sontecomapan Lagoon, which has an average depth of 1 m, warm water with an average temperature of 24 °C and predominantly mesohaline (5-18 ppt) and a sandy-clayey substrate.

Naranjo & Polaco (1997) reported the presence of 7 mollusk species in the Sontecomapan Lagoon and its sourroundings. The species were: *Tomocyclus lunai* Bartsch, 1945; *Pomacea flagellata* (Say, 1827), *Pachychilus* sp., *Aplexa* (*Stenophysa*) *spiculata* (Morelet) now *Mayabina spiculata* (Morelet, 1849), *Physa* (*Alampetis*) *squalida* Morelet, 1851 now *Haitia mexicana* (Philippi, 1841), *Succinea undulata* Say 1829, and *Omphalina modesta* Von Martens, 1892 now *Mesomphix* (*Zonyalina*) *modestus* (Von Martens, 1892).

Calva *et al.* (2005) in their work about the composition of aliphatic hydrocarbons in sediments from the Sontecomapan Lagoon, identified oyster banks of the *Crassostrea rhizophorae* and *Crassostrea virginica*, and of mussel *Brachidontes exustus* species.

Aké-Castillo *et al.* (2006) reported the decomposition of *Rizophora mangle*. They found that the decomposition rate during the rainy season was the fastest, coinciding with the fact that in this season the gastropod *Neritina reclivata* collected from the mangrove leaf litter bags, also reached its maximum biomass. Thus, the authors concluded that this gastropod activity is important for the *R. mangle* decomposition in the Sontecomapan Lagoon.

2.1.3. Crustaceans

The pentastomid *Porocephalus crotali* was found in the lungs of the *Bothrops asper* snake from Sontecomapan in a study carried out with 266 mexican ophidia from 19 states, which were examined by necropsy (Peláez & Juliá, 1983).

Rocha-Ramírez *et al.* (1992) studied the *Callinectes* genus found in coastal water bodies from the State of Veracruz, the Sontecomapan Lagoon was sampled at 17 sampling points in two periods: from October, 1980 to May, 1981 and from October, 1982 to March, 1983. The authors reported the presence of *Callinectes sapidus, C. similis* and *C. rathbunae*. 141 individuals were obtained from which 47% were *C. sapidus*, 37% *C. similis* and 16% *C. rathbunae*. In addition, they found that the greatest abundance was from individuals of smaller than 90 mm and that in general, the sex ratio was maintained at 1:1.

A list of decapod crustaceans found in Los Tuxtlas region was given and 17 species from the lagoon were included (Álvarez & Villalobos, 1997). Camacho *et al.* (1997) mentioned that *Macrobrachium acanthurus* was abundant in the lagoon and its tributaries, Basura and La Palma rivers. The existence of Atyidae species, *Potimirim mexicana* called "camaroncito" (little shrimp) by the local people of Los Tuxtlas, was reported in Sontecomapan Lagoon (Villalobos & Álvarez, 1997).

Álvarez *et al.* (1999) recorded the existence of the *Callinectes larvatus* crustacean decapod in the Sontecomapan Lagoon.

Escobar-Briones *et al.* (1999) reported the *Discapseudes holthuisi* tanaid shrimp as intermediate host of the acanthocephalan *Caballerorhynchus lamothei*, found in Alvarado and Sontecomapan Lagoons. *D. holthuisi* inhabits the sediment and feeds on small particles of organic matter, where it simultaneously ingests acanthocephalan eggs thus, when the shrimp is consumed by its predators, they can be infected as well.

Bortolini & Alvarez (2008) reported the alteration of the hepatopancreas of the *Callinectes sapidus* blue crab caused by the rhizocephalan parasite (Crustacea: Rhizocephala) *Loxothylacus texanus*, collected from the Sontecomapan Lagoon.

2.1.4. Insects

Three species of Odonata (*Hetaerina titia, Neoneura amelia* and *Erythemis haematogastra*) were reported for the Sontecomapan Lagoon by González-Soriano (1997).

Table 2. Species of invertebrates reported for the Sontecompan Lagoon, Veracruz

	Genus	Species	References
PLATYHELMINTHES			
	Allopyragraphorus	*winteri* (Caballero & Bravo-Hollis, 1965) Bravo-Hollis & Salgado-Maldonado, 1983	Lamothe-Argumedo *et al.* (1997)
	Gotocotyla	*acanthura* (Parona & Perugia, 1896) Meserve, 1938	Lamothe-Argumedo *et al.* (1997)
	Mexicotyle	*mexicana* (Meserve, 1938) Lebedev, 1984	Lamothe-Argumedo *et al.* (1997)
	Thoracocothyle	*crocea* MacCallum, 1913	Lamothe-Argumedo *et al.* (1997)
	Thoracocotyle	*mexicana* Bravo-Holiis & Lamothe, 1988	Lamothe-Argumedo *et al.* (1997)
TREMATODS			
	Bicotylophora	*trachinoti* (MacCallum, 1921)	Lamothe-Argumedo *et al.* (1997)
	Lobatostoma	*kemostoma* (MacCallum & MacCallum, 1913)	Lamothe-Argumedo *et al.* (1997)
	Lobatostoma	*ringens* (Linton, 1907)	Lamothe-Argumedo *et al.* (1997)
	Paragonimus	*mexicanus* Miyazaki e Ishii, 1968	Lamothe-Argumedo *et al.* (1997)
	Clinostomum	*tataxumui* n. sp.	Sereno-Uribe *et al.* (2013)
	Rauschiella (Glypthelmins)	*poncedeleoni* n. sp.	Razo-Mendivil & León-Règagnon (2001); Paredes-León *et al.* (2008)
NEMATODS			
	Aprocta	*travassosi* Caballero, 1938	Lamothe-Argumedo *et al.* (1997)
	Cucullanus	sp.	Lamothe-Argumedo *et al.* (1997)
ROTIFERS			
	Anuraeopsis	*fissa* Gosse, 1851	Castellanos-Páez *et al.* (2005)
	Asplanchna	*priodonta* Gosse, 1850	Castellanos-Páez *et al.* (2005)
	Asplanchna	*sieboldii* (Leydig, 1854)	Castellanos-Páez *et al.* (2005)
	Brachionus	*angularis* Gosse, 1851	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Brachionus	*bidentatus* Anderson, 1889	Castellanos-Páez *et al.* (2005)
	Brachionus	*calyciflorus* Pallas, 1766	Castellanos-Páez *et al.* (2005)
	Brachionus	*dimidiatus* Bryce, 1931	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Brachionus	*durgae* Dhanapathi, 1974	Castellanos-Páez *et al.* (2005)
	Brachionus	*falcatus* Zacharias, 1898	Castellanos-Páez *et al.* (2005)
	Brachionus	*havanaensis* Rousselet, 1911	Castellanos-Páez *et al.* (2005)
	Brachionus	*plicatilis* Müller, 1786	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)

Table 2. (Continuation)

	Genus	Species	References
ROTIFERS			
	Brachionus	*pterodinoides* Rousselet, 1913	Castellanos-Páez *et al.* (2005)
	Brachionus	*quadridentatus* Hermann, 1783	Castellanos-Páez *et al.* (2005)
	Brachionus	*rotundiformis* Tschugunoff, 1921	Castellanos-Páez *et al.* (2005)
	Brachionus	*rubens* Ehrenberg, 1838	Castellanos-Páez *et al.* (2005)
	Brachionus	*urceolaris* Müller, 1773	Castellanos-Páez *et al.* (2005)
	Brachionus	*variabilis* Hempel, 1896	Castellanos-Páez *et al.* (2005)
	Cephalodella	*gibba* (Ehrenberg, 1830)	Castellanos-Páez *et al.* (2005)
	Cephalodella	*hoodii* (Gosse, 1886)	Castellanos-Páez *et al.* (2005)
	Colurella	*colurus* (Ehrenberg, 1830)	Castellanos-Páez *et al.* (2005)
	Colurella	*hindenburgi* Steinecke, 1917	Castellanos-Páez *et al.* (2005)
	Colurella	*oblonga* Donner, 1943	Castellanos-Páez *et al.* (2005)
	Colurella	*obtusa* (Gosse, 1886)	Castellanos-Páez *et al.* (2005)
	Enteroplea	*lacustris* Ehrenberg, 1830	Castellanos-Páez *et al.* (2005)
	Epiphanes	*clavulata* (Ehrenberg, 1832)	Castellanos-Páez *et al.* (2005)
	Epiphanes	*macroura* (Barrois & Daday, 1894)	Castellanos-Páez *et al.* (2005)
	Filinia	*longiseta* (Ehrenberg, 1834)	Castellanos-Páez *et al.* (2005)
	Keratella	*americana* Carlin, 1943	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Keratella	*quadrata* (Müller, 1786)	Benítez-Díaz Mirón *et al.* (2014)
	Keratella	*serrulata* (Ehrenberg, 1838)	Castellanos-Páez *et al.* (2005)
	Euchlanis	*incisa* Carlin, 1939	Benítez-Díaz Mirón *et al.* (2014)
	Euchlanis	*dilatata* Ehrenberg, 1832	Benítez-Díaz Mirón *et al.* (2014)
	Lecane	*bifurca* (Bryce, 1892)	Castellanos-Páez *et al.* (2005)
	Lecane	*bulla* (Gosse, 1851)	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Lecane	*decipiens* (Murray, 1913)	Castellanos-Páez *et al.* (2005)
	Lecane	*hamata* (Stokes, 1896)	Castellanos-Páez *et al.* (2005)
	Lecane	*leontina* (Turner, 1892)	Castellanos-Páez *et al.* (2005)
	Lecane	*ludwigii* (Eckstein, 1883)	Castellanos-Páez *et al.* (2005)
	Lecane	*luna* (Müller, 1776)	Castellanos-Páez *et al.* (2005)
	Lecane	*lunaris* (Ehrenberg, 1832)	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Lecane	*nana* (Murray, 1913)	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Lecane	*nelsoni* Segers, 1994	Castellanos-Páez *et al.* (2005)

Table 2. (Continuation)

	Genus	Species	References
ROTIFERS			
	Lecane	*obtusa* (Murray, 1913)	Castellanos-Páez *et al.* (2005)
	Lecane	*perpusilla* (Hauer, 1929)	Castellanos-Páez *et al.* (2005)
	Lecane	*pertica* Harring & Myers, 1926	Castellanos-Páez *et al.* (2005)
	Lecane	*quadridentata* (Ehrenberg, 1830)	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Lecane	*satyrus* Harring & Myers, 1926	Castellanos-Páez *et al.* (2005)
	Lecane	*stokesii* (Pell, 1890)	Castellanos-Páez *et al.* (2005)
	Lecane	*subtilis* Harring & Myers, 1926	Castellanos-Páez *et al.* (2005)
	Lecane	*tenuiseta* Harring, 1914	Castellanos-Páez *et al.* (2005)
	Lecane	*venusta* Harring & Myers, 1926	Castellanos-Páez *et al.* (2005)
	Lepadella (Lepadella)	*donneri* Koste, 1972	Benítez-Díaz Mirón *et al.* (2014)
	Lepadella (Lepadella)	*ovalis* (Müller, 1786)	Castellanos-Páez *et al.* (2005)
	Lepadella (Lepadella)	*patella* (Müller, 1773)	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Lindia (Lindia)	*ecela* Myers, 1933	Castellanos-Páez *et al.* (2005)
	Mytilina	*acanthophora* Hauer, 1938	Castellanos-Páez *et al.* (2005)
	Mytilina	*bisulcata* (Lucks, 1912)	Benítez-Díaz Mirón *et al.* (2014)
	Mytilina	*mucronata* (Müller, 1773)	Castellanos-Páez *et al.* (2005)
	Mytilina	*ventralis* (Ehrenberg, 1830)	Benítez-Díaz Mirón *et al.* (2014)
	Notholca	*bipalium* (Müller, 1786)	Castellanos-Páez *et al.* (2005)
	Plationus	*patulus* (Müller, 1786)	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Platyias	*quadricornis* (Ehrenberg, 1832)	Castellanos-Páez *et al.* (2005); Benítez-Díaz Mirón *et al.* (2014)
	Synchaeta	*oblonga* Ehrenberg, 1832	Benítez-Díaz Mirón *et al.* (2014)
	Synchaeta	*bicornis* Smith, 1904	Benítez-Díaz Mirón *et al.* (2014)
	Testudinella	*patina* (Hermann, 1783)	Benítez-Díaz Mirón *et al.* (2014)
	Trichocerca	*bicristata* (Gosse, 1887)	Castellanos-Páez *et al.* (2005)
	Trichocerca	*marina* (Daday, 1890)	Castellanos-Páez *et al.* (2005)
	Trichocerca	*pusilla* (Jennings, 1903)	Castellanos-Páez *et al.* (2005)
	Trichotria	*tetractis* (Ehrenberg, 1830)	Benítez-Díaz Mirón *et al.* (2014)

Table 2. (Continuation)

	Genus	Species	References
ACANTHOCEPHALANS			
	Caballerorhynchus	*lamothei* Salgado-Maldonado, 1977	Salgado-Maldonado (1977); Lamothe-Argumedo *et al.* (1997); Escobar-Briones *et al.* (1999)
	Dollfusentis	*chandleri* Golvan, 1969	Salgado-Maldonado (1976)
	Dollfusentis	*bravoae* Salgado-Maldonado, 1976	Salgado-Maldonado (1976); García-Prieto *et al.* (2010)
	Floridosentis	*mugilis* (Machado-Filho, 1951)	Lamothe-Argumedo *et al.* (1997); García-Prieto *et al.* (2010); Rosas-Valdez *et al.* (2012)
	Gorgorhynchus	*medius* (Linton, 1908)	Salgado-Maldonado (1978); Lamothe-Argumedo *et al.* (1997); García-Prieto *et al.* (2010)
	Neoechinorhynchus	*mexicoensis* Pinacho-Pinacho, Sereno-Uribe and García-Varela, 2014	Pinacho-Pinacho *et al.* (2014)
	Porrorchis	*nickoli* Salgado-Maldonado and Cruz-Reyes, 2002	Salgado-Maldonado & Cruz-Reyes (2002); García-Prieto *et al.* (2010)
	Serrasentis	*sagittifer* (Linton, 1889)	García-Prieto *et al.* (2010)
MOLLUSKS			
	Neritina	*reclivata* (Say, 1822)	García-Cubas & Reguero (1995); Aké-Castillo *et al.* (2006)
	Neritina (Vitta)	*virginea* (Linnaeus, 1758)	García-Cubas & Reguero (1995)
	¿Littoridina (texadina)	*sphinctostoma* (Abbott y Ladd, 1951)	García-Cubas & Reguero (1995)
	Truncatella	*pulchella* (Pfeiffer, 1839)	García-Cubas & Reguero (1995)
	Caecum (Caecum)	*pulchellum* (Stimpson, 1851)	García-Cubas & Reguero (1995)
	Diastoma	*varium* (Pfeiffer, 1840)	García-Cubas & Reguero (1995)
	Cerithidea (Cerithideopsis)	*pliculosa* (Menke, 1829)	García-Cubas & Reguero (1995)
	Crepidula (lanacus)	*plana* (Say, 1822)	García-Cubas & Reguero (1995)
	Polinices (Neverita)	*duplicatus* (Say, 1822)	García-Cubas & Reguero (1995)
	Phos (Antillophos)	*candei* (d'Orbigni, 1845)	García-Cubas & Reguero (1995)
	Anachis (Costanachis)	*avara* (Say, 1822)	García-Cubas & Reguero (1995)
	Anachis (Costanachis)	*lafresnayi* (Fischer y Bernardi, 1856)	García-Cubas & Reguero (1995)
	Mitrella (Astyris)	*lunata* (Say, 1826)	García-Cubas & Reguero (1995)
	Nassarius (Nassarius)	*acutus* (Say, 1822)	García-Cubas & Reguero (1995)
	Nassarius (Nassarius)	*vibex* (Say, 1822)	García-Cubas & Reguero (1995)
	Olivella (Niteoliva)	*minuta* (Link, 1807)	García-Cubas & Reguero (1995)
	Odostomia (Menestho)	*impressa* (Say, 1822)	García-Cubas & Reguero (1995)

Table 2. (Continuation)

	Genus	Species	References
MOLLUSKS			
	Bulla	*striata* (Bruguiere, 1792)	García-Cubas & Reguero (1995)
	Haminoea	*succinea* (Conrad, 1846)	García-Cubas & Reguero (1995)
	Acteocina	*canaliculata* (Say, 1822)	García-Cubas & Reguero (1995)
	Nuculana (Saccella)	*acuta* (Conrad, 1831)	García-Cubas & Reguero (1995)
	Anadara (Larkinia)	*transversa* (Say, 1822)	García-Cubas & Reguero (1995)
	Anadara (Sectiarca)	*floridana* (Conrad, 1869)	García-Cubas & Reguero (1995)
	Anadara (Lunarca)	*ovalis* (Bruguiere, 1789)	García-Cubas & Reguero (1995)
	Brachidontes (Hormomya)	*exustus* (Linnaeus, 1758)	González-Fierro *et al.* (1994); García-Cubas & Reguero (1995); Calva *et al.* (2005)
	Musculus (Ryenella)	*lateralis* (Say, 1822)	García-Cubas & Reguero (1995)
	Atrina	*seminuda* (Lamarck, 1819)	García-Cubas & Reguero (1995)
	Isognomon	*alatus* (Gmelin, 1791)	García-Cubas & Reguero (1995)
	Isognomon	*bicolor* (C.B. Adams, 1845)	García-Cubas & Reguero (1995)
	Anomia	*simplex* (d'Orbigni, 1845)	García-Cubas & Reguero (1995)
	Mytilopsis	*leucophaeata* (Conrad, 1831)	García-Cubas & Reguero (1995)
	Lucina (Parvilucina)	*multilineata* (Tuomey y Holmes, 1857)	García-Cubas & Reguero (1995)
	Lucina (Phacoides)	*pectinata* (Gmelin, 1791)	García-Cubas & Reguero (1995)
	Diplodonta	*punctata* (Say, 1822)	García-Cubas & Reguero (1995)
	Mysella	*planulata* (Stimpson, 1857)	García-Cubas & Reguero (1995)
	Trachycardium (Dallocardia)	*muricatum* (Linnaeus, 1758)	García-Cubas & Reguero (1995)
	Cyclinella	*tenuis* (Récluz, 1852)	García-Cubas & Reguero (1995)
	Chione (Chione)	*cancellata* (Linaeus, 1767)	García-Cubas & Reguero (1995)
	Chione (Lirophora)	*paphia* (Linaeus, 1767)	García-Cubas & Reguero (1995)
	Chione (Timoclea)	*grus* (Holmes, 1858)	García-Cubas & Reguero (1995)
	Mercenaria	*campechiensis* (Gmelin, 1791)	García-Cubas & Reguero (1995)
	Mulinia	*lateralis* (Say, 1822)	García-Cubas & Reguero (1995)
	Rangia	*cuneata* (Gray, 1831)	García-Cubas & Reguero (1995)
	Rangia (Rangianella)	*flexuosa* (Conrad, 1839)	García-Cubas & Reguero (1995)
	Tellina (Eurytellina)	*alternata* (Say, 1822)	García-Cubas & Reguero (1995)
	Macoma	*tenta* (Say, 1834)	García-Cubas & Reguero (1995)
	Macoma (Austromacoma)	*constricta* (Bruguiere, 1799)	García-Cubas & Reguero (1995)
	Macoma (Psammacoma)	*brevifrons* (Say, 1834)	García-Cubas & Reguero (1995)
	Macoma (Psammacoma)	*tageliformis* (Dall, 1900)	García-Cubas & Reguero (1995)
	Donax	*variabilis* (Philippi, 1847)	García-Cubas & Reguero (1995)

Table 2. (Continuation)

	Genus	Species	References
MOLLUSKS			
	Tagelus (Mesopleura)	*divisus* (Spengler, 1794)	García-Cubas & Reguero (1995)
	Tagelus (Mesopleura)	*plebeius* (Lightfoot, 1786)	García-Cubas & Reguero (1995)
	Abra	*aequalis* (Say, 1822)	García-Cubas & Reguero (1995)
	Abra	*lioica* (Dall, 1881)	García-Cubas & Reguero (1995)
	Semeie	*proficua* (Pulteney, 1799)	García-Cubas & Reguero (1995)
	Sphenia	*antillensis* (Dall y Simpson, 1901)	García-Cubas & Reguero (1995)
	Corbula (Caryocorbula)	*contracta* (Say, 1822)	García-Cubas & Reguero (1995)
	Corbula (Caryocorbula)	*dietziana* (C.B. Adams, 1852)	García-Cubas & Reguero (1995)
	Corbula (Caryocorbula)	*swiftiana* (C.B. Adams, 1852)	García-Cubas & Reguero (1995)
	Martesia	*striata* (Linaeus, 1767)	García-Cubas & Reguero (1995)
	Crassostrea	*rhizophorae* Guilding	González-Fierro *et al.* (1994); Calva *et al.* (2005)
	Crassostrea	*virginica* (Gmelin, 1791)	Calva *et al.* (2005)
	Pomacea	*flagellata* (Say, 1827)	Rangel Ruiz (1987); Naranjo & Polaco (1997)
	Tomocyclus	*lunai* (Bartsch, 1945)	Naranjo & Polaco (1997)
	Pachychilus	sp.	Naranjo & Polaco (1997)
	Mayabina	*spiculata* (Morelet, 1849)	Naranjo & Polaco (1997)
	Haitia	*mexicana* (Philippi, 1841)	Naranjo & Polaco (1997)
	Succinea	*undulata* (Say 1829)	Naranjo & Polaco (1997)
	Mesomphix (Zonyalina)	*modestus* (Von Martens, 1892)	Naranjo & Polaco (1997)
CRUSTACEANS			
PENTASTOMID			
	Porocephalus	*crotali* (Humboldt, 1808)	Peláez & Juliá (1983); Paredes-León *et al.* (2008)
CLADOCERANS			
	Chydorus	sp.	Benítez-Díaz Mirón *et al.* (2014)
	Ceriodaphnia	sp.	Benítez-Díaz Mirón *et al.* (2014)
	Penilia	*avirostris* (Dana, 1849)	Benítez-Díaz Mirón *et al.* (2014)
COPEPODS			
	Acartia	*tonsa* Dana, 1852	Benítez-Díaz Mirón *et al.* (2014)
	Centropages	*velificatus* (Oliveira, 1947)	Benítez-Díaz Mirón *et al.* (2014)
	Paracalanus	*aculeatus* (Giesbrecht, 1888)	Benítez-Díaz Mirón *et al.* (2014)
	Phaenna	sp.	Benítez-Díaz Mirón *et al.* (2014)
	Temora	*turbinata* (Dana, 1852)	Benítez-Díaz Mirón *et al.* (2014)

Table 2. (Continuation)

	Genus	Species	References
COPEPODS			
	Pseudodiaptomus	sp.	Benítez-Díaz Mirón *et al.* (2014)
	Mesocyclops	sp.	Benítez-Díaz Mirón *et al.* (2014)
	Oithona	*nana* (Giesbrecht, 1892)	Benítez-Díaz Mirón *et al.* (2014)
	Oithona	sp.	Benítez-Díaz Mirón *et al.* (2014)
	Corycaeus	sp.	Benítez-Díaz Mirón *et al.* (2014)
	Oncaea	*venusta* Philippi, 1843	Benítez-Díaz Mirón *et al.* (2014)
	Euterpina	*acutiformis* Dana, 1852	Benítez-Díaz Mirón *et al.* (2014)
	Tisbe	sp.	Benítez-Díaz Mirón *et al.* (2014)
	Canuella	sp.	Benítez-Díaz Mirón *et al.* (2014)
RHIZOCEPHALAN			
	Loxothylacus	*texanus* Boschma, 1933	Bortolini & Alvarez (2008)
TANAIDS			
	Discapseudes	*holthuisi* Bacescu and Gutu, 1975	Escobar-Briones *et al.* (1999)
DECAPODS			
	Farfantepenaeus	*aztecus* (Ives, 1891)	Álvarez & Villalobos (1997)
	Macrobrachium	*acanthurus* (Wiegmann, 1836)	Álvarez & Villalobos (1997); Camacho *et al.* (1997)
	Macrobrachium	*olfersii* (Wiegmann, 1836)	Álvarez & Villalobos (1997)
	Macrobrachium	*hobbsi* Nates and Villalobos, 1990	Álvarez & Villalobos (1997)
	Callinectes	*sapidus* M. J. Rathbun, 1896	Rocha-Ramírez *et al.* (1992); Álvarez & Villalobos (1997); Bortolini & Alvarez (2008)
	Callinectes	*rathbunae* Contreras, 1930	Rocha-Ramírez *et al.* (1992); Álvarez & Villalobos (1997)
	Callinectes	*similis* A. B. Williams, 1966	Rocha-Ramírez *et al.* (1992); Álvarez & Villalobos (1997)
	Callinectes	*larvatus* Ordway, 1863	Álvarez *et al.* (1999)
	Arenaeus	*cribarius* (Lamarck, 1818)	Álvarez & Villalobos (1997)
	Cardisoma	*guanhumi* Latreille, 1828	Álvarez & Villalobos (1997)
	Goniopsis	*cruentata* (Latreille, 1802)	Álvarez & Villalobos (1997)
	Armases	*americanum* (De Saussure, 1858)	Álvarez & Villalobos (1997)
	Armases	*cinereum* (Bosc, 1802)	Álvarez & Villalobos (1997)
	Pachygrapsus	*gracilis* (De Saussure, 1858)	Álvarez & Villalobos (1997)
	Aratus	*pisonii* (H. Milne Edwards, 1837)	Álvarez & Villalobos (1997)
	Sesarma	*curacaoense* De Man, 1892	Álvarez & Villalobos (1997)
	Uca	*rapax* (S. I. Smith, 1870)	Álvarez & Villalobos (1997)

Table 2. (Continuation)

	Genus	Species	References
DECAPODS			
	Uca	*vocator* (J. F. W. Herbst, 1804)	Álvarez & Villalobos (1997)
	Potimirim	*mexicana* (De Saussure, 1857)	Villalobos & Álvarez (1997)
INSECTS			
ODONATES			
	Hetaerina	*titia* (Drury, 1773)	González-Soriano (1997)
	Neoneura	*amelia* Calvert, 1903	González-Soriano (1997)
	Erythemis	*haematogastra* (Burmeister, 1839)	González-Soriano (1997)

2.2. Vertebrates

Table 3 lists the vertebrate species that have been reported for the Sontecomapan Lagoon. There are a total of 185 species: 159 birds, 14 hosting-parasite fish,[4] 1 amphibian, 8 reptiles and 3 mammals.

2.2.1. Birds

Rappole *et al.* (1997) reviewed the neartic migratory birds reported from Los Tuxtlas mountain chain and the Papaloapan basin wetlands. The list included three species from the Sontecomapan Lagoon: *Aythya americana* and *Falco peregrinus* as winter residents, and *Pluvialis dominica* as passerby. The authors defined the period from November 1 to February 28 as winter.

The nesting of the *Buteogallus anthracinus* black hawk was studied by Barradas-García *et al.* (2004). They found that for construction of the nest *B. anthracinus* uses long branches from the three species of mangrove (*Rhizophora mangle*, *Laguncularia racemosa* and *Avicennia germinans*) present in the lagoon and for the its center it uses the litter which comes mainly from *Avicennia germinans*.

In the lagoon area a total of 156 species of birds were reported in Monroy-Ojeda & Isern (2013).

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[4] In another chapter, Castillo-Rivera and Lara-Domínguez reviewed historical records to evaluate the fish species richness of Sontecomapan Lagoon.

2.2.2. Reptiles

The presence of the *Staurotypus triporcatus* turtle (Wiegmann, 1828) in Sontecomapan Lagoon was reported (Vogt *et al.*, 1997).

Carmona-Díaz *et al.* (2004) pointed out the coexistence of the *Lutjanus cyanopterus* fish, the *Crocodylus moreletii* crocodile, the green Iguana (*Iguana iguana*) and at least four species of turtles: *Staurotypus triporcatus*, *Claudius angustatus*, *Trachemys scripta* and *Kinosternon leucostomum* at the mangroves. They also refer the presence of the *Charadrius melodus* and *Buteogallus anthracinus* birds, and the *Lontra longicaudis* otter.

To the best of our knowledge, there are no published works that list the amphibians and mammals species for the Sontecomapan Lagoon.

Table 3. Species of vertebrates reported for the Sontecomapan Lagoon, Veracruz

	Genus	Species	References
Birds			
	Dendrocygna	*autumnalis* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Cairina	*moschata* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Anas	*americana* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Anas	*discors* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Aythya	*americana* (Eyton, 1838)	Rappole *et al.* (1997); Monroy-Ojeda & Isern (2013)
	Aythya	*affinis* (Eyton, 1838)	Monroy-Ojeda & Isern (2013)
	Oxyura	*jamaicensis* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Ortalis	*vetula* (Wagler, 1830)	Monroy-Ojeda & Isern (2013)
	Crax	*rubra* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Gavia	*immer* (Brunnich, 1764)	Monroy-Ojeda & Isern (2013)
	Tachybaptus	*dominicus* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Podilymbus	*podiceps* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Podiceps	*nigricollis* (C. L. Brehm, 1831)	Monroy-Ojeda & Isern (2013)
	Fregata	*magnificens* (Mathews, 1914)	Monroy-Ojeda & Isern (2013)
	Phalacrocorax	*brasilianus* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Phalacrocorax	*auritus* (Lesson, 1831)	Monroy-Ojeda & Isern (2013)
	Pelecanus	*occidentalis* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Botaurus	*pinnatus* (Wagler, 1829)	Monroy-Ojeda & Isern (2013)
	Tigrisoma	*mexicanum* (Swainson, 1834)	Monroy-Ojeda & Isern (2013)
	Ardea	*herodias* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013); Sereno-Uribe *et al.* (2013)

Table 3. (Continuation)

	Genus	Species	References
	Ardea	*alba* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
Birds			
	Egretta	*thula* (Molina, 1782)	Monroy-Ojeda & Isern (2013)
	Egretta	*caerulea* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Egretta	*tricolor* (Statius Muller, 1776)	Monroy-Ojeda & Isern (2013)
	Bubulcus	*ibis* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Butorides	*virescens* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Nycticora	*nycticorax* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Nyctanassa	*violacea* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Cochlearius	*cochlearius* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Eudocimus	*albus* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Plegadis	*chihi* (Vieillot, 1817)	Monroy-Ojeda & Isern (2013)
	Coragyps	*atratus* (Bechstein, 1793)	Monroy-Ojeda & Isern (2013)
	Cathartes	*aura* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Cathartes	*burrovianus* (Cassin, 1845)	Monroy-Ojeda & Isern (2013)
	Pandion	*haliaetus* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Elanus	*leucurus* (Vieillot, 1818)	Monroy-Ojeda & Isern (2013)
	Chondrohierax	*uncinatus* (Temminck, 1822)	Monroy-Ojeda & Isern (2013)
	Rostrhamus	*sociabilis* (Vieillot, 1817)	Monroy-Ojeda & Isern (2013)
	Circus	*cyaneus* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Accipiter	*striatus* (Vieillot, 1808)	Monroy-Ojeda & Isern (2013)
	Buteogallus	*anthracinus* (Deppe, 1830)	Barradas *et al.* (2004); Carmona-Díaz *et al.* (2004); Monroy-Ojeda & Isern (2013)
	Buteogallus	*urubitinga* (Gmelin, 1788)	Monroy-Ojeda & Isern (2013)
	Rupornis	*magnirostris* (Gmelin, 1788)	Monroy-Ojeda & Isern (2013)
	Laterallus	*ruber* (P. L. Sclater & Salvin, 1860)	Monroy-Ojeda & Isern (2013)
	Laterallus	*exilis* (Temminck, 1831)	Monroy-Ojeda & Isern (2013)
	Aramides	*axillaris* (Lawrence, 1863)	Monroy-Ojeda & Isern (2013)
	Aramides	*cajanea* (Statius Muller, 1776)	Monroy-Ojeda & Isern (2013)
	Fulica	*americana* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Aramus	*guarauna* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Pluvialis	*dominica* (Statius Muller, 1776)	Rappole *et al.* (1997)
	Pluvialis	*squatarola* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Himantopus	*mexicanus* (Statius Muller, 1776)	Monroy-Ojeda & Isern (2013)
	Jacana	*spinosa* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)

Table 3. (Continuation)

	Genus	Species	References
	Actitis	*macularius* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
Birds			
	Calidris	*alba* (Pallas, 1764)	Monroy-Ojeda & Isern (2013)
	Leucophaeus	*atricilla* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Hydroprogne	*caspia* (Pallas, 1770)	Monroy-Ojeda & Isern (2013)
	Thalasseus	*maximus* (Boddaert, 1783)	Monroy-Ojeda & Isern (2013)
	Thalasseus	*sandvicensis* (Latham, 1787)	Monroy-Ojeda & Isern (2013)
	Columba	*livia* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Patagioenas	*cayennensis* (Bonnaterre, 1792)	Monroy-Ojeda & Isern (2013)
	Patagioenas	*flavirostris* (Wagler, 1831)	Monroy-Ojeda & Isern (2013)
	Columbina	*inca* (Lesson, 1847)	Monroy-Ojeda & Isern (2013)
	Columbina	*talpacoti* (Temminck, 1810)	Monroy-Ojeda & Isern (2013)
	Leptotila	*verreauxi* (Bonaparte, 1855)	Monroy-Ojeda & Isern (2013)
	Piaya	*cayana* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Crotophaga	*sulcirostris* (Swainson, 1827)	Monroy-Ojeda & Isern (2013)
	Glaucidium	*brasilianum* (Gmelin, 1788)	Monroy-Ojeda & Isern (2013)
	Nyctidromus	*albicollis* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Streptoprocne	*zonaris* (Shaw, 1796)	Monroy-Ojeda & Isern (2013)
	Chaetura	*vauxi* (J. K. Townsend, 1839)	Monroy-Ojeda & Isern (2013)
	Amazilia	*candida* (Bourcier & Mulsant, 1846)	Monroy-Ojeda & Isern (2013)
	Amazilia	*tzacatl* (De la Llave, 1833)	Monroy-Ojeda & Isern (2013)
	Amazilia	*yucatanensis* (Cabot, 1845)	Monroy-Ojeda & Isern (2013)
	Trogon	*melanocephalus* (Gould, 1836)	Monroy-Ojeda & Isern (2013)
	Trogon	*caligatus* (Gould, 1838)	Monroy-Ojeda & Isern (2013)
	Momotus	coeruliceps (Gould, 1836)	Monroy-Ojeda & Isern (2013)
	Megaceryle	*torquata* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Megaceryle	*alcyon* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Chloroceryle	*amazona* (Latham, 1790)	Monroy-Ojeda & Isern (2013)
	Chloroceryle	*americana* (Gmelin, 1788)	Monroy-Ojeda & Isern (2013)
	Chloroceryle	*aenea* (Pallas, 1764)	Monroy-Ojeda & Isern (2013)
	Ramphastos	*sulfuratus* (Lesson, 1830)	Monroy-Ojeda & Isern (2013)
	Melanerpes	*aurifrons* (Wagler, 1829)	Monroy-Ojeda & Isern (2013)
	Dryocopus	*lineatus* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Campephilus	*guatemalensis* (Hartlaub, 1844)	Monroy-Ojeda & Isern (2013)
	Micrastur	*semitorquatus* (Vieillot, 1817)	Monroy-Ojeda & Isern (2013)

Table 3. (Continuation)

	Genus	Species	References
	Caracara	*cheriway* (Jacquin, 1784)	Monroy-Ojeda & Isern (2013)
Birds			
	Herpetotheres	*cachinnans* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Falco	*femoralis* (Temminck, 1822)	Monroy-Ojeda & Isern (2013)
	Falco	*peregrinus* (Tunstall, 1771)	Rappole *et al.* (1997)
	Falco	*sparverius* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Aratinga	*nana* (Vigors, 1830)	Monroy-Ojeda & Isern (2013)
	Amazona	*albifrons* (Sparrman, 1788)	Monroy-Ojeda & Isern (2013)
	Amazona	*autumnalis* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Amazona	*oratrix* (Ridgway, 1887)	Monroy-Ojeda & Isern (2013)
	Thamnophilus	*doliatus* (Linnaeus, 1764)	Monroy-Ojeda & Isern (2013)
	Synallaxis	*erythrothorax* (P. L. Sclater, 1855)	Monroy-Ojeda & Isern (2013)
	Camptostoma	*imberbe* (P. L. Sclater, 1857)	Monroy-Ojeda & Isern (2013)
	Todirostrum	*cinereum* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Contopus	*virens* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Empidonax	*minimus* (W. M. Baird & S. F. Baird, 1843)	Monroy-Ojeda & Isern (2013)
	Attila	*spadiceus* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Myiarchus	*tyrannulus* (Statius Muller, 1776)	Monroy-Ojeda & Isern (2013)
	Pitangus	*sulphuratus* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Megarynchus	*pitangua* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Myiozetetes	*similis* (Spix, 1825)	Monroy-Ojeda & Isern (2013)
	Tyrannus	*melancholicus* (Vieillot, 1819)	Monroy-Ojeda & Isern (2013)
	Tyrannus	*forficatus* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Tityra	*semifasciata* (Spix, 1825)	Monroy-Ojeda & Isern (2013)
	Pachyramphus	*aglaiae* (Lafresnaye, 1839)	Monroy-Ojeda & Isern (2013)
	Vireo	*griseus* (Boddaert, 1783)	Monroy-Ojeda & Isern (2013)
	Vireo	*solitarius* (A. Wilson, 1810)	Monroy-Ojeda & Isern (2013)
	Cyclarhis	*gujanensis* (Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Psilorhinus	*morio* (Wagler, 1829)	Monroy-Ojeda & Isern (2013)
	Stelgidopteryx	*serripennis* (Audubon, 1838)	Monroy-Ojeda & Isern (2013)
	Troglodytes	*aedon* (Vieillot, 1809)	Monroy-Ojeda & Isern (2013)
	Campylorhynchus	*zonatus* (Lesson, 1832)	Monroy-Ojeda & Isern (2013)
	Pheugopedius	*maculipectus* (Lafresnaye, 1845)	Monroy-Ojeda & Isern (2013)
	Ramphocaenus	*melanurus* (Vieillot, 1819)	Monroy-Ojeda & Isern (2013)
	Polioptila	*caerulea* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)

Table 3. (Continuation)

	Genus	Species	References
	Turdus	*grayi* (Bonaparte, 1838)	Monroy-Ojeda & Isern (2013)
Birds			
	Dumetella	*carolinensis* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Helmitheros	*vermivorum* (J. F. Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Parkesia	*motacilla* (Vieillot, 1809)	Monroy-Ojeda & Isern (2013)
	Mniotilta	*varia* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Oreothlypis	*celata* (Say, 1822)	Monroy-Ojeda & Isern (2013)
	Geothlypis	*poliocephala* (S. F. Baird, 1865)	Monroy-Ojeda & Isern (2013)
	Geothlypis	*trichas* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Setophaga	*citrina* (Boddaert, 1783)	Monroy-Ojeda & Isern (2013)
	Setophaga	*ruticilla* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Setophaga	*americana* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Setophaga	*magnolia* (A. Wilson, 1811)	Monroy-Ojeda & Isern (2013)
	Setophaga	*petechia* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Setophaga	*virens* (J. F. Gmelin, 1789)	Monroy-Ojeda & Isern (2013)
	Basileuterus	*rufifrons* (Swainson, 1838)	Monroy-Ojeda & Isern (2013)
	Cardellina	*pusilla* (A. Wilson, 1811)	Monroy-Ojeda & Isern (2013)
	Icteria	*virens* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Thraupis	*episcopus* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Thraupis	*abbas* (Deppe, 1830)	Monroy-Ojeda & Isern (2013)
	Volatinia	*jacarina* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Sporophila	*torqueola* (Bonaparte, 1850)	Monroy-Ojeda & Isern (2013)
	Saltator	*atriceps* (Lesson, 1832)	Monroy-Ojeda & Isern (2013)
	Piranga	*rubra* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Habia	*fuscicauda* (Cabanis, 1861)	Monroy-Ojeda & Isern (2013)
	Cardinalis	*cardinalis* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)
	Passerina	*cyanea* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Agelaius	*phoeniceus* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Dives	*dives* (Deppe, 1830)	Monroy-Ojeda & Isern (2013)
	Quiscalus	*mexicanus* (Gmelin, 1788)	Monroy-Ojeda & Isern (2013)
	Molothrus	*aeneus* (Wagler, 1829)	Monroy-Ojeda & Isern (2013)
	Icterus	*spurius* (Linnaeus, 1766)	Monroy-Ojeda & Isern (2013)
	Icterus	*mesomelas* (Wagler, 1829)	Monroy-Ojeda & Isern (2013)
	Icterus	*gularis* (Wagler, 1829)	Monroy-Ojeda & Isern (2013)
	Icterus	*galbula* (Linnaeus, 1758)	Monroy-Ojeda & Isern (2013)

Table 3. (Continuation)

	Genus	Species	References
	Psarocolius	*montezuma* (Lesson, 1830)	Monroy-Ojeda & Isern (2013)
Birds			
	Euphonia	*affinis* (Lesson, 1842)	Monroy-Ojeda & Isern (2013)
	Euphonia	*hirundinacea* (Bonaparte, 1838)	Monroy-Ojeda & Isern (2013)
	Charadrius	*melodus* (Ord, 1824)	Carmona-Díaz *et al.* (2004)
Marsupials			
	Didelphis	*virginiana* (Kerr, 1792)	Salgado-Maldonado & Cruz-Reyes (2002); García-Prieto *et al.* (2010)
	Philander	*oppossum* (Linnaeus, 1758)	Lamothe-Argumedo *et al.* (1997)
Fish (only fish hosting parasites)			
	Arius	*melanopus* (Günther, 1864)	Lamothe-Argumedo *et al.* (1997)
	Bagre	*marinus* (Mitchill, 1815)	García-Prieto *et al.* (2010)
	Cathorops	*melanopus* (Günther, 1864)	Salgado-Maldonado (1978); García-Prieto *et al.* (2010)
	Centropomus	*undecimalis* (Bloch, 1792)	García-Prieto *et al.* (2010)
	Diapterus	*auratus* (Ranzani, 1842)	Lamothe-Argumedo *et al.* (1997); García-Prieto *et al.* (2010)
	Diapterus	*olisthostomus* (Goode & Bean, 1882)	Lamothe-Argumedo *et al.* (1997); Salgado-Maldonado (1977)
	Dormitator	*maculatus* (Bloch, 1792)	Sereno-Uribe *et al.* (2013); Pinacho-Pinacho *et al.* (2014)
	Eugerres	*plumieri* (Cuvier in Cuvier and Valenciennes, 1830)	Lamothe-Argumedo *et al.* (1997); García-Prieto *et al.* (2010)
	Lutjanus	*cyanopterus* (Cuvier in Cuvier and Valenciennes, 1828)	Carmona-Díaz *et al.* (2004)
	Mugil	*cephalus* (Linnaeus, 1758)	Lamothe-Argumedo *et al.* (1997); García-Prieto *et al.* (2010); Rosas-Valdez *et al.* (2012)
	Pomadasys	*crocro* (Cuvier in Cuvier and Valenciennes, 1830)	Salgado-Maldonado (1976); García-Prieto *et al.* (2010)
	Caranx	*hippos* (Linnaeus, 1766)	Lamothe-Argumedo *et al.* (1997)
	Trachinotus	*carolinus* (Linnaeus, 1766)	Lamothe-Argumedo *et al.* (1997)
	Scomberomorus	*maculatus* (Mitchill, 1815)	Lamothe-Argumedo *et al.* (1997)
Amphibians			
	Leptodactylus	*melanonotus* (Hallowell, 1861)	Razo-Mendivil & León-Règagnon (2001); Paredes-León *et al.* (2008)
Reptiles			
	Bothrops	*asper* (Garman, 1884)	Peláez & Juliá (1983); Paredes-León *et al.* (2008)
	Crocodylus	*moreletii* (Duméril and Bibron, 1851)	Carmona-Díaz *et al.* (2004)

Table 3. (Continuation)

	Genus	Species	References
	Iguana	*iguana* (Linnaeus, 1758)	Carmona-Díaz *et al.* (2004)
Reptiles			
	Staurotypus	*triporcatus* (Wiegmann, 1828)	Vogt *et al.* (1997); Carmona-Díaz *et al.* (2004)
	Claudius	*angustatus* (Cope, 1865)	Carmona-Díaz *et al.* (2004)
	Trachemys	*scripta* (Thunberg in Schoepff, 1792)	Carmona-Díaz *et al.* (2004)
	Kinosternon	*leucostomum* (Duméril & Bibron in Duméril & Duméril, 1851)	Carmona-Díaz *et al.* (2004)
Otter			
	Lontra	*longicaudis* (Olfers, 1818)	Carmona-Díaz *et al.* (2004)

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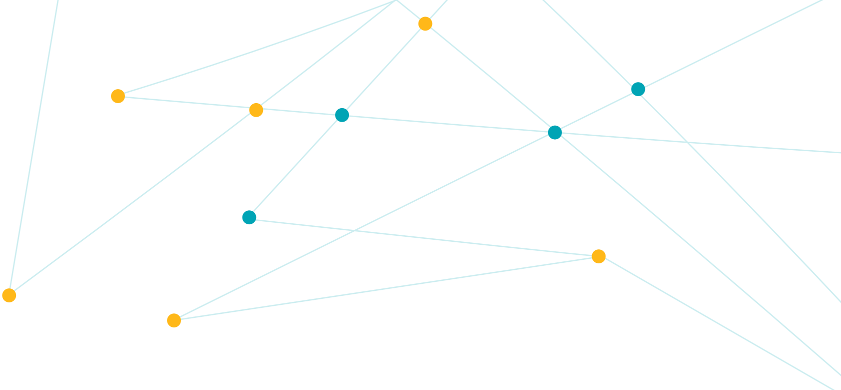

PART 3

Microbial components

Bacterial community contribution to nitrogen fixation and nitrous oxides production in the Sontecomapan Lagoon

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Bacterial community contribution to nitrogen fixation and nitrous oxides production in the Sontecomapan Lagoon

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Abstract: Nitrogen fixation and denitrification are processes poorly studied in Mexican coastal lagoons. In this study the N incorporation and loss rates of the bottom water and sediments were analyzed in six zones of Sontecomapan lagoon during "Nortes" and rainy seasons and their relation with the rivers outflows and the coastal marine waters entrance. The rate of N_2 fixation was calculated through acetylene reduction and the denitrification through N_2O production techniques. The N_2 fixation was lower than the nitrogen loss as N_2O production (denitrification). The greatest fixation rate was registered in the "Nortes" season; being favored by oxic conditions and high organic matter concentrations in the sediments. In the rainy season, the N_2 fixation was lowered by 80% compared to the "Nortes" season. Denitrification was favored by low salinities in November and was 23% higher than in February, 2005. In the rainy season the nitrous oxide production was related to the high $N\text{-}NO_3^-$ concentrations and decreased by 18% when the salinity increased from 13 to 22 PSU.

Keywords: Coastal lagoon, denitrification rate, bottom water, N_2 fixation rate, sediment.

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Introduction

The coastal waters are continuously enriched with anthropogenic inorganic nutrients, from polluting sources as wastewaters generated by peripheral human settlements or by farming activities. As they are shallow ecosystems it is assumed that the processes that occur it the water-sediment interface can control the biological processes that happen at the water column, thus influencing the system productivity (Capone, 1983; Day *et al.*, 1989).

Microbial metabolic processes play an essential role in the mineralization of the organic matter that is deposited into the sediments, thus enriching the interstitial water with soluble forms of nitrogen (NH_4^+, NO_2^-, NO_3^-), phosphorus (HPO_4^{2-}), sulphur (SH^-, SO_4^{2-}, H_2S) and iron (Fe^{2+}, Fe^{3+}); these produced ions are then transported towards supernatant waters by diffusion and by biological processes. The nitrogen (N) cycle is a key biogeochemical process in which transformations of multiple nitrogen compounds are mainly driven by bacterial activity. The cycle controls the nitrogen nutrient availability for biological productivity in aquatic systems, as well as, atmospheric CO_2 fixation and biogenic carbon dioxide export from the surface layer towards the atmosphere (Zehr & Ward, 2002).

The N enrichment of the water column may be caused by rainfalls and supply due to the superficial and underground drainage as well as by biological fixation. The loss of N may be due to the discharges of lagoon waters towards the coastline under current effects and/or the reduction of nitrate to molecular nitrogen (N_2), through denitrifying bacterial activity with the subsequent return of N to the atmosphere, and the permanent loss of organic and inorganic nitrogen compounds by deposition process (Wetzel, 1981; Andersen *et al.*, 2004; Liu & Qiu, 2007). The microbial nitrogen fixation is the most important process that turns atmospheric nitrogen to fixed nitrogen, whereas bacterial denitrification is the most significant process of molecular nitrogen regeneration.

The biological nitrogen fixation is confined to specialized groups of prokaryotes, which own the nitrogenase enzyme and can have heterotrophic or autotrophic metabolism. There is a wide variety of factors that can affect N_2 fixation, no matter the type of ecosystem: oxygen, pH, temperature, light, salinity, inorganic nitrogen and organic substrates available, all of them can govern the nitrogenase activity in specific conditions (Viner, 1982; Hebst, 1998; Evans *et al.*, 2000). Diverse methods have been used to quantify the N_2 fixation rate, one of them is the stable isotopes technique (^{15}N) however, the more widely used is acetylene reduction of (C_2H_2) to ethylene (C_2H_4) (Larkum *et al.*, 1988; Herbert, 1999; Falcón *et al.*, 2002, 2007). The C_2H_2 reduction rate, compared with the N_2 incorporation rate to the cellular biomass, has a conversion factor of approximately 3 moles of C_2H_4 formed with respect to 1 mole of fixed N_2 (Atlas & Bartha, 2002).

In coastal environments, most of the recycled N is in the form of ammonium (NH_4^+). This NH_4^+ is formed by bacterial decomposition and deamination of organic matter, and it diffuses from the sediments towards the superficial layer of the water column where it can be assimilated by phytoplankton. In presence of oxygen, part of the regenerated NH_4^+ is oxidized to nitrate (NO_3^-), which can be used as terminal electron acceptor by the denitrifying bacteria (*Pseudomonas, Thiobacillus, Thiosphaera*) producing gaseous forms of N (N_2O, N_2) (Kemp *et al.*, 1990).

On the other hand, in the sediment, the vertical distribution and abundance of N species are governed by the redox state of the different substrates thus, in the sediment oxygenated superficial layer, environmental ammonification and nitrification processes will be more important than those of denitrification. In the sediments, temperature, NH_4^+ concentration, pH, dissolved CO_2, salinity, macrofauna activity and macrophytes presence are also important for regulating N_2 fixation, nitrification and denitrification processes.

The denitrification is especially active in shallow waters that present anaerobic zones rich in organic matter (Thamdrup & Dalsgaard, 2002; Madigan *et al.*, 2004). It is considered as one of the most important mechanisms in coastal waters biogeochemistry because it is energetically the most favorable form of anaerobic metabolism and also removes significant amounts of NO_3^- therefore, denitrification can influence primary productivity and seems to act as a "buffer" system, preventing the excessive NO_3^- increase in water bodies (Groffman, 1994). Generally, the zone of denitrification is typically located at few millimeters below the oxic superficial sediment layer, where there are limited oxygen concentrations (Ferrara & Bianchi 1990; Revsbech *et al.*, 1980). Some of the current techniques for evaluating denitrification rates are based on the quantification of N_2O formation in anaerobic incubations; this can be obtained by inhibiting the passage from N_2O to N_2 by non-competitive N_2O-reductase enzyme inhibition using acetylene (C_2H_2) (Taylor, 1983; Seitzinger, 1988).

The importance of N as a primary regulator for productivity has induced to the examination of the N-metabolism nature and importance in the coastal benthic communities and the establishment of these activities connection with the pelagic ecosystem (Herrera-Martinez *et al.*, 2000).

The denitrification rate is controlled by the nitrification one, which supplies NO_3^- as a substrate. Thus, the connection process between nitrification and denitrification represents a way to turn nitrogen outside of the recycling routes due to the loss of N in gaseous form towards the atmosphere. This connection is quantitatively important for N cycle in the coastal continental and estuarine sediments, where the N lost by denitrification could represent around half of the entries for continental contributions (Seitzinger, 1988).

In addition to the nitrification process, there are two other main sources for NO_3^-: diffusion towards the sediments from the water column, and its transport through underground waters (Capone & Bautista, 1985; Seitzinger, 1988; Day *et al.*, 1989).

The study of the N biogeochemical cycle in coastal lagoons is fundamental to elucidate the role of these ecosystems within the global carbon cycle and the global climate change, since it has a relevant role in the water column productivity and eutrophication. In the same way, the biological N_2 fixation helps to recover the N lost by denitrification in the coastal ecosystems and providing new nitrogen.

Currently in Mexico, there are few scientific reports on N_2 fixation and denitrification processes in aquatic ecosystems. Although both processes are major components for nitrogen cycling in coastal lagoons, only some of these studies consider nitrogen fixation and denitrification processes together (Valenzuela-Siu *et al.*, 2007; Hakspiel-Segura, 2014).

Because of the importance of N species within Mexican lagoon ecosystems, the aim of this investigation was to analyze the incorporation and loss processes of N species from the bottom water as well as from superficial sediments, in order to contribute to a better understanding of the nitrogen cycle functioning in Sontecomapan Lagoon.

Materials and Methods

Study Site and sampling

Six sampling sites were selected according to the salt and fresh water interchanges from the coastline and the rivers flowing into the lagoon: Arroyo La Boya (near the pass of the lagoon to the sea). La Palma and Sábalo (both near the wouhs of small rivers). El Cocal and Punta Levisa (on the navigation canal) and Costa Norte in the interior of the lagoon and a zone of submerged vegetation (*Ruppia maritima*) (Figure 1). Samplings were performed at three dates (November, 2004, and February and June, 2005) corresponding to two seasons of the year: "Nortes" winds (November and February) and rains (June).

The bottom water samples were obtained with a horizontal one-liter Van Dorn bottle. The oxygen concentration was determined by the Winkler technique (Aminot & Chaussepied, 1983). The salinity was measured with an ATAGOS MILL-E refractometer (± 1‰) and its pH with an Orion potentiometer. The water samples for the determination of nitrogen inorganic species ($N\text{-}NH_4^+$, $N\text{-}NO_2^-$ and $N\text{-}NO_3^-$) were filtered through 45 mm diameter Whatman GF/F membranes and deep frozen immediately until their processing, following the colorimetric methods recommended by Aminot & Chaussepied (1983).

The sediment samples were collected by diving, using 22 x 5cm inner diameter (i. d.) polycarbonate corers, avoiding disturbance of the sediment column. The pH and Eh were measured using ORION (2mm and 3mm i. d. respectively) needle-type mini-electrodes, in order to avoid disturbing the sediment during penetration they were inserted in a stepped way each 5 mm of depth. The interstitial water for measuring the nutrients was obtained from the first centimeter of sediment, using a polycarbonate capillary tube (10 cm length and 5 mm i.d.) sealed in its lower bound and pierced with a series of 1 mm i.d. holes throughout their last centimeter. The interstitial water from the first centimeter of depth was obtained by suction using a 50 mL syringe connected to the capillary tube by a latex hose. The water samples were filtered with a 25 mm diameter Whatman GF/F membrane and stored in 60 ml hermetic bottles previously gasified with N_2 to prevent oxidation from the samples' air, and then deep frozen to -18 °C until their processing.

The remaining sediment from the cores was used for determining the percentage of organic matter (%OM) and sediment texture. Replicated cores were sliced into 1 cm thick layers and treated separately for %OM, using the potassium dichromate excess titration method, with a 0.5N solution of ferrous sulphate for organic matter oxidation (Gaudette *et al.*, 1974) and the sediment texture by Bouyoucos' hydrometer method (Holme & McIntyre, 1984).

Incubations for N_2 fixation and denitrification rates

The bottom water samples were obtained with a horizontal Van Dorn bottle and 300 mL from the sample were incubated in 500 mL hermetic glass jars. The first centimeter of the sediment was placed in another series of jars containing 300 mL of bottom water previously filtered through Whatman GF/F membranes. The incubations were performed *in situ* (two clear bottles and one dark) for four hours, one series by test. At the beginning of the incubations, 10% of the air volume from each jar was extracted through a cork with turnover septum stoppers —folding rubber apron— (Fisher Scientific) placed in jar cover (Moran-Villa *et al.*, 2009), and the same volume of acetylene was injected into the jar (C_2H_2). The jars stayed within a water bath to keep them at room temperature.

N_2 fixation measurements

After the four-hour incubation period, a sub-sample from the gaseous phase was taken and stored in a hermetically closed vial. The N_2 fixation rate was estimated by acetylene reduction technique (ARA) (Capone, 1993; Morán-Villa *et al.*, 2009). The ethylene analyses (C_2H_4) were performed with a Perkin Elmer Auto Analyzer gas chromatograph with a flame ionization detector (FID) equipped with a 15 x 0.55 mm i.d. and 1 mm thick RESTEK Corporation capillary column. The temperature of the oven, injector and detector was 165 °C and 230 °C respectively. As a gas carrier, a 1 mL min^{-1} He flow was used. A 100 μl aliquot from C_2H_2/C_2H_4 gas sample was injected. The gas chromatograph was calibrated with a standard 100 ppm C_2H_4, and the reduction rate of C_2H_2 was converted to N-fixed using the conventional molar conversion factor 3:1 proposed by Postgate (1982) and Seitzinger & Garber (1987), and the Bunsen coefficient of solubility (Flett *et al.*, 1976). The total fixation of N_2 rate was calculated by adding the phototrophic fixation rate (in the light) to the heterotrophic one (in darkness).

N_2O measurements

The analysis of N_2O present in the gaseous phase was made with a VARIAN chromatograph with a flame ionization electron capture detector ^{63}Ni. As a gas carrier, a 30 mL min^{-1} He flow was used. The temperatures of the column and the detector were 50 °C and 300 °C respectively. A 1 ml aliquot was injected and the peak area was measured with a peak integrator. A standard of 100 ppm N_2O (Alltech) and the Bunsen coefficient of solubility (Flett *et al.*, 1976) were used.

$$\mu mol\left(N_2O\ l^{-1}\ t^{-1}\right)=\left\{\left(\frac{peak\ area\ sample}{peak\ area\ standard}\right)\times\left(\frac{C.S.\ injected\left(Vgas\times VH_2O\times 0.1\right)}{Vol.injected}\right)\right\}/\Delta t$$

where:

C.S. = Coefficient of N_2O standard
V_{gas} = Volume of the gas phase in the incubation bottle
V_{H20} = Volume of incubated water (or sediment, as the case might be)
0.1 = Bunsen coefficient for nitrous oxide at the appropriate salinity and temperature
Δt = Incubation time
The volume of the gas phase injected into the gas chromatograph was 100 μl

Statistical analysis

In order to determine the physical and chemical factors that influence the N_2 fixation (FixN) and denitrification (N_2O) rates during the “Nortes” and the rainy seasons, a Canonic Correspondence Analysis (CCA) using Multi-Variate Statistical Package (MVSP) 3.22 ® program was performed on the $\log_e$ standardized data, using the centered data options and the Kaiser rule for the axes extraction (Kovach, 1999).

Results

Physical and chemical parameters

Bottom water

Table 1 shows the mean values of oxygen, temperature, pH and salinity at the bottom water of the six stations. The bottom water was found in suboxic conditions but with acceptable values for most of the pelagic organisms. Oxygen varied between 2.6 and 10.1 mg/l^{-1} in the zones with greater boat traffic. The pH was around neutrality all year long (7.23 ± 0.55). Temperatures varied between 20 and 30 °C. Salinity varied from 0.2 to 37 PSU (mean value = 19.6 ± 11.4 PSU), with the lowest values measured in November (from 0.2 to 28 PSU). The Eh values were highly variable but generally positive (Table 1).

$N\text{-}NH_4^+$ concentration had mean values of 3.98 ± 3.21 μmol l^{-1}, with the lowest values registered at La Boya and Costa Norte (0.01 and 0.39 μmol l^{-1} respectively) and the highest at Punta Levisa and La Palma (10.6 and 7.9 μmol l^{-1} respectively) during "Nortes" season (February). In general, the ammonium concentration was 6 times higher in February that in November (6. 08 ± 3.3 and 1.04 ± 0.65 μmol l^{-1} respectively). During the three surveys, the $N\text{-}NO_2^-$ concentrations fluctuated between 0.15 and 9.0 μmol l^{-1} (La Boya and El Cocal, respectively); the low concentrations of this nutrient in the bottom water could be due to an intense ammonium oxidation to nitrate. The high concentration of $N\text{-}NO_2^-$ found in El Cocal was due to the effect of the sediment re-suspension provoked by the constant boat transit in the navigation canal, and it is in this same sampling area where the smallest $N\text{-}NO_3^-$ concentration was obtained (0.8 μmol l^{-1}) in February. In June (rain season), high concentrations of $N\text{-}NO_3^-$ in all the sampling areas (81±35.5 μmol l^{-1}) were generally registered.

Sediments

The sediments were of the silty-sandy type, moderately rich in organic matter (1.73 ± 1.5%), with the highest values in the sampling sites near the two river mouths (Table 2). The Eh varied from -240.9 to 291 mV.

The highest concentrations of ammonium were recorded during the "Nortes" period (November 2004, February 2005) at Punta Levisa (26.5 μmol l^{-1} $N\text{-}NH_4^+$), La Palma (22.6 μmol l^{-1} $N\text{-}NH_4^+$) and La Boya (25.2 μmol l^{-1} $N\text{-}NH_4^+$). Generally, ammonium concentrations in the sediment were higher than at the bottom water (Tables 1 and 2). Nevertheless, the diffusive flow of the sediment towards the water column sometimes allows leads to higher concentration in the bottom water than in the sediment (Tables 1 and 2).

Table 1. Variation of physical and chemical parameters in bottom water. Values obtained in the three samplings seasons. “Nortes” (November=N and February=F) and rains (June=J). nd=undetermined

Study Sites	O_2 (mg L^{-1})			Temperature (°C)			pH			Salinity (PSU)			Eh (mV)			$N\text{-}NH_4^+$ (μmol L^{-1})			$N\text{-}NO_2^-$ (μmol L^{-1})			$N\text{-}NO_3^-$ (μmol L^{-1})		
	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J
Costa Norte	9.2	5.3	9.7	25	27	30	6.9	6.9	7.5	14.5	24	11	-66.5	115.5	-99	0.39	7.4	3.9	2.2	0.55	0.37	14.4	1.8	121.4
Punta Levista	6.2	10.1	5.8	26	28	29	6.7	7.4	7.8	17.3	29	13	-192.3	291	-53	0.46	10.6	nd	3.7	1.6	nd	11.3	10.0	128.6
Sábalo River	6.8	6.1	2.6	25	25	32	6.9	6.8	7.7	12	12	1.5	nd	109	nd	0.48	5.9	6.1	2.6	2.3	0.17	8.2	6.9	64.3
El Cocal	6.6	7.3	4.9	24	25	29	6.8	7.4	7.6	28	31	30	nd	57	114	1.58	1.3	nd	1.8	9.0	nd	15.6	0.8	57.1
La Palma River	5.3	5.9	5.3	25	24	27	6.7	6.4	7.3	0.2	5	29	- 16.7	135	12	1.64	7.9	3.3	2.3	0.8	0.20	10.6	6.0	42.9
La Boya	8.4	6.9	5.5	20	21	27	6.7	8.4	8.1	25	37	33	-53.9	79	-6.6	1.66	3.5	0.01	2.1	1.3	0.15	14.1	8.7	71.4
Mean ± Sd	6.6±1.9			26.1± 3.0			7.2 ± 0.6			19.6 ±11.4			31.2 ±120.8			3.98 ± 3.21			1.95 ± 2.15			33.1 ± 40.01		

Table 2. Variation of physical and chemical parameters in sediments (1-cm depth). Values obtained in the three samplings seasons. “Nortes” (November =N and February=F, and rains (June=J). nd = undetermined

	Org. C (% dry weight)			Clays (%)			Silt (%)			Sand (%)			Salinity (PSU)			pH			Eh (mV)			N-NH_4^+ (μmol L^{-1})			N-NO_2^- (μmol L^{-1})			N-NO_3^- (μmol L^{-1})		
Study Sites	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J	N	F	J
Costa Norte	1.2	0.7	2.2	4	1	5	6	6	14	90	94	81	15	30	8	8.5	6.9	6.2	-67	100	-40	8	14	0.06	14	7	0.2	2	2	121
Punta Levisa	2.1	1.0	0.6	6	15	2	2	2	1	88	65	96	11	32	19	8.7	7.4	2.3	-192	291	-38	27	17	0.7	17	1	0.1	0.3	10	129
Sábalo River	2.1	5.9	3.4	nd	5	4	8	8	9	84	53	88	13	32	18	8.5	6.9	6.7	32	96	33	8	2	0.9	2	0.7	0.1	4	7	64
El Cocal	0.3	0.8	1.3	10	11	3	30	30	10	50	61	88	18	33	2	8.6	7.4	6.6	151	45	-26	8	5	0.01	5	0.4	0.7	3	0.8	57
La Palma River	3.6	0.5	1.4	2	15	14	10	10	4	84	63	94	4	32	0.2	7.1	6.3	6.9	-17	125	49	0.6	23	0.6	23	110	2	2	6	43
La Boya	2.6	0.5	3.0	6	9	3	10	10	7	82	69	91	23	32	33	7.7	8.4	7.4	-54	73	-241	25	2	4	2	1	1	2	9	71
Mean ± Sd	1.8 ± 1.4			6.8 ± 4.7			9.8 ± 8.1			78.9 ± 14.8			19.7 ± 11.6			7.1 ± 1.5			17.8 ± 122.4			8.1 ± 9.2			10.4 ± 25.7			29.6 ± 42.1		

The highest concentrations of N-NO_2^- were registered in November at La Palma (22.6 μmol l^{-1}), Punta Levisa (16.5 μmol l^{-1}) and Costa Norte (13.9 μmol l^{-1}) and were related to negative Eh, pH between 7.1 and 8.7 and high percentages of organic carbon (3.6, 2.1 and 1.2 % org C respectively). The highest concentrations of N-NO_3^- in sediment were registered in June 2005, at Costa Norte and Punta Levisa (121 and 129 μmol l^{-1} respectively). No significant difference in the concentrations of N-NO_3^- between the bottom water and the interstitial water of the sediments were found.

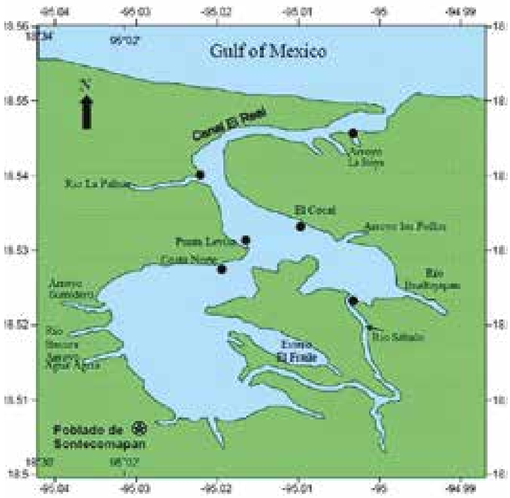

Figure 1. Sampling sites at the Sontecomapan Lagoon, Veracruz.

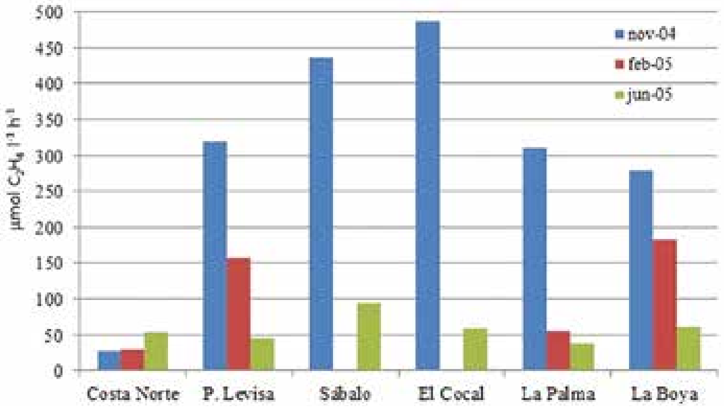

Figure 2. Variation of N_2 fixation rate of the bottom water during the two studied seasons, November, 2004 to February, 2005 ("Nortes" season) and June, 2005 (rainy season) at the Sontecomapan Lagoon.

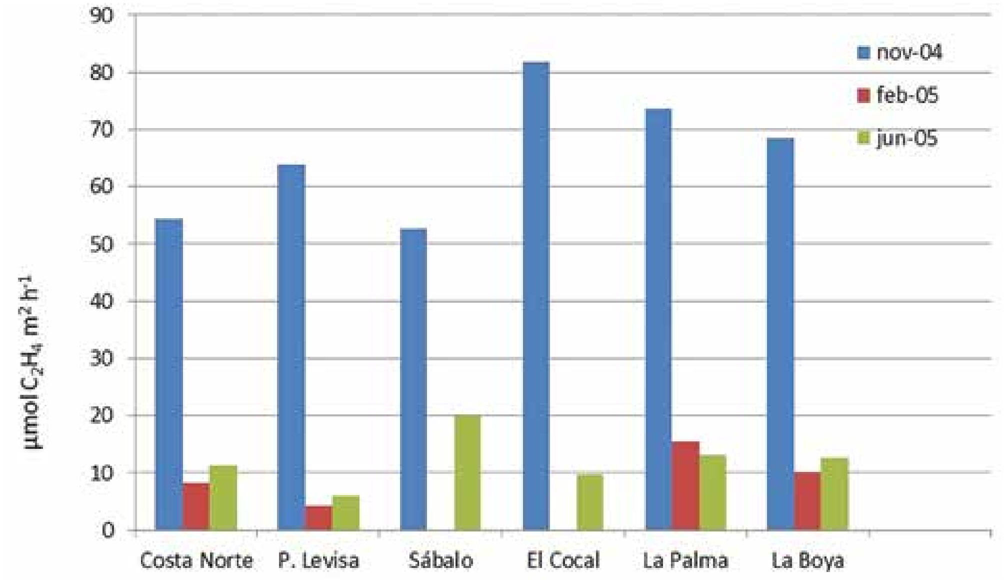

Figure 3. Variation of N_2 fixation rate in the sediment (1 cm deep) during the two seasons studied, November, 2004 to February, 2005 ("Nortes" season) and June, 2005 (rainy season) in the Sontecomapan Lagoon.

Nitrogen fixation

In all the sampling stations, nitrogenase activity was detected. In the bottom water, the highest N_2 fixation rates (309.98± 160.35 μmol C_2H_4 m^2 h^{-1}) were observed in November ("Nortes" season). A drastic decrease (81%) was observed in rainy season (58.72 ± 19.5 μmol C_2H_4 m^2 h^{-1}), when the highest salinity was noted at the bottom water (19.6 ± 12.8 PSU). In the sediments the fixation rate of N_2 was 10 times lower (31.62 ± 28.4) than in the bottom water (Fig. 3), and the lowest values were recorded in February and June, 2005 in Punta Levisa (4.17 and 5.98 μmol C_2H_4 m^2 h^{-1} respectively).

Denitrification rate

Generally, it was observed, as well as for the N_2 fixation, that the highest N_2O production occurred in November (764 ± 114,3 μmol N_2O m^2 h^{-1}), this being the reason why this process was favored by the low salinities (19,6 ± 11,06 PSU) and the highest oxygen concentrations registered (6.6±1.53 mg L^{-1}) at this season. The lowest rates were observed in February (593.9 ± 107.4 μmol N_2O m^2 h^{-1}), being 23% lower than those from November. During the rainy season, the denitrification rate varied between 465.5 and 1094 μmol N_2O h^{-1} m^2. At this season the highest values were obtained at La Boya (Fig. 4).

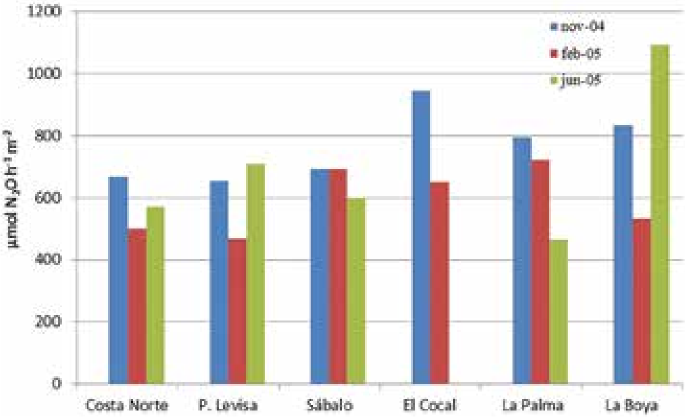

Figure 4. Denitrification rate in sediments (1 cm deep) measured by N_2O production during the two seasons studied, November, 2004 to February, 2005 ("Nortes" season) and June 2005 (rainy season) at the Sontecomapan Lagoon.

CCA analysis

The CCA analysis showed that in November the high N_2 fixation rate recorded at the bottom water had an inverse relationship with the slightly electronegative Eh (r=-0.663), and a positive relationship with N-NO_2^- (r=0.92). Within the same period, the high N_2 fixation rate in the sediments was also correlated to slightly electronegative Eh (r= -0.680), but also to high sand concentrations (r= -0.667), low ammonium concentrations (r= 0.696) low nitrates (r= -0.782) and high O_2 concentrations in bottom water (Fig. 5).

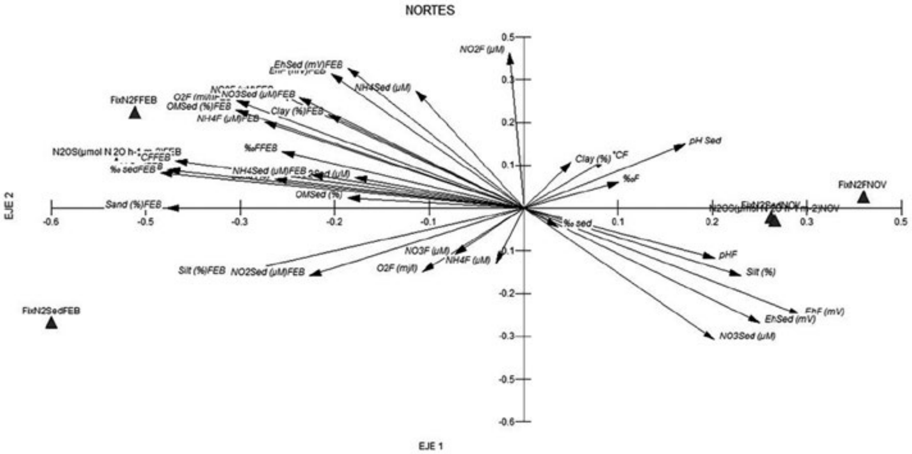

Figure 5. Multivariate CCA analysis performed for the "Nortes" season (November to February) from the 18 physical and chemical variables, which gives a contribution to the total variance of 98.46% in relation to the values of N_2fixation ($FixN_2$) and denitrification (N_2O) rates, in bottom water and sediment.

In February, the low N_2 fixation rate in bottom water was correlated positively with electropositive values of Eh (r=0.795), high O_2 (r=0.636), and negatively with high N-NH_4^+ concentrations (r= -0.696), low N-NO_2^- (r= -0.659) and low OM percentage (r= -0.77) in the sediment. Within the same period, the low N_2 fixation rates in the sediment were positively linked to the greatest percentage of silt (r=-0.788), and negatively to the lowest percentage of sands (r= -0.958), slightly basic pHs (r= -0.955) and low N-NO_3^- concentrations (r= -0.602) in the interstitial water. In November, the high N_2 fixation rate in the sediments (higher than in February) was correlated to slightly electronegative Eh (r=-0.680), high sand concentrations (r=-0.667), low ammonium concentrations (r=0.696), low nitrates (r=-0.782) and high O_2 concentrations in the bottom water (Fig. 5).

As for the N_2 fixation, the denitrification rate in the "Nortes" season, was correlated to slightly electronegative Eh values (r=-0.680), low N-NO_3^- concentrations in the interstitial water (r=0.782) and low salinities (r=0.98). The rainy season (June) was related to high N-NO_3^- concentrations in the interstitial water (Fig. 5 and 6).

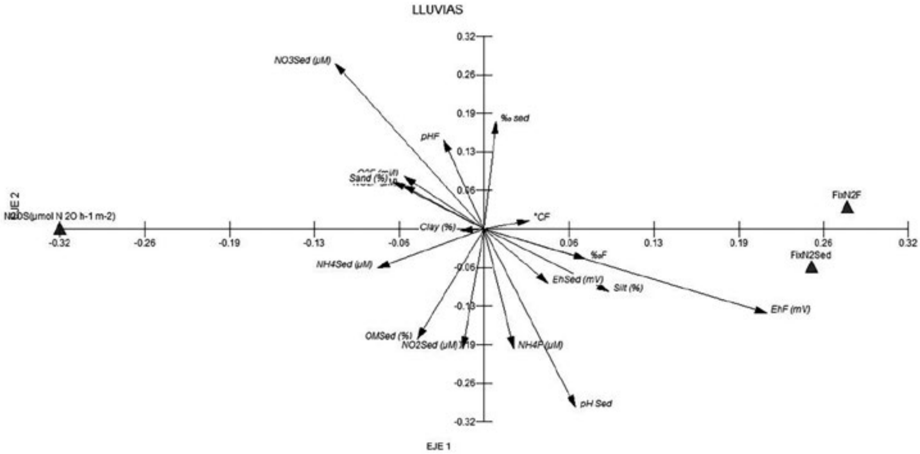

Figure 6. Multivariate CCA analysis performed for the rainy season (June) from the 18 physical and chemical variables, which gives a contribution to the total variance of 100% in relation to the values of N_2 fixation ($FixN_2$) and denitrification (N_2O) rates, in bottom water and sediment.

Discussion

Although it is considered that the N_2 fixation is a O_2-sensitive process, the highest rates were recorded during the "Nortes" season (November, 2004 and February, 2005), when the oxygen concentrations in the bottom water (7 ± 1.5 mg L^{-1}) were the highest. This can be partly due to the development of diazotrophic microorganisms of conformational protection mechanisms against nitrogenase inactivation by high O_2 concentrations, as reported by Soto-Urzúa & Baca (2001). A second reason should be linked to wind re-suspension of sediment particles driven by Nortes winds at this season. Indeed, re-suspended particles may constitute a microhabitat (within a few microns deep into the particle's interior) with aerobic and anaerobic spaces favoring the bacterial processes of N_2 fixation and denitrification (Ferrara-Guerrero *et al.*, 2007). In the sediments the deposit of labile organic matter is also related to these hydrodynamic processes and bio-perturbation from benthic organisms, which will induce environmental changes in each sampling station (Brune *et al.*, 2000).

The oxygen sediment concentrations in the two seasons studied were high enough to allow recent organic matter mineralization and to provide enough ammonium to the sediment, which would explain the lower rates of N_2 fixation compared to bottom water.

The fact that the highest ammonium concentrations of the bottom water were registered in February, 2005 is due to strong winds and tidal effects that provoked sediment re-suspension thus allowing NH_4^+ diffusion towards the bottom water, as has been already demonstrated for this type of environment by Kemp *et al.* (1990). In such shallow ecosystems, the interchange of dissolved substances through the water-sediment interface is a process that affects the chemical composition of the water column since it regenerates the ammonium and nitrate required for the primary biomass production in the superficial layers (Maksymowska-Brossard & Piekarek-Jankowska, 2001). Generally, the relatively low concentrations of $N\text{-}NH_4^+$ in bottom water favored N_2 fixation in both seasons.

The high salinities registered at the bottom water in June inhibited the N_2 fixation, so it was about 80% lower than in November and February, this coincide with Herbst (1998), Serraj & Drevon (1998) who reported that salinity is a N_2 fixation inhibitor (Figs. 2 and 5). Generally, planktonic nitrogen fixation is greater in freshwater than in marine ecosystems (Nielsen *et al.*, 2003) however, it appears to be no difference in nitrogen fixation rate in the sediments in relation to different salinities from bottom water (Figs. 3 and 5).

The denitrification rates measured in our study match with those reported for eutrophic estuarine ecosystems impacted by waste water inputs (Seitzinger, 1988). The relationship between the high denitrification rates and the high $N\text{-}NO_2^-$ concentrations in reduced sediments was not too clear, as high denitrification rates were also recorded in reduced sediments with low $N\text{-}NO_2^-$ concentration during the rainy season. According to Codispoti *et al.* (2001) nitrites tend to accumulate in the water column only when the oxygen concentrations are $<2.5\ \mu M$, however, this trend is hardly observed in the sediments due to the interment activity of the benthic meiofauna provoking the formation of patches in the superficial sediments (Hendricks, 1993; Sheibley *et al.*, 2003) and thus the formation of slightly oxidized microhabitats with low N-NO_2^- concentrations. This was observed in the “Nortes” season at El Cocal station where high denitrification rates, low NO_2^- concentrations and slightly oxidized sediments were observed.

The denitrification rate was also affected by salinity and sediment organic matter in agreement with Richardson *et al.* (2004) and Rysgaard *et al.* (1999) who reported, in estuarine ecosystems, 50% decrease of denitrification with an increase of salinity from 0 to 10 PSU. In our study we found a 18% decrease of denitrification rate when the salinity changed from 13 to 32 PSU.

The high N_2O concentrations produced during the denitrification depend mainly on electronegative Eh values, nitrate concentration and sediments texture. Nevertheless, the denitrification process was extremely variable temporarily and spatially, because when the N-NO_3^- concentration increases, it is not fully used for denitrification process since a proportion can be reduced to NH_4^+ (Ullah & Zinati, 2006).

On the other hand, given that the mineralization of the organic matter that has been recently deposited in the superficial sediments produces the greatest quantity of N-NH_4^+ and N-NO_3^- used for the denitrification; it can be expected that the sites with highest deposition of labile OM present a high N_2O production.

Conclusions

The exchange of ammonium and nitrate ions between the water-sediment interface and the bottom water is a key factor for nitrogen regeneration that strongly varies between the different seasons of the year (“Nortes” and rains).

The results suggest that increased salinity, from 24 to 29 PSU, may limit nitrogen fixation in bottom water.

The greatest loss of nitrogen as N_2O through denitrification happened in the reduced sediments with greatest silt percentage and high NO_3^- concentrations, located at the deepest zones of the navigation canal.

The strong variations of N_2 fixation and denitrification rates registered in the “Nortes” and rainy seasons can be linked to the runoff of terrestrial nitrogen within the lagoon, originating from the bordering zones that have been deforested for farming purposes.

For future investigations on the global N cycle it will be necessary to extend the measurements of N_2 fixation, denitrification and nitrification rates as well as their connection with the regeneration and loss of N, to other Mexican coastal lagoons that are still poorly studied.

Acknowledgements

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PART 4

Primary producers

Phytoplankton of the Sontecomapan Lagoon, Veracruz, Mexico

María Guadalupe Figueroa Torres, Saúl Almanza Encarnación,
María Jesús Ferrara-Guerrero & Marc Pagano

The effect of mangrove leaf litter extracts on primary productivity and phytoplankton growth

José Antolín Aké-Castillo & Gabriela Vázquez

Phytoplankton of the Sontecomapan Lagoon, Veracruz, Mexico

María Guadalupe Figueroa Torres,* Saúl Almanza Encarnación,*
María Jesús Ferrara-Guerrero* & Marc Pagano**

ABSTRACT: Phytoplankton plays a key role in aquatic ecosystems as an oxygen producer, a CO_2 trap, a primary source of food in trophic chains and as an indicator of changes in the environment. However, despite this positive importance, it can also develop into harmful algal blooms. With the aim of increasing knowledge about this group of microorganisms in Mexican aquatic ecosystems, a list of the phytoplankton species of the Sontecomapan Lagoon was made indicating those that potentially can provoke red tides. Besides, the distribution and abundance of these species was studied in two seasons, the rainy one (June, 2015) and the dry one (February, 2016), on eight sampling stations. Phytoplankton samples were collected with a Van Dorn bottle to measure environmental factors (transparency, salinity, temperature, pH and dissolved oxygen). A list with 357 species with a clear dominance of diatoms (67.8%) and dinoflagellates (20.16%) was obtained from literature review and materials derived from this study. Among them, 19.88% can potentially form red tides, and some of them are toxic. From the samples collected, 102 species of phytoplankton were recorded; 42 of them during the rainy season, 65 during the dry one and 7 presents in both. Among these species, 17 can potentially form red tides and from these, only two can be toxic for

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humans: *Dinophysis caudata* and *Lyngbya majuscula*. The cluster analysis of the environmental factors showed the formation of four groups in the rainy season and three in the dry season, associated to the salinity gradients.

KEYWORDS: Red tides, Harmful Algal Blooms (HAB), tropical coastal lagoon, Mexico.

Introduction

The phytoplankton of the Sontecomapan lagoon has been studied by Suchil (1990), who carried out a seasonal sampling of communities and reported a succession of diatom and dinoflagellates blooms. Guerra-Martínez (1996) studied the variations in nano and microplankton biomass at the lagoon mouth in 1992 and 1993; Meave del Castillo & Lara-Villa (1997) made an inventory of planktonic diatoms and Figueroa-Torres *et al.* (2009) made another one on the thecate dinoflagellates from the lagoon. Camacho *et al.* (1994) studying the circadian rhythms of phytoplankton in October 1993, recorded 90 taxa of which *Skeletonema subsalsum* was the dominant throughout the cycle. Muciño-Márquez *et al.* (2011) studied phytoplankton populations in October, 1999 and reported 179 species, from which *Fragilaria exigua*, *F. tenuicollis*, *F. ulna var ulna*, *Prorocentrum gracile* and *Scrippsiella trochoidea* were the dominant ones. Both studies agreed that diatoms and dinoflagellates were the most abundant and frequent groups in their samples. On the other hand, Muciño-Márquez *et al.* (2012), at the same period, recorded 27 red tide-forming species in this lagoon. Specific taxonomic aspects of some species have been studied by others investigators in this lagoon; Aké-Castillo *et al.* (1995) reported morphological variations of some species from the *Skeletonema* genus (*S. subsalsum, S. pseudocostatum* and *S. costatum)* while Aké-Castillo *et al.* (2000, 2004) reported species from the *Chaetoceros* genus (*Chaetoceros subtilis* var *abnormis* f. *abnormis, Chaetoceros subtilis* var *abnormis* f. *simplex*). Aké-Castillo & Vázquez-Hurtado (2011) described the presence of *Peridinium quinquecorne* var *trispiniferum* and Aké-Castillo (2015) reported the *Thalassiosira cedarkeyensis* species. In summary, according to the different studies, a very great variability in composition and abundance of phytoplankton species has been observed at different sampling stations and seasons at Sontecomapan Lagoon with no clear distribution patterns.

Therefore, this work is a contribution to the phytoplankton studies of the Sontecomapan Lagoon by developing an up-to-date list of species with new sampling observations at rainy and dry seasons. Also, this work is a plea for a constant monitoring of the phytoplankton in the lagoon in order to analyze time-changes in the species' composition and detect the eventual presence of toxic species in the ecosystem to prevent damage to human health.

Materials and Methods

The Sontecomapan Lagoon is a tropical coastal ecosystem characterized by three zones according to the horizontal salinity gradients: fresh, brackish and marine waters; with two-layer flows and vertical mixtures, associated with the climatic conditions of the region (Lankford, 1977). The average depth is 1.5 meter. The combination of physical and chemical factors leads to the formation of zones with different microenvironmental conditions of location and variable dimensions, which favor the differential development of phytoplankton species. Eight sampling stations were set up along the lagoon: La Boya, Río La Palma, El Real, Estero El Fraile, Punta Levisa, El Chancarral, Río Sábalo and Río Basura (Fig. 1). Sampling was performed in June, 2015 (rainy season) and February, 2016 (dry season).

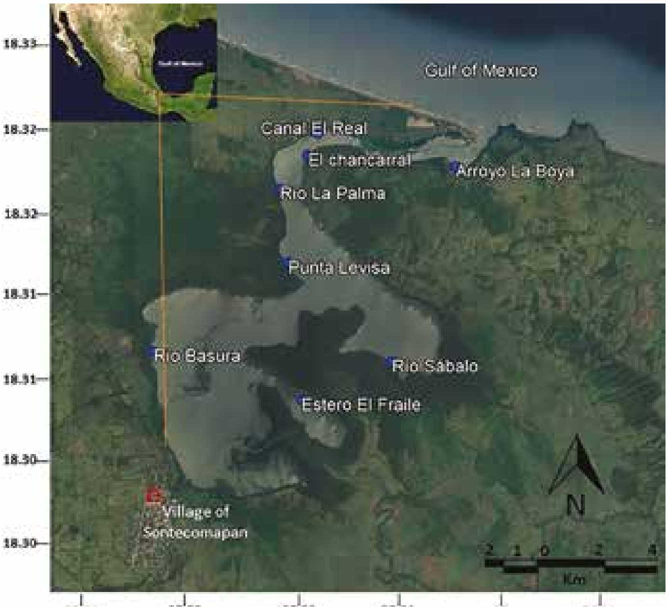

Figure 1. Location of the Sontecomapan Lagoon and sampling stations (Modified from Image @DogotalGlobe, Image@2016TerraMetrics)

Sixteen phytoplankton samples were collected at subsurface level of the water column using a Van Dorn bottle and were placed in 500 ml glass bottles preserved with acetate-lugol at 1%. To perform the counts, samples were homogenized and placed in one-milliliter chambers to be checked under a Zeiss Axiovert 135 inverted microscope, by triplicate. For taxonomic determination, the keys and descriptions by Taylor (1976), Dodge (1982), Balech (1988), Fukuyo *et al.* (1990), Licea *et al.* (1995), Bravo (2004), Okolodkov (2010), Gómez (2010, 2013) and García–Mendoza *et al.* (2016) among others, were consulted. Nomenclature was updated based on the work of Guiry & Guiry (2017).

Salinity was measured with an Atago SMill-E refractometer series 0183181, pH with an Orion pH meter 250A series 017210; depth and transparency with a Secchi disk, water temperature with a cuvette thermometer (0-100 °C ± 1 °C) and the dissolved oxygen concentration was calculated according to Winkler's technique (Strickland & Parsons, 1972).

A cluster analysis was performed using the Statistica 8 software, for the environmental conditions of the lagoon.

Results

Physical and chemical factors

During the rainy season, water depth was greater in stations Canal el Real, Río Sábalo, El Chancarral and La Boya with values of 230, 200, 105 and 100 cm respectively, compared to the dry season, which was shallower, with depths of 73, 180, 44 and 15 cm. Spatially, the deepest stations were recorded far from the mouth at Río Basura and Río Sábalo with values of 320 and 200 respectively in rainfall and 340 and 180 cm in dry. It is clear that the rains and the continental water contributions have a great influence on the depth increase throughout the system (Fig. 2A).

Transparency was generally high in the two sampling periods, with the exception of Río Basura where transparency was 85 cm in the rainy season and 90 cm in the dry season (Fig. 2B).

The water temperature ranged between 25 and 33.5 °C and was higher in the rainy season, where the highest value was obtained in El Fraile (33.5 °C) and the minimum in Río La Palma (25.9 °C). In the dry season, it varied very little: from 25 to 27°C at the different sampling sites (Fig. 2C).

As for pH, it remained close to neutrality in both seasons (Fig. 2D); in the rainy season the maximum value was 7.9 at El Real and the minimum at Río Sábalo with a 6.0 value. In the dry season, Río Sábalo showed the maximum value (7.65) and Río La Palma the minimum (6.7). Río Sábalo, El Fraile and La Boya presented the highest pH values in the dry season and the lowest in the rainy season.

Concentrations of dissolved oxygen showed a spatial variation in the rainy season with highest values at El Chancarral (11.6 mg/L) and Río La Palma (8.8 mg/L) and lowest values at La Boya and Río Sábalo (3.3 mg/L both). In the dry season, values were more homogeneous and varied between 7.8 mg/L at El Chancarral and 5.3 mg/L at Punta Levisa (Fig. 2E).

Salinity presented great fluctuations, from 0 to 20 psu (Fig. 2F). During the rainy season, maxima of 18 and 19 psu were observed at La Boya and El Real, near the mouth of lagoon, and a minimum of 0 psu in Río La Palma and Río Basura, which are further away from the lagoon's mouth. It is worth noting that in this season (although not shown in the figures) salinity of the bottom water at El Chancarral and El Real showed values of 35 and 34 psu respectively, forming a wedge-like halocline. In the dry season the highest salinity (20 psu) was recorded at Río Sábalo, although this station is distant from the mouth of the lagoon, whereas the lowest (3 psu) was recorded at Río Basura.

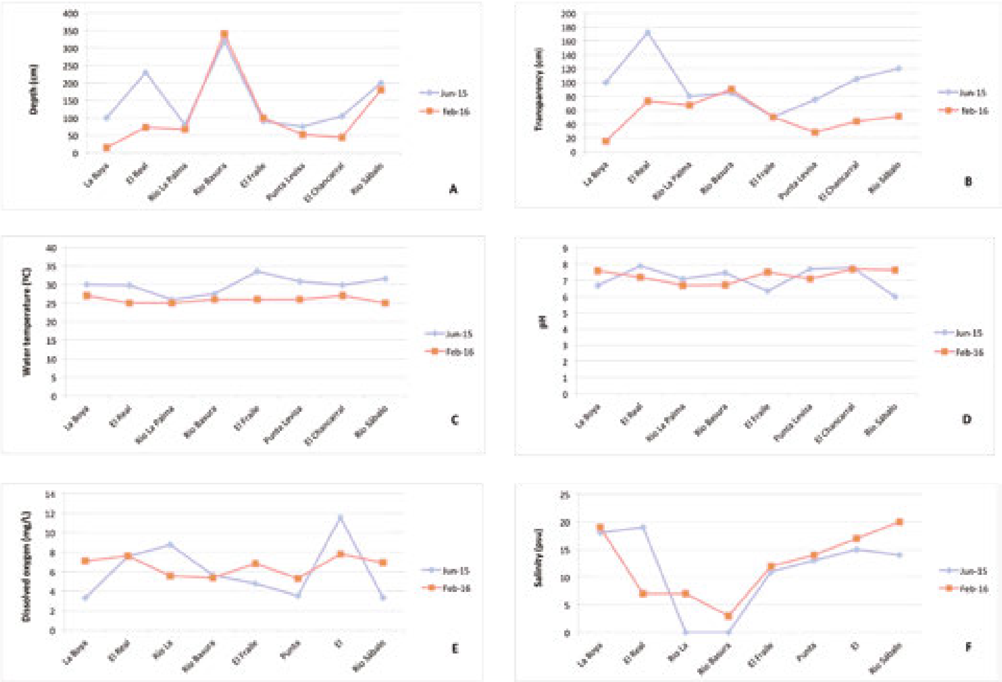

Figure 2. Environmental variables registered in the Sontecomapan Lagoon, Veracruz in rainy season (June, 2015) and dry season (February, 2016). A) depth, B) transparency, C) temperature, D) pH, E) dissolved oxygen and F) salinity.

The cluster analyses based on physical and chemical data (Fig. 3A-B) established clusters (corresponding to geographic zones) at a link distance of 50. The rainy season was more heterogeneous with four clusters, while the dry season formed three. In the dendrogram corresponding to the rainy season (Fig. 3A), the Río Sábalo and the Canal El Real sampling stations were separated in spite of having the highest values of depth and transparency, salinity, pH and temperature. Probably the morphology of the basins was the main factor since Río Sábalo is more isolated from the zone under tidal influence and therefore, with more stagnant waters, while El Real is under oceanic coastal waters influence and high hydrodynamic conditions. Río Basura that has the lowest salinity formed the third zone and all other sampling stations the fourth zone. In the dry season, Río Basura also formed a single cluster characterized by the lowest salinity, as well as Río Sábalo characterized by the highest. The third cluster grouped the rest of the sampling stations with intermediate salinities. Apparently, the other factors were less important than salinity in this season. However, we can point that at both seasons Río Sábalo, which correspond to the more confined zone, clearly distinguished from all the other stations.

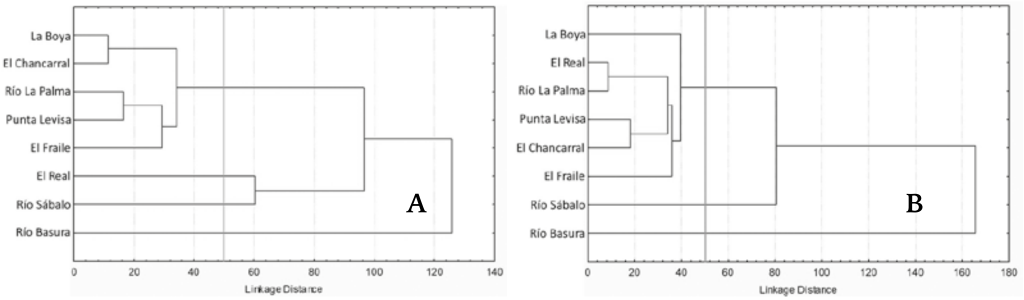

Figure 3 A-B. Dendrograms of Sontecomapan Lagoon, Veracruz, based on physical and chemical data: A) rainy season (June, 2016) and B) dry season (February, 2016)

Phytoplankton

From literature revision and this study, a list of 357 species was generated for the lagoon (Table 1), from which 241 (67.8%) were Diatoms, 72 (20.16%) Dinoflagellates, 13 (3.64 %) Chlorophytes, 11(3.08%) Euglenophytes, 16 (4.20%) Cyanoprokariotes, 2 (0.56%) Raphydophyceae and 2 (0.56%) Silicoflagellates.

We observed that 46 (12.88%) from the species registered in the general list, are potentially red tide forming (Table 2), and are mainly represented by Dinoflagellates with 20 species (43.48%) and Diatoms with 19 (41.30%).

Among them, 12 species are toxic and five affect humans: *Pseudo-nitzschia pungens*, *Pseudo-nitzschia pungens var atlántica* and *Pseudo-nitzschia seriata* diatoms, which produces domoic acid that causes amnesic poisoning by consumption of contaminated shellfish, *Dinophysis caudata* dinoflagellate that causes diarrhea and Cyanoprocariota *Lyngbya majuscula* that produces dermatitis and respiratory diseases.

Five other species affect the marine fauna: dinoflagellates *Gonyaulax spinifera*, producer of yessotoxin that can kill fish and invertebrates, *Phalacroma rotundatum* that may cause problems to salmon in captivity, *Prorocentrum gracile* that causes fish mortality, *Prorocentrum micans* that produces verupine (hepatoxin) toxin affecting clams and shellfish, and *Tripos furca* which may cause great mortality of tuna in captivity. Likewise, the Cyanoprokariota *Anabaenopsis circularis* has been reported as toxic, but not enough information is available, as well as the Raphydophycea *Olisthodiscus luteus*, which seems to cause red blood cells lysis.

From the water samples analyzed in this study, 102 phytoplankton species (Table 1, Plates 1 to 3) were recorded, of which 61 are reported for the first time in the lagoon. Among these 102 species, 69 (67.64%) belong to the group of diatoms, 17 (16.67%) to Dinoflagellates, 10 (9.80%) to Cyanoprokaryotes, 4 (3.93%) to Euglenophytes and 2 (1.96%) to Chlorophytes. It was detected that 17 of these species are potentially red tide forming, eight of them being Diatoms, six Dinoflagellates and three Cyanoprokariotes. Only two are toxic for humans: *Dinophysis caudata* dinoflagellate, that causes diarrhea from contaminated shellfishes consumption, and cyanoprocariota *Lyngbya majuscula*, which may cause dermatitis and respiratory irritation.

Fourty two (42) species were collected in the rainy season and 65 in the dry season (Fig. 4A); 7 species were common to both seasons.

In the dry season, La Boya had the highest richness with 31 species, followed by Río Basura with 29 and El Fraile with 27. In the rainy season, the greatest species richness was registered at Punta Levisa with 18 and Río Sábalo with 14. It is noteworthy that at this season, only two species were registered at El Chancarral.

Phytoplankton abundance reached values up to 300 x 10^4 cells/l (Fig. 4B), but mats of the cyanoprokaryote *Merismopedia convoluta* were also observed in the dry season at El Fraile sampling station with 1600 x 10^4 cells/l; this value was discarded in the graph (Fig. 4B) because it gives the impression of a massive bloom. However, this species is very small (3-5 µm in diameter), hardly perceptible in site and sample (Plate 3, Fig. 24), considering that most of the phytoplankton organisms measure more than 50 μm, which is 15 times greater than *M. convoluta*, and with biovolumes 3,000-fold higher.

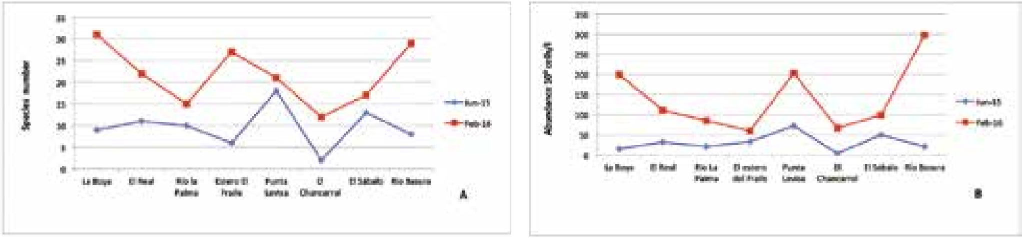

Figure 4 A-B. Phytoplankton species richness and abundance in Sontecomapan Lagoon, Veracruz, in the rainy (June, 2015) and dry (February, 2016) seasons

Table 1. Phytoplankton of the Sontecomapan Lagoon, Veracruz, Mexico

Taxa	Previous records in the lagoon	This study		Taxa	Previous records in the lagoon	This study	
		Jun 2015	Feb 2016			Jun 2015	Feb 2016
Diatoms				*Biddulphia tuomeyi* (J. W. Bailey) Roper	*13*		
Actinocyclus curvatulus Janisch	*8*			*Biddulphia* sp.	*14*		
Actinocyclus sp.	*13*			*Biddulphia* sp. *1*	*13*		
Actinoptychus splendens (Shadbolt) Ralfs	*8*			*Biddulphia* sp. *2*	*13*		
Amphiprora angustata Hendey	*14*			*Brachysira procera* Lange-Bertalot & Gerd Moser	*none*		*X*
Amphora pediculus (Kützing) Van Heurck	*14*			*Caloneis permagna* (Bailey) Cleve	*8*		
Amphora proteus Gregory	*15*			*Campylodiscus echeneis* Ehrenberg ex Kützing	*13*		
Amphora sp.	*14*			*Campylodiscus* sp.	*13*		
Amphicocconeis disculoides (Hustedt) Stefano & Marino	*14*			*Cerataulina pelágica* (Cleve) Hendey	*13*		
Anemastus tuscula (Ehrenberg) D. G. Mann & A. J. Stickle	*14*			*Cerataulus smithii* Ralfs	*13, 17,*		*X*
Asterionellopsis glacialis (Castracane) Round	*8,13*			*Chaetoceros affinis* Lauder	*13*		
Aulacoseira granulata (Ehrenberg) Simonsen	*none*	*X*	*X*	*Chaetoceros atlanticus* Cleve	*10, 14,16*		
Aulodiscus sp.	*14*			*Chaetoceros compressus* Lauder	*13*		
Bacillaria paxillifera (O. F. Müller) T. Marsson	*5,14,16,*		*X*	*Chaetoceros curvisetus* Cleve	*13*		
Bacteriastrum delicatulum Cleve	*14*			*Chaetoceros cf debilis* Cleve	*14, 16*		
Bacteriastrum hyalinum Lauder	*13, 14*			*Chaetoceros denticulatus* H. S. Lauder	*13*		
Bacteriastrum sp.	*14*			*Chaetoceros holsaticus* F. Schütt	*5, 11*		
Bellerochea sp.	*13*			*Chaetoceros laevis* Leuduger-Fortmorel	*13*		
Biddulphia biddulphiana (J. E. Smith) Boyer	*14*			*Chaetoceros lorenzianus* Grunow	*8,13*		
Biddulphia biddulphiana (J. E. Smith) Boyer	*14*			*Chaetoceros aff muelleri* Lemmermann	*13*		
Chaetoceros peruvianus Brightwell	*13*			*Coscinodiscus pavillardii* Forti	*13*		
Chaetoceros aff subtilis Cleve	*13*			*Coscinodiscus aff radiatus* Ehrenberg	*14,*		*X*
Chaetoceros subtilis var abnormis f abnormis Proschkina-Lavrenko	2			*Coscinodiscus* sp.	*13*		*X*
Chaetoceros subtilis var abnormis f simplex Proschkina-Lavrenko	2			*Coscinodiscus* sp. *1*	*13*		

Table 1. (Continuation)

Taxa	Previous records in the lagoon	This study		Taxa	Previous records in the lagoon	This study	
		Jun 2015	Feb 2016			Jun 2015	Feb 2016
Chaetoceros throndsenii var trondsenia (Marino, Montresor & Zingone) Marino, Montresor & Zingone	2			*Coscinodiscus* sp. 2	13		
Chaetoceros throndsenii var trisetosa Zigone	2			*Coscinodiscus* sp. 3			X
Chaetoceros sp. 1	none	X		*Coscinodiscus* sp. 4			X
Chaetoceros sp. 2	none	X		*Craticula cuspidata* (Kützing) D. G. Mann	8, 13		
Chaetoceros sp. 3	14			*Cyclotella meneghiniana* Kützing	13, 14,16	X	
Cocconeis placentula Ehrenberg	14,	X		*Cyclotella* sp.	none		X
Cocconeis scutellum var scutellum Ehrenberg	8, 14			*Cyclotella* sp. 1	13		
Cocconeis sp.	14			*Cyclotella* sp. 2	13		
Corethron hystrix Hensen	8,13			*Cylindrotheca closterium* (Ehrenberg) Reimann & J. C. Lewin	8, 13, 14,	X	
Coscinodiscus af. asteromphalus Ehrenberg	none	X		*Cymatopleura solea* (Brébisson) W. Smith	8, 16		
Coscinodiscus centralis Ehrenberg	8, 14, 16		X	*Cymbella minuta* Hilse	none		X
Coscinodiscus concinnus W. Smith	13, 14, 16			*Cymbella prostata* (Berkeley) Cleve	none		X
Coscinodiscus curvatulus	14			*Cymbella* sp.	none		X
Coscinodiscus granii L. F. Gough	13, 14,16			*Denticula elegans* Kützing	14		
Coscinodiscus jonesiana (Greville) E. A. Sar & I. Sunesen	13, 17,		X	*Detonula pumila* (Castaracane) Schütt	13		
Diatoma sp.	14			*Fragilaria goulardii* (Brébisson ex Cleve) Lange-Bertalot	13, 14		
Diploneis bombus (Ehrenberg) Ehrenberg	14			*Fragilaria tenuicollis* Heib.	13		
Diploneis ovalis (Hilse) Cleve	14			*Fragilaria ulna var ungeriana* (Grunow) Lange-Bertalot	13		
Diploneis puella (Schumann) Cleve	none		X	*Fragilaria* sp. 1	none		X
Diploneis smithii (Brébisson) Cleve	none	X		*Fragilaria* sp. 2	13, 14		
Diploneis sp.	14			*Fragilaria* sp. 3	13,14		
Ditylum brightwellii (T. West) Grunow	14			*Fragilaria* sp. 4	14		
Encyonema neomesianum Krammer	14			*Fragilaria* sp. 5	14		

Table 1. (Continuation)

Taxa	Previous records in the lagoon	This study		Taxa	Previous records in the lagoon	This study	
		Jun 2015	Feb 2016			Jun 2015	Feb 2016
Entomoneis alata (Ehrenberg) Ehrenberg	13		X	*Fragilariforma exigua* (Grunow) M. G. Kelly	14		
Entomoneis ornata (Bailey) Reimer	none	X		*Gomphoneis clevei* (Frike) Gil	8		
Entomoneis paludosa (W. Smith) Reimer	14			*Gomphoneis herculeana var robusta* (Grunow) Cleve	14		
Eunotia sp.	none		X	*Gomphonema parvulum* (Kützing) Kützing	none	X	X
Eupodiscus radiatus Bailey	13, 17			*Guinardia flaccida* (Castracane) H. Peragallo	none		X
Fallacia pygmaea (Kützing) Stickle & D. G. Mann	14			*Guinardia striata* (Stolterfoth) Hasle	13		
Fragilaria acus (K¨tzing) Lange-Bertalot	14			*Gyrosigma acuminatum* (Kützing) Rabenhorst	none	X	
Fragilaria capucina Desmasières	14			*Gyrosigma attenuatum* (Kützing) Rabenhorst	14	X	X
Tabularia af fasciculata (C. Agardh) D. M. Williams & Round	13			*Gyrosigma balticum* (Ehrenberg) Rabenhorst	13	X	X
Gyrosigma distortum (W. Smith) Griffith & Henfrey	13			*Mastogloia pusilla* Grunow	14		
Gyrosigma eximium (Thwaites) Boyer	none	X		*Mastogloia* sp.	none		X
Gyrosigma fasciola (Ehrenberg) J. W. Griffith & Henfrey	14		X	*Melosira moniliformis* (O. F. Müller) C. Agardh	none	X	X
Gyrosigma af scalproides (Rabenhorst) Cleve	none	X		*Melosira nummuloides* C. Agardh	13	X	X
Gyrosigna sp.	13			*Navicula capitatoradiata* H. Germain	14	X	
Gyrosigma sp. *1*	13			*Navicula cincta* (Ehrenberg) Ralfs	8		
Halamphora clara (A. Schmidt) Levkov	14			*Navicula cryptocephala* Kützing	14		
Hantzchia amphioxys (Ehrenberg) Grunow	14			*Navicula veneta* Kützing	14		
Hantzschia sp.	13			*Navicula distans* (Wm Smith) Ralfs	8		
Helicotheca tamesis (Shrubsole) M. Ricard	13			*Placoneis grastrum* (Ehrenberg) Kützing	14		
Hemiaulus sinensis Greville	13			*Navicula gottlandica* Grunow	14		
Hyalodiscus scoticus (Kützing) Grunow	8			*Navicula menisculus* Schumann	8		
Hyalodiscus sp.	13			*Navicula*	14		
Hyalodiscus sp. *1*	13			*Navicula pennata* A. Schmidt	8		X
Lauderia annulata Cleve	13			*Navicula platyventris* Meister	14		X
Lithodesmium undulatum Ehrenberg	13, 14			*Navicula rhynchocephala* Kützing	14		X

Table 1. (Continuation)

Taxa	Previous records in the lagoon	This study		Taxa	Previous records in the lagoon	This study	
		Jun 2015	Feb 2016			Jun 2015	Feb 2016
Luticola mutica (Kützing) D. G. Mann	*14*			*Navicula tenella* Brébisson ex Kützing	*14*		
Lyrella lyra (Ehrenberg) Karajeva	*none*		*X*	*Navicula veneta* Kützing	*none*	*X*	
Navicula sp. *1*	*none*		*X*	*Odontella af longicruris* (Greville) M. A. Hoban	*none*		*X*
Navicula sp. *2*	*8*	*X*		*Odontella mobiliensis* (J. W. Bailey) Grunow	*8, 13, 14*		*X*
Navicula sp. *3*	*14*			*Odontella* sp.	*13*		
Navicula sp. *4*	*14*			*Palmerina hardmaniana* G. R. Hasle	*13*		
Navicula sp. *5*	*14*			*Paralia sulcata* (Ehrenberg) Cleve	*14*		
Neocalyptrella robusta (G. Norman ex Ralfs) Hernández-Becerril & Meave del Castillo	*13*			*Paraplaconeis placentula* (Ehrenberg) M. S. Kulikovskly & Lange-Bertalot	*14*		
Neostreptotheca subindica Van Stosch	*13*			*Petrodictyon gemma* (Ehrenberg) D. G. Mann	*13*		
Nitzschia angularis W. Smith	*14*			*Petroneis humerosa* (Brébisson ex W. Smith) Stickle & D. G. Mann	*14*	*X*	
Nitzschia longissima (Brebisson) Ralfs	*8, 13,14*			*Pinnularia borealis var scalaris* (Ehrenberg) Rabenhorst	*8*		
Nitzschia macilenta W. Gregory	*none*		*X*	*Pinnularia lata* (Brébisson) W. Smith	*14*		
Nitzschia marginata Hustedt	*8*			*Pinnularia mesolepta* (Ehrenberg) W. Smith	*14*		
Nitzschia obtusa W. Smith	*8*			*Pinnularia nobilis* (Ehrenberg) Ehrenberg	*14*		
Nitzschia sigma (Kützing) W. Smith	*13*		*X*	*Pinnularia* sp.	*8*		*X*
Nitzschia sigmoidea (Nitzsch) W. Smith	*none*		*X*	*Pinnularia* sp. *1*	*14*		
Nitzschia vidovichii (Grunow) Grunow	*14*			*Pinnularia* sp. *2*	*14*		*X*
Nitzschia sp.	*14*			*Placoneis disparilis* (Hustedt) Metzeltin & Lange-Bertalot	*14*		
Nitzchia sp. *1*	*13*			*Plagiotropis lepidoptera* (W. Gregory) Kuntze	*none*	*X*	
Nupela poconoensis (Patrick) Potapova	*14*			*Plagiotropis vitrea* (W. Smith) Grunow	*none*		*X*
Plagiotropis sp. *1*	*none*		*X*	*Rhizosolenia acuminata* (H. Peragallo) H. Peragallo	*14*		
Plagiotropis sp. 2	*none*		*X*	*Rhizosolenia bergonii* H. Peragallo	*14*		
Planothidium lanceolatum (Brébisson ex Kützing) Lange- Bertalot	*14*			*Rhizosolenia castracanei* H Peragallo	*8*		

Table 1. (Continuation)

Taxa	Previous records in the lagoon	This study		Taxa	Previous records in the lagoon	This study	
		Jun 2015	Feb 2016			Jun 2015	Feb 2016
Pleurosigma fasciolata var closterioides (Grunow) Peragallo	*13*			*Rhizosolenia imbricata* Brightwell	*13*		
Pleurosigma rigidum var incurvatum Brun	*14*			*Rhizosolenia setigera* Brightwell	*13*		
Pleurosigma sp.	*13*			*Rhizosolenia* sp.	*13*		
Pleorosigma sp. *1*	*13*			*Rhoicosphenia curvata* (Kützing) Grunow	*none*	*X*	
Pleurosira laevis (Ehrenberg) Compère	*none*		*X*	*Rhopalodia* sp.	*14*		
Pleurosira sp.	*13*			*Sellaphora americana* (Ehrenberg) D. G. Mann	*14*		
Proboscia alata (Brightwell) Sundström	*13,14,16*			*Skeletonema costatum* (Greville) Cleve	*3, 8, 13*		*X*
Proboscia sp.	*14*			*Skeletonema pseudocostatum* Medlin	*3*		
Psammodictyon panduriforme (W. Gregory) D. G. Mann	*14*			*Skeletonema subsalsum* (Cleve-Euler) Bethge	*3, 4, 7, 13*		
Pseudo-nitzchia delicatissima (Cleve) Heiden	*8*			*Skeletonema tropicum* Cleve	*3*		
Pseudo-nitzschia pungens (Grunow ex Cleve) Hasle	*14,16*			*Staurosira construens* Ehrenberg	*14*		
Pseudo-nitzschia pungens var atlantica (Cleve) Moreno & Licea	*8, 10, 14*			*Staurosirella pinnata* (Ehrenberg) D. M. Williams & Round			*X*
Pseudo-nitzschia seriata (Cleve) H. Peragallo	*10, 14, 16*			*Stenoneis insconpicua* (W. Gregory) Cleve	*14*		
Pseudosolenia calcar-avis (Schultze) B. G. Sundström	*13*			*Stephanopyxis palmeriana* (Greville) Grunow	*13*		
Pseudostaurosira parasítica (W. Smith) E. Morales	*14*			*Surirella af guatemalensis* Ehrenberg	*none*		*X*
Surirella fastuosa var cuneata O. Witt	*13*		*X*	*Thalassiosira* sp.	*13,14*		
Surirella febigerii F. W. Lewis	*13*		*X*	*Trieres regia* (M. Schultze) M. P. Ashworth & E. C. Theriot	*13*		
Surirella gemma (Ehrenberg) Kützing	*none*		*X*	*Tryblionella compressa* (J. W. Bailey) Poulin	*14*		
Surirella minuta Brébisson ex Kützing	*8*			*Ulnaria ulna* = (Nitzch) CompèreSynedra ulna	*14*		
Surirella splendida (Ehrenberg) Kützing	*none*		*X*	*Zigoceros ehrenbergii* E. A. SAr	*13*		
Surirella striatula Turpin	*8,14*			*Dinoflagellates*			
Synedra superva Kützing	*none*	*X*		*Achradina* sp.	*14*		

Table 1. (Continuation)

Taxa	Previous records in the lagoon	This study		Taxa	Previous records in the lagoon	This study	
		Jun 2015	Feb 2016			Jun 2015	Feb 2016
Synedra tabulata (C. Agardh) Kützing	*none*		*X*	*Akashiwo sanguínea* (K. Hirasaka) G. Hansen & Moestrup	*14, 16*		*X*
Synedra ulna var ulna (Nitzsch) Ehrenberg	*8*			*Ceratium horridum var buceros* (Zacharias) Sournia	*9*		
Synedra sp.	*13*			*Ceratium massiliense f armatum* (Karsten) Schiller	*9*		
Tabularia fasciculata (C. Agardh) William & Round	*none*	*X*		*Ceratium tripos var atlanticum* (Ostenfeld) Paulsen	*9*		
Terpsinoë musica Ehrenberg	*13*		*X*	*Ceratium* sp.	*14*		
Thalassionema frauenfeldii (Grunow)Tempère & Peragallo	*13*			*Ceratium* sp. *1*	*14*		
Thallasionema nitzschiodes (Grunow) Mereschkowsky	*13*	*X*		*Cuneolus aff skvortzowii* (Nikolae) Medlin	*13*		
Thalassionema nitzschioides var lanceolata Grunow	*14*			*Dinophysis caudata* Saville-Kent	*5; 9, 10, 14, 16*		*X*
Thalassiosira cedarkeyensis A.K.S.K. Prasad	*1*			*Dinophysis* sp.	*14*		
Thalassiosira plicata Schrader inc.	*14*			*Diplopsalis* sp.	*13,14*		
Glenodinium sp.	*14*			*Oxytoxum sceptrum* (Stein) Schröder	*9*		
Goniodoma sphaericum Murray & Whitting	*14*		*X*	*Peridiniella danica* (Paulsen) Y. B. Okolodkov & J. D. Dodge	*14*		
Gonyaulax digitalis (Pouchet) Kofoid	*9*			*Peridiniopsis polonica* (Woloszynska) Bourrelly	*14*		
Gonyaulax polygramma Stein	*9*		*X*	*Peridinium quadridentatum* (F. Stein) Gert Hansen	*5, 6*		
Gonyaulax spinifera (Claparède et lachmann) Diesing	*9*			*Petroneis humerosa* (Brébisson ex Smith) Stickle & D. G. Mann	*13*		
Gonyaulax turbynei Murray & Whitting	*14, 16*			*Phalacroma rotundatum* (Claparède et Lachmann) Kofoid & Michener	*5, 916*		
Gymnodinium sp.	*14*			*Podolampas bipes Stein*	*9*		
Gymnodinium sp. *1*	*14*			*Podolampas bipes var reticulata* (Kofoid) Taylor	*14*		
Gyrodinium fusiforme Kofoid & Swezy	*14*			*Podolampas palmipes* Stein	*9*		
Gyrodinium spirale (Bergh) Kofoid & Swezy	*14, 16*			*Prorocentrum cordatum* (Ostenfeld) Dodge	*1*		
Noctiluca scintillans (Macartney) Kofoid & Sweay	*5, 14, 16*			*Prorocentrum gracile* Schütt	*5; 9, 14,15, 16*		
Operculodinium israelianum (M. Rossignol) Wall	*14*			*Prorocentrum mexicanum* Osorio-Tafall	*14, 15*		

Table 1. (Continuation)

Taxa	Previous records in the lagoon	This study		Taxa	Previous records in the lagoon	This study	
		Jun 2015	Feb 2016			Jun 2015	Feb 2016
Ornithocercus magnificus Stein	9			*Prorocentrum micans* Ehrenberg	5, 9, 14, 15, 16		
Ostreopsis sp.	14			*Prorocentrum scutellum* Schröder	14, 15		
Oxyrrhis marina Dujardin	none	X		*Prorocentrum* sp.	14		
Oxytoxum globosum Schiller	14			*Protoperidinium argentinense* Balech	none	X	
Oxytoxum mediterraneum Schiller	14			*Protoperidinium brevipes* (Paulsen) Balech	none	X	
Protoperidinium brochii (Kofoid &Swezy) Balech	9			*Protoperidinium thulesence* (Balech) Balech	none	X	
Protoperidinium claudicans (Paulsen) Balech	9			*Pyrocystis lunula* (Schütt) Schütt	14		
Protoperidinium corniculum Kofoid et Michener	9			*Pyrophacus* horologium Stein	9, 14		
Protoperidinium crassipes (Kofoid) Balech	9	X		*Pyrophacus steinii* (Schiller) Wall & Dale	9		
Protoperidinium curtipes (Jørgensen) Balech	none		X	*Scrippsiella acuminata* (Ehrenberg) Kretschmann, Elbrächter, Zinssmeister, Soehner, Kirsch, Kusber & Gottschling	14, 16,		X
Protoperidinium depressum (Bailey) Balech	9, 14, 16		X	*Tripos deflexus* (Kofoid) F. Gómez	9		
Protoperidinium divergens (Ehrenberg) Balech	9			*Tripos furca* (Ehrenberg) F. Gómez	9, 10, 11, 14		
Protoperidinium oceanicum (Vanhöffem) Balech	9			*Tripos fusus* (Ehrenberg) F. Gómez	9, 10, 14, 16		
Protoperidinium ovatum Pouchet	14			*Tripos hircus* (Schröder) F Gómez	1, 9, 11, 12		X
Protoperidinium paulseni Abe	none		X	*Tripos lunula* (Schimper ex Karsten) F. Gómez	9		
Protoperidinium pentagonum (Gran) Balech	9			*Tripos teres* (Kofoid) Gómez	9		
Protoperidinium puntulatum (Paulsen) Balech	14	X		*Tripos tripos* (O. F. Müller) F Gómez	14		X
Protoperidinium thorianum (Paulsen) Balech	none	X		*Tryblionella compressa* (J. W. Bailey)	14, 15,16		
Zygabikodinium lenticulatum Loeblich Jr et Koeblich III	9, 14			*Euglena cf stellata* Mainx	14,16		
CHLOROPHYTES				*Euglena cf viridis* (O. F. Müller) Ehrenberg	14, 16	X	
Actinastrum hantzschii Lagerheim	14			*Euglena* sp.	14		
Ankistrodesmus gracilis (Reinsch) Korshikov	14			*Euglena* sp. 1	14		
Closterium moniliferum Ehrenberg ex Ralfs	none		X	*Euglena* sp. 2	14		
Coelastrum microporum Nägeli	none	X		*Pleuromonas* sp.	14		

Table 1. (Continuation)

Taxa	Previous records in the lagoon	This study		Taxa	Previous records in the lagoon	This study	
		Jun 2015	Feb 2016			Jun 2015	Feb 2016
Pandorina morum (O. F. Müller) Bory	*14*			*Trachelomonas hispida* (Perty) F. Stein	*none*	*X*	
Raphidonema nivale Lagerheim	*14*			*Trachelomonas lefebrei* Deflandre	*none*	*X*	
Scenedesmus arcuatus (Lemmermann) Lemmermann	*14*			*Trachelomonas volvocina* (Ehrenberg) Ehrenberg	*none*	*X*	
Scenedesmus acuminatus (Lagerheim) Chodat	*14*			*Trachelomonas* sp.	*14*		
	14			*Cyanoprokaryotes*			
Scenedesmus maximus (West & G. S. West) Chodat	*14*			*Anabaenopsis circularis* (G. S. West) Woloszynska & V. Miller	*none*	*X*	
Staurastrum leptocladum Nordstedt	*14*			*Arthrospira platensis* Gomont	*none*	*X*	*X*
Tetradesmus lagerheimii M. J. Wynne & Guiry	*14*			*Arthrospira subsalsa* (Oersted ex Gomont) W.B. Crow	*none*	*X*	
Ulotrhrix sp.	*14*			*Johannesbaptistia pellucida* (Dickie) W. R. Taylor & Drouet	*14*		
EUGLENOPHYTES				*Limnoraphis hieronymusii* (Lemmermann) J. Komárek, E. Zapomelová, J. Smarda, J. Kopecky, E. Rejmánková, J. Woodhouse, B. A. Neilan & J. Kommárková	*none*		*X*
Euglena polymorpha P. A. Dangeard	*14*			*Lyngbya majuscula* Harvey ex Gomont	*none*		*X*
Merismopedia convoluta Brébisson ex Kützing	*none*		*X*	*Phormidium retzii* Kützing ex Gomont	*14*		
Merismopedia elegans A. Braun ex Kützing	*14*			*Spirulina nordstedtii* Gomont	*14*		
Merismopedia glauca (Ehrenberg) Kützing	*14*			*Silicoflagellate*			
Mersimopedia sp.	*14*			*Dictyocha fibula* Ehrenberg	*14, 16*		
Oscillatoria aff limosa C. Agardh ex Gomont	*none*		*X*	*Dictyocha* sp.	*14*		
Oscillatoria obtusa N. L. Gardner	*none*		*X*	*Raphidophyceae*			
Oscillatoria sp.	*none*	*X*		*Chatonella* sp.	*14*		
Phormidium nigroviride (Thwaites ex Gomont) Anagnostidis & Kpmárek	*none*	*X*		*Olisthodiscus luteus* N. Cartes	*14, 16*		

(1) Aké-Castillo 2015, (2) Aké-Castillo *et al*. 2004, (3) Aké-Castillo *et al*. 1995, (4) Aké-Castillo *et al*. 1997, (5) Aké-Castillo & Vázquez-Hurtado 2008, (6) Aké-Castillo & Vázquez-Hurtado 2011, (7) Camacho *et al*. 1994, (8) Carbajal 2009, (9) Figueroa-Torres *et al*. 2009, (10) Guerra-Martínez 1996, (11) Guerra-Martínez & Lara-Villa (1996), (12) Guerra-Martínez & Meave del Castillo (1997), (13) Meave del Castillo & Lara-Villa 1997, (14) Muciño-Márquez *et al*. 2011 a, (15) Muciño-Márquez *et al*. 2011 b, (16) Muciño-Márquez *et al*. 2012, (17) Meave del Castillo & Moreno-Ruiz (1997).

In the rainy season, due to their abundance, only two species dominated: the *Melosira nummuloides* diatom with 21 x10^4 cells/l at Punta Levisa and the euglenophyte *Trachelomonas volvocina* with 23 x10^4 cells/l, at Río Sábalo. In contrast, in the dry season, high abundances of diatoms were observed: *Eunotia* sp. with 87 x10^4 cells/l at Río Basura and 4 x10^4 cells/l at Río La Palma, *Melosira moniliformis* was frequent in all sampling stations and reaching up to 28 x10^4 cells/l, *Gomphonema parvulum* with a high abundance of 26 x 10^4 cells/l present only at Río Basura. At this latter station, *Nitzschia macilenta* was observed with 21 x 10^4 cells/l, although it was also present in seven of the eight sampling stations. Among the dinoflagellates, only *Tripos hircus* can be highlighted, as it was present in seven of the eight sampling stations, reaching its maximum density of 20 x10^4 cells/l at Río Sábalo. The cyanoprokaryote *Oscillatoria* af. *limosa* showed values of 10.2 x10^5 cells/l at La Boya, 61 x10^4 cells/l at El Real, 81 x10^4 cells/l at Punta Levisa and 27 x10^4 cells/l at El Chancarral sampling stations. *Lyngbya majuscula* reached 35 x10^4 cells/l at Estero El Fraile, *Limnoraphis hieronymusii* 29 x10^4 cells/l only at Río Sábalo, and, as already mentioned, *Merismopedia convoluta* reached a very high density of 1600 x10^4 cells/l only at El Fraile.

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Discussion

It was observed that depth varied little in the sampling stations between the two periods, except at El Real which was far deeper in the rainy season (near 230 cm) than in the dry one (73 cm), probably due to erosion caused by currents. The rest of the sampling stations were shallow, reaching only a few centimeters deep, comparable to the mean depth of Mexican lagoons (150 cm; Lankford, 1977).

The temperatures recorded in our study (25 to 33.5 °C) were higher than those reported by Lankford (1977) and García-Cubas & Reguero (1995), with values close to 24 °C. Likewise, we recorded the highest temperatures in the rainy season, whereas Morán (1994) reported highest values in dry season.

The highest salinities found in dry season (20 psu) were similar to those previously reported by Morán (1994). At both seasons, the lowest values (0-7 psu) were observed closed to river outlet (La Palma and Basura rivers) and the highest (18-19 psu) in Arroyo la Boya near the mouth of the lagoon. In Río Sábalo salinity fluctuated from 14 to 20 psu, probably due to water stagnation and evaporation, and the low contribution of fresh water from the river, coinciding with that reported by Lankford (1977) and Muciño-Márquez *et al.* (2012). In this regard, García-Cubas & Reguero (1995) point out that the lagoon is predominantly mesohaline, with frequent fluctuations of 5 to 18 psu.

The lagoon has a pH close to neutrality (from 6.7 to 7.9) and is rather well oxygenated with concentration showing greater fluctuations in the rainy season (3.3. mg/L to 11.6 mg/L) than in the dry season (5.3 mg/L to 7.8 mg/L), remaining within the appropriate intervals for aquatic life. These values coincide with those reported by Carbajal (2009) for this lagoon, and by Contreras & Warner (2004), Contreras *et al.* (2005) and López Ortega *et al.* (2012), for other lagoons in the State of Veracruz.

We did not measure nutrients during this study, but according to Castro-Gutiérrez *et al.* (1985), nutrients are not usually limiting for the development of phytoplankton in the lagoon due to river inputs and contributions of matter from the adjacent vegetation together with the recycling mechanisms and processes.

According to Lankford (1977), Castro-Gutiérrez *et al.* (1985), Martínez (1987), Morán (1994), Figueroa-Torres *et al.* (2009), Muciño-Márquez *et al.* (2011a, 2012) and Esquivel & Soto-Castor (this issue), the Sontecomapan lagoon presents three zones, based on morphology, and on gradients of salinity and nutrient concentration. The first is oligohaline with high fresh water influence and highest nutrient level. The second, in the central channel, is considered as a transition zone between mesohaline and polyhaline water, with the lowest nutrient values and greater salinity variations. The third zone, euhaline, corresponds to the lagoon's mouth. This zone behaves as an area of water masses exchange with the coastal zone and over time, recycling of nutrients has been observed at different scales as a consequence of the process of water mass regeneration and population succession. These processes are directly influenced by climatic seasonal changes, hydrometeorological conditions, and circadian and tidal rhythms, all of them widely recognized as important in the study of phytoplankton (Castro-Gutierrez *et al.*, 1985; Guerra-Martínez, 1996; Figueroa-Torres *et al.*, 2009 & Muciño-Márquez *et al.*, 2011 a, b and 2012), and were also probably important drivers for the variability observed in this study.

The dendrogram showed a spatial arrangement in the dry season very similar to the salinity gradients described above, with three different areas clearly related to salinity. However, in the rainy season our results suggest differences also linked to morphological characteristics of the sites. At this period, the Río Sábalo was separated from El Real probably due to its location in a confined area with little current and evaporation processes, maintaining high salinity values while El Real, located near the mouth of the lagoon, had high seawater influence linked to tidal effect (Muciño-Márquez, 2011a, b). Río Basura, farther from the mouth of the lagoon, has higher hydrodynamic and greater fresh water influence than El Sábalo.

From the samples analyzed in this work, a total of 102 phytoplankton species were recorded. The predominance of diatoms and dinoflagellates agrees with that reported by Muciño-Márquez, (2011a, b). The highest abundance occurred in February, 2017, corresponding to the dry season, in agreement with Suchil (1990) who points out that the dry months produce stability and adequate conditions for the best development of phytoplankton.

Of the species identified, 61 are new records for the lagoon, this suggests that species richness may be underestimated, possibly due to the presence of very diverse microenvironmental conditions and to the constant introduction and species exchange of continental and marine origin by the water currents.

Additionally, in the dry season, it was observed that the richness and abundance of species were very variable throughout stations and sampling sites, with importance of freshwater species in the rainy season and of potentially toxic or harmful algae of marine lineage, aspects that were repeatedly observed in previous studies (Guerra-Martínez & Lara-Villa, 1996; Figueroa-Torres *et al.*, 2009).

The most frequent and abundant species was *Tripos hircus*, especially in the dry season, with a maximum density of 20 x 10^4 cells/l. Guerra-Martínez and Lara-Villa (1996) recorded high densities of this species in the lagoon, at salinities of 13-35 psu and temperatures of 30 to 34 °C. The decrease of its population was associated with the increase of freshwater inputs in the system. Figueroa-Torres (1990), Zamudio-Reséndiz (1998) and Okolodkov (2010), among others, have observed that this species is common in the waters of the Gulf of Mexico, and López (1980) considers it as an indicator of nutrient-rich and warm water.

Within the species recorded, 17 are potentially harmful, and 2 may affect human health: the *Dinophysis caudata* dinoflagellate that may cause diarrhea through consumption of contaminated shellfish and, the cyanoprocariota *Lyngbya majuscula* that may cause dermatitis and respiratory irritation.

By combining our results with historical data, we could establish a current register of 357 phytoplankton species for the Sontecomapan Lagoon. This inventory is dominated by diatoms and dinoflagellates, characteristic of brackish and marine environments, but also includes freshwater microalgae such as chlorophytes and euglenofites showing the polyhaline nature of the lagoon ecosystem.

Among this list 46 species formed algal blooms, and 12 of them are potentially toxic, five species affect humans and five more affect marine fauna.

Table 2. Species of phytoplankton of the Sontecomapan Lagoon, Veracruz, that can form blooms.

	Characteristics	References
DIATOMS		
Bacillaria paxillifera (O. F. Müller) T. Marsson	Abundant in the Sontecomapan lagoon in high salinities. Form macroscopic aggregates of brown colonies	1, 18, 20, **28**
Cerataulus smithii Ralfs	Apparently not toxic	27, **28**
Chaetoceros atlanticus Cleve	Some authors report it as a harmfull species in other bodies of water	12, 18, 20 13, 25
Chaetoceros cf. *debilis* Cleve	May cause anoxia due to high number of organisms	18, 20
Chaetoceros holsaticus F. Schütt	Form spring blooms in the Baltic Sea	22
Coscinodiscus centralis Ehrenberg	Non toxic	18, 20, **28**
Coscinodiscus concinnus W. Smith	Harmful species	13, 18, 20
Coscinodiscus granii L. F. Gough	Form mucilaginous aggregates brown green	7, 18, 20
Coscinodiscus jonesiana (Greville) E. A. Sar & I. Sunesen	Apparently non toxic	27, **28**
Cyclotella meneghiniana Kützing	Not toxic. It is found in eutrophic waters	19, 20, **28**
Eupodiscus radiatus Bailey	Apparently non toxic	27, **28**
Melosira moniliformis (O. F. Müller) C. Agardh	It can form long chains on macroalgae in saline waters, flowering has been reported in Hawaiian Stream Ecosystems	16, **28**
Melosira nummuloides C. Agardh	Forms flowering in other bodies of water	6, **28**
Nitzschia longissima (Brebisson) Ralfs	Forms flowering in other bodies of water	20
Pleurosigma rigidum var. *incurvatum* Brun	Forms flowering in other bodies of water	18
Proboscia alata (Brightwell) Sundström	Apparently non toxic	18, 20
Pseudo-nitzschia pungens (Grunow ex Cleve) Hasle	It forms pale green blooms. It produces domoic acid, causing amnesic poisoning in humans, by the consumption of contaminated molluscs	12, 18, 20
Pseudo-nitzschia pungens var. *atlantica* (Cleve) Moreno & Licea	Toxic, produces domoic acid	12, 18
Pseudo-nitzschia seriata (Cleve) H. Peragallo	Toxic, produces domoic acid Causes amnestic poisoning from contaminated shellfish consumption Flowering of this species has been reported in the bays of Sechura and Pisco, Peru.	12, 18, 20, 24

Table 2. (Continuation)

	Characteristics	References
DINOFLAGELLATES		
Akashiwo sanguinea (K. Hirasaka) G. Hansen & Moestrup	It forms yellowish brown blooms. It causes anoxia in other bodies of water	18, 20, **28**
Dinophysis caudata Saville-Kent	Causes diarrhea in humans due to consumption of contaminated shellfish	1, 12, 15, 18, 20, **28**
Gonyaulax polygramma Stein	Form harmful flowers in the southern Gulf of Mexico	8, 15, **28**
Gonyaulax spinifera (Claparède et lachmann) Diesing	Producer of yessotoxin, fish and invertebrates were apparently killed by this bloom, which was rapidly dispersed by tides and wind-forcing.	9, 20
Gonyaulax turbynei Murray & Whitting	Causes anoxia in other bodies of water	18, 20
Gyrodinium spirale (Bergh) Kofoid & Swezy	Orange blooms. Abundant in the Sontecomapan lagoon, in high salinity, it causes anoxia in other bodies of water	18, 20
Noctiluca scintillans (Macartney) Kofoid & Sweay	It forms milky pink, pink or orange blooms. It is found in high concentrations of ammonium	1, 18, 20
Peridinium quadridentatum (F. Stein) Gert Hansen	It has been observed in the Sontecomapan lagoon associated with *Tripos hircus* and *Prorocentrum cordatum*	1
Phalacroma rotundatum (Claparède & Lachmann) Kofoid & Michener	Toxic species, can cause problems to the salmon in captivity by its long silicified chains.	3, 10, 20
Prorocentrum cordatum (Ostenfeld) J. D. Dogge	It has been observed in the lagoon of Sontecomapan, associated to *P. quadridentatum* and *Tripos hircus*	1
Prorocentrum gracile Schütt	Abundant in the Sontecomapan lagoon in high salinities. Has caused fish mortality in Tabasco	14, 18, 19, 20, 21
Prorocentrum mexicanum	It forms algae blooms in other bodies of water.	19
Prorocentrum micans Ehrenberg	Pale-brown blooms. A flowering was reported in Bahía Magdalena, in shallow waters. Occasionally causes hyperoxia or anoxia Produces toxin verupine (hepatoxin) affecting clams and shellfish	15, 18, 19, 20, 21
Protoperidinium depressum (Bailey) Balech	Flowering of this species has been reported in the bays of Sechura and Pisco, Peru.	20, 24, **28**
Scrippsiella acuminata (Ehrenberg) Kretschmann, Elbrächter, Zinssmeister, Soehner, Kirsch, Kusber & Gottschling	Coffee Flowering. Abundant in the Sontecomapan lagoon in high salinity, it causes anoxia in other bodies of water	15, 18, 20, **28**
Tripos furca (Ehrenberg) F. Gómez	It forms orange blooms. May cause anoxia. It has been observed in the Sontecomapan lagoon, at the time dry, sometimes associated with *P. quadridentatum* and *Prorocentrum cordatum*. It has been reported as harmful in Puerto Escondido, Baja California, causing great mortality of tuna in captivity.	11, 12, 15, 18, 20, 23

Table 2. (Continuation)

	Characteristics	References
DINOFLAGELLATES		
Tripos fusus (Ehrenberg) F. Gómez	Produces hydrogen sulfide	11, 12, 18,20
Tripos hircus (Schröder) F Gómez	When it is very abundant it produces anoxia	12, **28**
Tripos muelleri Bory	Apparently non-toxic	18
Tryblionella compressa (J.W.Bailey) Poulin	Yellow brown flowering .Apparently non-toxic	18, 19, 20
EUGLENOPHYTES		
Euglena cf. *stellata* Mainx	Apparently non-toxic	18, 20
Euglena cf. *viridis* (O. F. Müller) Ehrenberg	Apparently non-toxic	18, 20
CYANOPROKARYOTES		
Anabaenopsis circularis (G. S. West) Woloszynska & V. Miller	Produces toxins	5, 18, 26, **28**
Arthrospira platensis Gomont	Apparently non-toxic	2, **28**
Lyngbya majuscula Harvey ex Gomont	May cause respiratory dermatitis and irritation	4, **28**
SILICOFLAGELLATE		
Dictyocha fibula Ehrenberg	Causes anoxia in other bodies of water	18, 20
RAPHIDOPHYCEAE		
Chatonella sp	Produces massive blooms	18
Olisthodiscus luteus N. Cartes	Causes anoxia in other bodies of water. Preliminary data show is hemolytic (disintegrates erythrocytes or red blood cells)	17. 18, 20

(1) Aké-Castillo & Vázquez-Hurtado 2008, (2) Arulmoorthy *et al.* 2017, (3) Cassis *et al.* 2002, (4) Chi *et al.* 2012, (5) Churro *et al.* 2009, (6) Dimar 2011, (7) Fukao *et al.* 2009, (8) Gárate-Lizárraga *et al.* 2006, (9) Gárate-Lizárraga *et al.* 2014, (10) González-Gil *et al.* 2011, (11) Guerra-Martínez 1996, (12) Guerra-Martínez & Lara-Villa 1995, (13) Guiry & Guiry 2017, (14) LESP 2005, (15) Licea *et al.* 2002, (16) Mathew 2007, (17) Moestrup 2002, (18) Muciño-Márquez *et al.* 2011 a, (19) Muciño-Márquez *et al.* 2011 b, (20) Muciño-Márquez *et al.* 2012, (21) Muciño-Márquez *et al.* 2015, (22) Nommann & Kaasik 1992, (23) Orellana-Cepeda *et al.* 2002, (24) Orozco *et al.* 2017, (25) Rivera & Sánchez 2011, (26) Walker 2004, (27) Meave y Moreno (1997), (28) this study.

It is worth noting that most of the toxic species were present in the dry season, so special attention should be paid to their presence at this season. It is not possible to rule out the possibility that the above-mentioned species may be present again and in higher concentrations. Besides new harmful and toxic species may also develop because of accelerated processes of eutrophication of anthropic origin in the last years in the coastal lagoons, which can affect economic production, human health and marine biota.

Conclusions

The Sontecomapan lagoon is an ecosystem with marked hydrodynamic processes at different times of the year. Salinity appears as a determinant factor, which confers a polyhaline character to the system, forming three characteristic zones: the first with strong influence of fresh water, the second brackish and the third with more marine influence.

Other physical and chemical factors such as pH, temperature and dissolved oxygen generate particular microenvironmental conditions that drive the distribution and abundance of phytoplankton species, in addition to which the intrinsic characteristics of the species must be considered in order to tolerate certain environmental conditions. However, there are few studies on this subject and so far, no clear behavioral patterns can be established.

From this study, a list of 102 species was obtained, with 61 new records for the area, with only two species toxic to man and none for aquatic fauna. Combining this list with previous historical data, we could establish a current list of 357 phytoplankton species for the lagoon, of which 46 are potentially harmful or toxic.

Due to the importance of this resource, it is necessary to continue sampling and monitoring the phytoplankton of the Sontecomapan Lagoon, since the harmful and toxic species can develop and reach alarming densities at any time, emphasizing the dry season, when more problematic species are present. It is also important to consider that the accelerated process of anthropic origin eutrophication that occurred in the last years in the Mexican coastal lagoons may affect the economic production, human health and marine biota.

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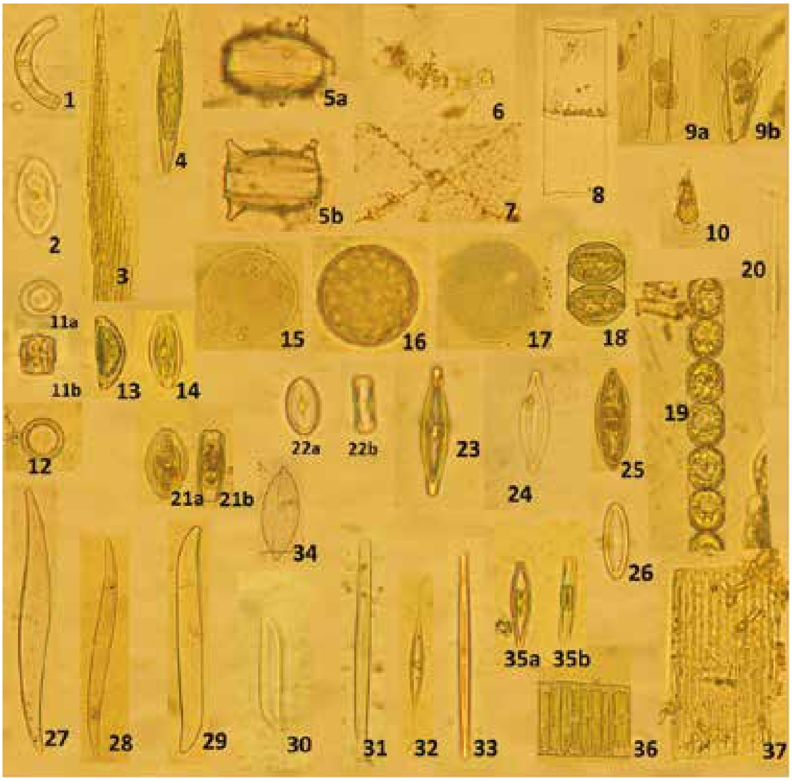

Plate 1 Figures 1. *Aulacoseira granulata*; 2. *Cocconeis placentula*; 3. *Bacillaria paxillifera*; 4. *Brachysira procera*; 5a y b. *Cerataulus smithii;* 6. *Chaetoceros* sp. 1; 7. *Chaetoceros* sp. 2; 8. *Guinardia flaccida*; 9a y b. *Entomoneis alata;* 10. *Entomoneis ornata*; 11a y b. *Cyclotella meneghiniana*; 12. *Cyclotella* sp; 13. *Cymbella minuta;* 14. *Cymbella próstata*; 15. *Coscinodiscus* af *jonesiana*; 16. *Coscinodiscus* af *radiatus*; 17. *Coscinodiscus* sp. 1; 18. *Melosira moniliformis*; 19. *Melosira nummuloides*; 20. *Cylindrotheca closterium*; 21a y b. *Diploneis puella;* 22a y b. *Diploneis smithii;* 23. *Navicula capitatoradiata*; 24. *Navicula veneta*; 25. *Navicula* sp. 1; 26. *Navicula* sp. 2; 27. *Gyrosigma acuminatum;* 28. *Gyrosigma attenuatum;* 29. *Gyrosigma balticum;* 30. *Gyrosigma eximium;* 31. *Gyrosigma* af *scalproides*; 32. *Gyrosigma fasciola*; 33. *Fragilaria* af *fasciculata*; 34. *Lyrella lyra*; 35a y b. *Gomphonema parvulum*; 36. *Eunotia* sp; 37. *Fragilaria* sp.

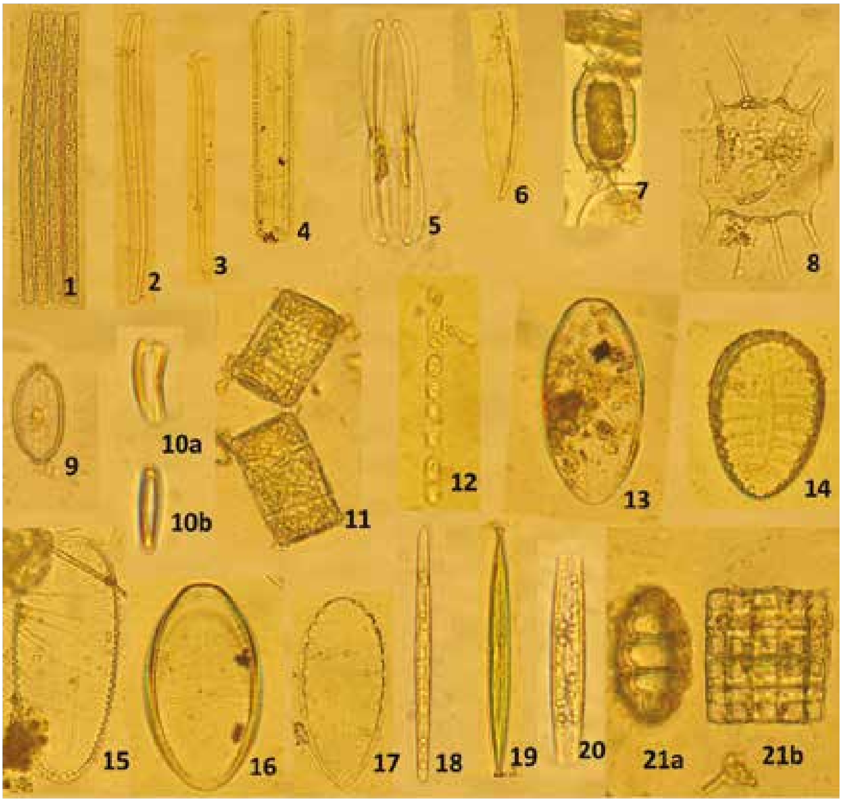

Plate 2 Figures 1. *Nitzschia macilenta*; 2. *Nitzschia sigma*; 3. *Nitzschia sigmoidea*; 4. *Pinnularia* sp; 5. *Plagiotropis lepidoptera*; 6. *Plagiotropis vítrea*; 7. *Odontella longicruris*; 8. *Odontella mobiliensis*; 9. *Petroneis humerosa;* 10. *Rhoicosphenia curvata*; 11. *Pleurosira laevis*; 12. *Skeletonema costatum*; 13. *Surirella* af *guatemalensis;* 14. *Surirella fastuosa* var *Cuneata*; 15. *Surirella febigerii*; 16. *Surirella gemma;* 17. *Surirella splendida*; 18. *Synedra superva;* 19. *Synedra tabulata*; 20. *Thallasionema nitzschioides;* 21a y b. *Terpsinoe musica*.

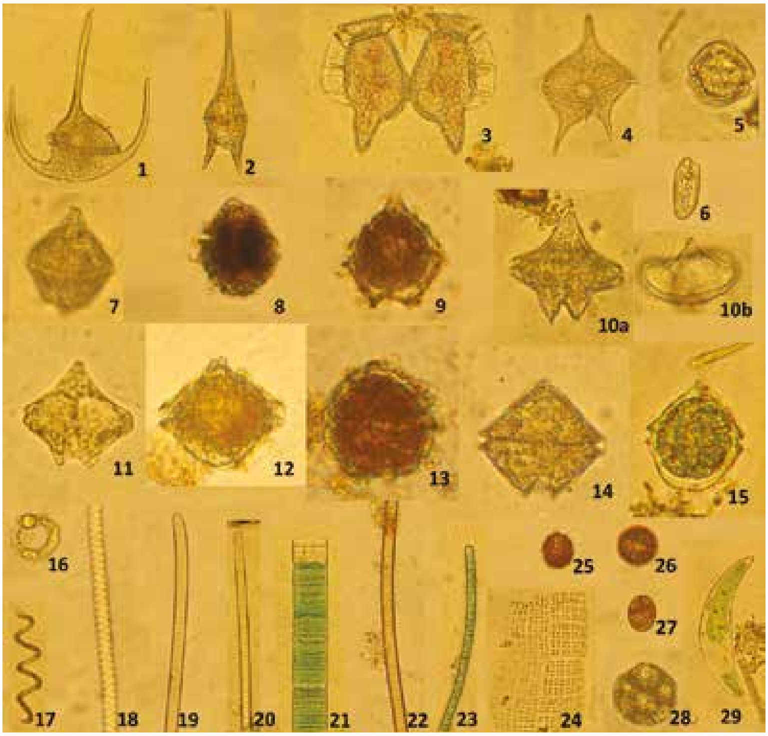

Plate 3 Figures 1. *Tripos tripos*; 2. *Tripos furca* var *hircus*; 3. *Dinophysis caudata*; 4. *Protoperidinium depressum*; 5. *Goniodoma sphaericum*; 6. *Oxyrrhis marina;* 7. *Gonyaulax polygramma*; 8. *Protoperidinium argentinense*; 9. *Protoperidinium brevipes*; 10a y b. *Protoperidinium curtipes*; 11. *Protoperidinium crassipes*; 12. *Protoperidinium punctulatum*; 13. *Protoperidinium thorianum;* 14. *Protoperidinium paulseni*; 15. *Scrippsiella acuminata*; 16. *Anabaenopsis circularis;* 17. *Arthrospira platensis;* 18. *Arthrospira subsalsa;* 19. *Phormidium nigroviride*; 20. *Oscillatoria* af *limosa;* 21. *Oscillatoria obtusa;* 22. *Oscillatoria* sp; 23. *Limnoraphis hieronymusii*; 24. *Merismopedia convoluta*; 25. *Trachelomonas hispida;* 26. *Trachelomonas lefebrei*; 27. *Trachelomonas volvocina;* 28. *Coelastrum microporum*; 29. *Closterium moniliferum*.

The effect of mangrove leaf litter extracts on primary productivity and phytoplankton growth

José Antolín Aké-Castillo* & Gabriela Vázquez**

Abstract: The relationship between dissolved organic matter and phytoplankton function has either a positive or a negative effect on primary productivity and phytoplankton growth. Sontecomapan is a coastal lagoon located at the south of Veracruz, Mexico, which is bordered by a mangrove forest where *Rhizophora mangle* is the dominant species. In the lagoon, the concentrations of folin phenol active substances (FPAS) are indicative of the high input of plant organic matter. Because of the different effects that organic matter can have on phytoplankton function, we performed bioassays to determine the effects of mangrove leaf litter extracts on primary productivity and phytoplankton growth within different seasons. The inhibitory and stimulatory effects observed on primary productivity and phytoplankton growth, are indicative that leachate mangrove is made off of a mixture of both, stimulatory and inhibitory substances. Results suggest that phytoplankton species' tolerance to concentrations of FPAS in the extracts is important for the response. *Chaetoceros muelleri* var *subsalsum*, *Cyclotella cryptica*, and *C. meneghiniana* are sensitive to high concentrations of FPAS while *Skeletonema subsalsum* was able to tolerate moderate concentrations of FPAS. These

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responses support the hypothesis that tolerances to organic compounds in natural systems influence the dynamics of phytoplankton communities.

KEYWORDS: Bioassay, Organic matter, Phytoplankton growth, Primary productivity.

Introduction

Plant organic matter is a source of nutrients and humic substances that affect primary production and influence the phytoplankton dynamics in both, marine and freshwater environments (Herrera-Silveira & Ramírez-Ramírez, 1996; Rivera-Monroy *et al.*, 1998; Danilov & Ekelund, 2001; Klug, 2002). The relationship between dissolved organic matter and phytoplankton function has either a positive or a negative effect on primary productivity and phytoplankton growth, so the net effect of dissolved organic matter is under debate (Klug, 2002; Sánchez-Marín & Beiras, 2014).

Jackson & Hecky (1980) evaluated the effect of dissolved organic matter on the primary productivity of a lake and reservoirs in Canada and found an inverse relationship between primary productivity and dissolved organic carbon. In contrast, Rivera-Monroy *et al.* (1998) demonstrated the stimulatory effect on primary productivity in a series of a runoff addition series from a fringe mangrove forest in water from Terminos Lagoon, Mexico.

The response of phytoplankton growth has been tested using specific humic compounds such as humic acid, fulvic acid and tannins on different microalgae species. These studies show that the stimulatory effect of humic substances results from their capacity to chelate damaging metallic ions such as copper (Toledo *et al.*, 1980, 1982), possible cell sensitization, chelation of essential ions that penetrate the cell (Prakash *et al.*, 1973), and by supplying nitrogen as a nutrient (Granéli *et al.*, 1985). The inhibitory effect is due mainly to humic substances that decrease the availability of trace minerals such as iron (Jackson & Henry, 1980; Imai *et al.*, 1999).

Tropical coastal lagoons and estuaries are characterized by a high input of terrestrial organic materials from surrounding mangrove communities (Flores-Verdugo *et al.*, 1987; Tam *et al.*, 1990). Mangrove leaves have a high concentration of tannins which are liberated rapidly in the early stages of decomposition (Cundell *et al.*, 1979); in coastal systems, the concentration of these substances can be high (Kalesh *et al.*, 2001). Sontecomapan is a coastal lagoon bordered by a mangrove forest where *Rhizophora mangle* L. is the dominant species. The concentration of folin phenol active substances in this lagoon varies seasonally from 0 to 0.236 mg l^{-1} (Aké-Castillo & Vázquez, 2008), with the highest values in the dry season. These concentrations are indicative of the high input of plant organic matter. The phytoplankton dynamics in the lagoon is characterized by blooms of diatoms and dinoflagellates, which dominate the species' composition seasonally (Aké-Castillo & Vázquez, 2008).

Due to the different effects that organic matter can have on phytoplankton function, we experimented with the effects of mangrove leaf litter extracts on primary productivity and phytoplankton growth in different seasons. Our aim was to determine if the mangrove leaf litter was a source of nutrients throughout the decomposition process. To test the effect of substances from mangrove leachate, products from different days of decomposition were used to test the effect on the primary productivity and phytoplankton growth from a Sontecomapan Lagoon natural community.

Materials and methods

Site description

Sontecomapan Lagoon is a shallow coastal lagoon permanently connected to the Gulf of Mexico. It is located in Los Tuxtlas Biosphere Reserve between 18° 30'-18° 34' N and 94° 59'-95° 04' W (Fig. 1). This region is has three climatic seasons: dry from March to May, rainy from June to September, and "Nortes" from October to February. The latter is determined by strong winds coming from the North and sporadic rainfalls. The lagoon is bordered by a mangrove forest where *Rhizophora mangle* is the dominant species (Aké-Castillo *et al*., 2006): it is a brackish water system where salinity varies spatially and temporally from 0 to 35%. Variation in the phytoplankton community is reflected by a dominant species in each season: in the "Nortes" season, the community is dominated by the diatoms *Skeletonema subsalsum*, *S. pseudocostaum* and *S. costatum*; during the dry season, the dinoflagellates *Peridinium quinquecorne*, *Prorocentrum cordatum*, *Ceratium furca* var *hircus*, *Scrippsiella* sp., and the diatom *Thalassiosira cedarkeyensis* are the dominating species; and during the rainy season the lagoon is again dominated by diatoms: *Cyclotella cryptica*, *C. meneghiniana*, *C. striata*, *Chaetoceros holsaticus* and *C. simplex* (Aké-Castillo & Vázquez, 2008). Phytoplankton production is higher in the rainy season and it is correlated significantly to different groups of the zooplankton community (Benítez-Díaz Mirón *et al*., 2014).

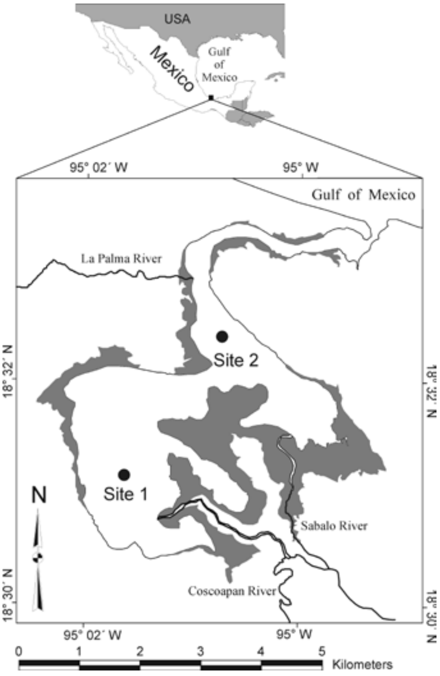

Figure 1. Sontecomapan Lagoon's location indicating primary productivity experiments sites.

Extract obtained from decomposing Rhizophora mangle leaves

Senescent leaves of *Rhizophora mangle* were collected by hand from different sites of the water edge at Sontecomapan Lagoon in February 2003, and transported in a cooler with ice to the laboratory. Over there, leaves from the sites were mixed up and gently washed to eliminate mud. Substances from the decomposing leaves were obtained by incubating 10 g of the wet fresh leaves into 3 l of distilled water in 3 plastics bottles previously washed with a HCI solution at 5%. The chosen concentration (10 g/ 3 l) is an intermediate value of experimental designs of leaf decomposition used in different previous works (Ashton *et al.*, 1999; Aké-Castillo *et al.*, 2006). Three bottles without leaves were used as controls, all bottles were maintained at room/laboratory temperature (26 °C) under natural daylight-darkness conditions and were aerated with low pressure using aquarium pumps. At day 2 of the experiment, water level was maintained by adding distilled water when necessary (without replenish the water from analyses).

As wet fresh leaves were used for the incubations, initial dry weight was estimated by weighing 3 samples of the fresh leaves and then drying them to constant weight at 60 °C in a muffle furnace (Davis III *et al.*, 2003).

Considering classical negative exponential model of mangrove leaf litter decay (Flores-Verdugo *et al*., 1987; Aké-Castillo *et al*., 2006), on days 0, 2, 5, 10, 25 and 45 a sample of 160 ml of water was taken from each bottle for chemical analyses. Ammonium (N-NH_4^+), nitrate (N-NO_3^-), orthophosphate (P-PO_4^{3-}) and folin phenol active substances (tannins and lignins) (FPAS) were determined using colorimetric techniques following the Strickland and Parsons methods (1977), Horwitz (1980), and APHA (1998). Each day a 20 ml sample was stored in ambar jars at 4 °C for later use in the primary productivity and phytoplankton growth experiments. We assumed not evolution of inorganic concentration occurred in refrigeration (APHA, 1998).

Effect on primary productivity

Experimental design. The effect of mangrove leaf extracts on primary productivity (PP) was evaluated during 2003, close to the time when senescent leaves were collected, in three different months representing the three climatic seasons: dry season (May), rainy season (August) and "Nortes" season (October). There were five treatments including controls and three extracts from mangrove leaves in different stages of decomposition. One control consisted in no addition of any extract, and the other one with addition of the freshwater media used for incubating leaf litter. The extracts tested were those obtained on days 2, 10, and 45, which represent times in early, late and inflexion point of leaf litter decay model.. The treatments for the experimental design and characteristics of the extracts are shown in Table 1. Each treatment was triple-tested.

Treatments were tested in water samples from two sites with different marine influence: one in the interior of the lagoon and the other close to the channel connection with the sea (Fig. 1).

Table 1. Characteristics of extracts obtained from decomposing mangrove leaf litter and used in experiments on primary productivity and phytoplankton growth.

Treatment	inoculums	N-NH_4^+µM	N-NO_3^- µM	P-PO_4^{3-}µM	FPAS mgL^{-1}
1	No extract (control)	WL	WL	WL	WL
2	Incubating medium from day 0 (control)	0	13.54 (0)	0.58 (0)	0
3	Extract from day 2	171.52(37.57)	2.43(0.37)	8.32(1.19)	7.30(1.15)
4	Extract from day 10	104.88(8.16)	1.59(0.49)	11.52(2.98)	8.49(0.57)
5	Extract from day 45	110.70(9.28)	1.27(0.36)	5.8(0.71)	12.50(1.96)

WL: water lagoon from each month of experiment (standard deviation)

Primary productivity experiment. Each month, primary production was determined using the light-dark DBO bottle method, and oxygen concentration was evaluated using the Winkler method (Vollenweider, 1974). DBO bottles (300 ml) were filled with water collected from both sites. Except for treatment 1, DBO bottles were inoculated with 1 ml of each extract (0.3% of DBO volume) and were hung 20 cm deep into the lagoon to incubate for 4 hours. The results are expressed as gross primary productivity (GPP) transformed to mg C $L^{-1}hr^{-1}$with a conversion factor of 0.375 mol carbon production with a photosynthetic coefficient of 1.2 (Wetzel & Likens, 2000).

The effect on phytoplankton growth

Experimental design. The effect of mangrove leaf extract on phytoplankton growth was evaluated in phytoplankton cultures from water samples taken in May, August and October 2003 at site 1 in Sontecomapan Lagoon (Fig.1). The phytoplankton cultures represented the three climatic seasons: dry, rainy and "Nortes". The same treatments as in the PP experiment were tested on phytoplankton growth (Table 1). Each treatment was tested in 5 replicates.

Phytoplankton cultures. We used natural water from the lagoon as culture medium. This was prepared by collecting 3 l of water from the Sontecomapan Lagoon at site 1 a month before May, August and October when the phytoplankton inoculations were done (Fig. 1). The collected water was taken to the laboratory and then filtered using a 1.2 µm Millipore membrane and sterilized in an autoclave at 120 °C for 15 minutes. 25 ml Pyrex culture tubes were filled with 20 ml of the medium.

Each month, culture tubes were transported to the field for *in situ* phytoplankton inoculations. Lagoon water with phytoplankton was collected and a 1 ml sample was inoculated in the tubes (cell density in the lagoon is above 1000 cells per milliliter in previous observations). The tubes for each treatment were inoculated with 0.2 ml of the mangrove extracts with the exception of treatment 1 (1% of the total volume of cultures representing final concentrations of humic substances within the range detected in the lagoon). A 125 ml water sample from the lagoon was fixed with acetate-lugol (Throndsen, 1978) to determine the phytoplankton abundance and the species composition at the beginning of the experiment (Table 2).

Table 2. Species composition and cell density (cell ml^{-1}) at the beginning of experiments of phytoplankton growth.

Species	May	August	October
*Chaetoceros muelleri var subsalsum**	174	0	0
Thalassiosira cedarkeyensis	166	0	0
Scrippsiella sp.	6	0	0
Chaetoceros simplex	863	964	0
Cylindroteca closterium	2	2	0
Scenedesmus armatus	0	19	0
Chaetoceros subtilis var abnomis f. simplex	0	2	0
Cyclotella spp.*	0	2540	225
Skeletonema pseudocostatum	0	0	9938
*Skeletonema subsalsum**	0	0	3403
Skeletonema costatum	0	0	0
Fragilaria ulna	16	8	23
Total	1227	3534	13589

* Species that dominated at the end of the experiment

The tubes were taken to the laboratory and placed in a culture chamber (LAB-LINE) at 26 °C with a 12/12 light-dark cycle and 9.9 μmol m^{-2} s^{-1} irradiance. All tubes were gently shaken daily and every 2 or 3 days, a drop was extracted from one tube of each treatment to monitor phytoplankton growth. We wanted to test the effect on cell density as a result of growth, so we established weekly samplings. On days 7, 14, 21, 28, 35, 42, and 49 a 1 ml sample from each tube was fixed with acetate-lugol for cell counting and species identification. This was done in a Neubauer chamber (Semina, 1978) under a light microscope (LEICA).

Statistical analysis

A two-way repeated analysis of variance (ANOVA) was executed to test the differences between the incubating leaves and the controls (treatments), and other one to determine differences in phytoplankton growth. Differences obtained in GPP between treatments were analyzed with a nested factorial ANOVA. Factors were treatments, months, and site nested by month. All data were $\log_{10}$ (x+1) transformed for normalization (Zar, 1999). The Statistica ver 7.0 software was used for the analysis (StatSoft, 2004).

Results

Extract obtained from Decomposing Rhizophora Mangle Leaves

The differences between incubated bottles with leaf litter and controls show that leaf litter decomposition produced N-NH_4^+ (ANOVA: F = 459.81, P < 0.001), N-NO_3^- (ANOVA: F = 55.47, P < 0.001), and FPAS (ANOVA: F = 459.81, P < 0.001). P-PO_4^{3-} concentration in the extract was not significantly different from the control (ANOVA: F = 4.8, P = 0.09).

During the first two days of decomposition there was a rapid increase in N-NH_4^+, N-NO_3^-, and FPAS concentrations (Figs. 2a, 2b, and 2d). After the 5th day there were changes in the nitrogen compounds concentration, but tendency in bottles with leaf litter was to decrease (Figs. 2a, 2b). Variation in P-PO_4^{3-} concentration did not differ over time (Fig. 2c). Also, after the 5th day, FPAS concentration increased slowly in the bottles with leaf litter until the end of the experiment (Fig. 2d).

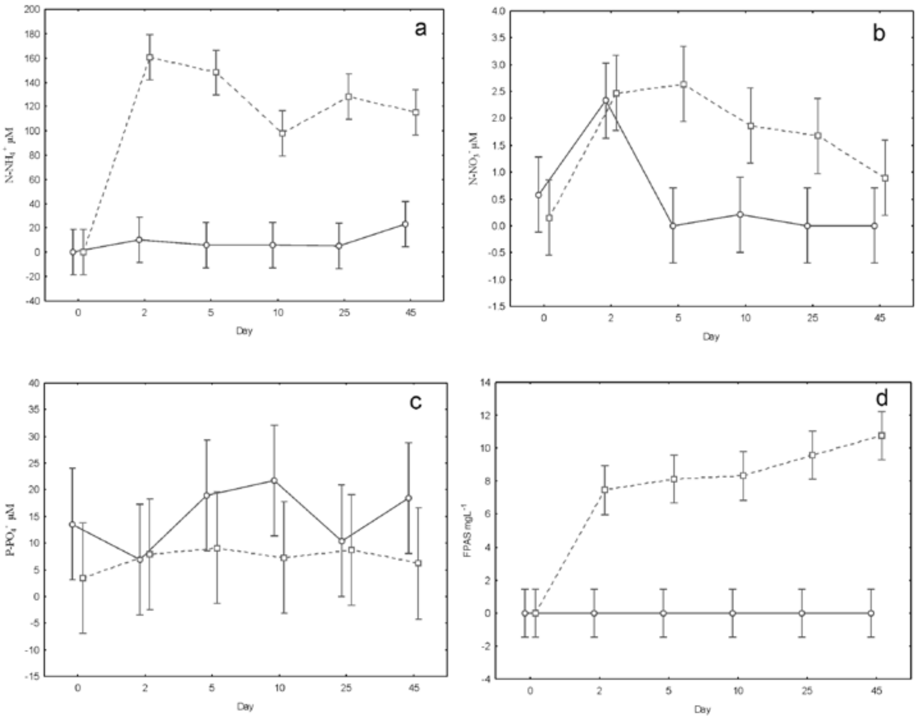

Figure 2. Mean concentration of *a)* N-NH_4^+, *b)* N-NO_3^-, *c)* P-PO_4^{3-} and *d)* FPAS released from 1 g of leaf litter decomposition in 1 l of water over time. Dotted line represents the leaf litter treatment; continuous line represents the control without litter. Whiskers are 95% confidence intervals

The effect on primary productivity

The different treatments had an effect on GPP (ANOVA: F = 12.52, P < 0.001). However significant differences in the effect of treatments were only detected in the experiments carried out in May and August.

In May, treatment 5 (extract from day 45) inhibited PP compared to controls and to the extracts of leaf litter from days 2 and 10 (P-values < 0.001) at site 1 (Fig. 3). At site 2 no significant differences were detected. In August, although no significant effect was detected at site 1, treatment 3 (extract from day 2) stimulated PP. At site 2, the effect of treatment 3 differed significantly from treatment 5 (P = 0.01). These differences resulted from treatment 3 stimulating PP while treatment 5 inhibited it (Fig. 3). In October there were no differences among treatments within the sites, but treatment 3 showed stimulation in GPP at site 2 (Fig. 3).

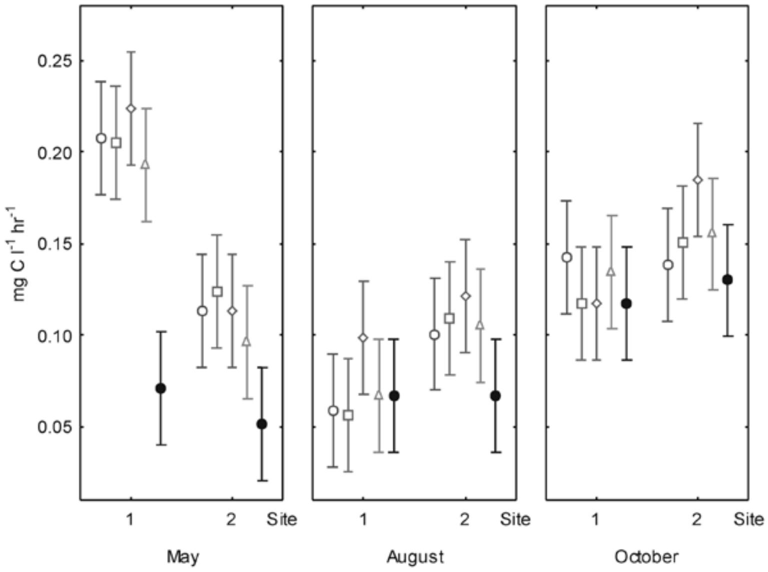

Figure 3. Mean gross primary productivity for two sites on Sontecomapan Lagoon for three different months in 2003. Treatments tested are given in table 1. Whiskers are 95% confidence intervals. Empty circles are treatment 1 (control), squares are treatment 2 (control), the rhombus treatment 3, triangles are treatment 4, and the filled circles treatment 5.

The effect on phytoplankton growth

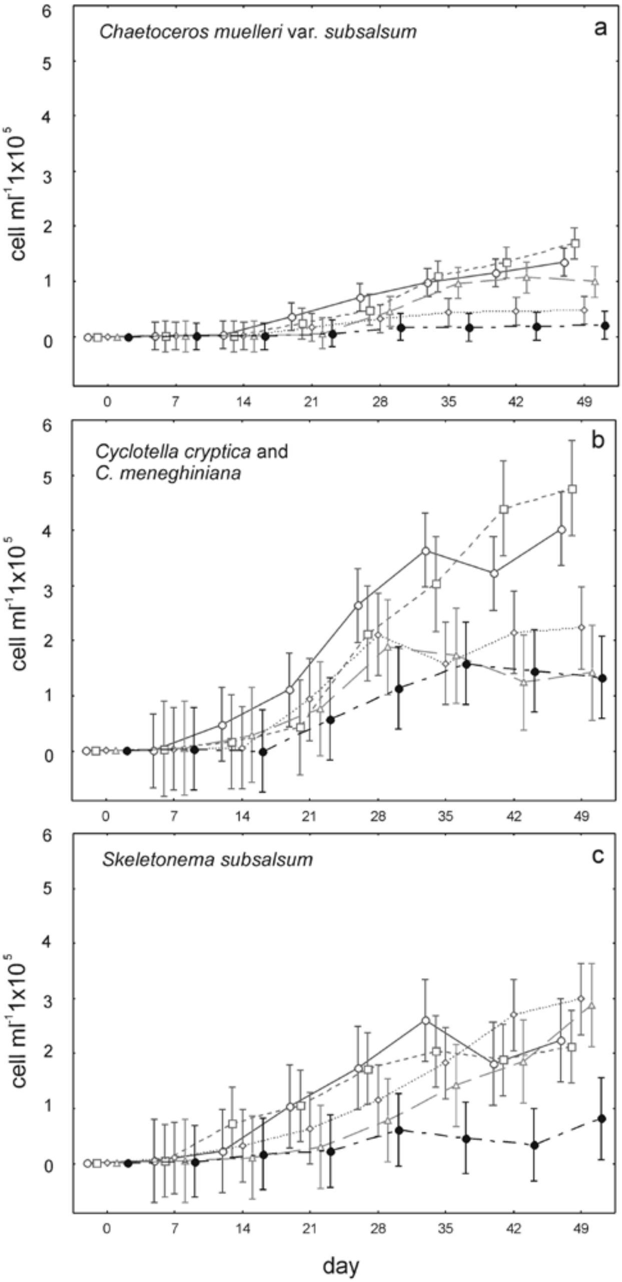

Figure 4. Mean cell density during phytoplankton growth: *a*) *Chaetoceros muelleri* var *subsalsum*, *b*) *Cyclotella* spp., and *c*) *Skeletonema subsalsum* subjected to different treatments (see Table 1). Whiskers are 95% confidence intervals. Empty circles are treatment 1 (control), squares are treatment 2 (control), the rhombus treatment 3, triangles are treatment 4, and the filled circles treatment 5.

All phytoplankton cultures began with at least 4 species in each month evaluated (Table 2), but at the end of the experiments all cultures were dominated by only one species (or genus). In May, *Chaetoceros muelleri* var *subsalsum* (Lemmermann) J.R.Johansen & Rushforth accounted for more than 80% of total cell density, in August, a complex of two *Cyclotella* species (*C. cryptica* and *C. meneghiniana*) accounted for more than 95% and in October, *Skeletonema subsalsum* accounted for more than 92%. The rest of the species did not survive or did not increase cell density significantly, so we only present the treatments effect on the dominant species growth.

On the experiment conducted in May, treatments had effects on the cell density of *Chaetoceros muelleri* var *subsalsum* (ANOVA: F= 4.46, P = 0.01) and cell density was different among days (ANOVA: F= 57.24, P < 0.001). However, differences among treatments did not depend on the day of cell evaluation (ANOVA: F= 0.69, P = 0.84). Treatment 5 (extract of leaf litter obtained on day 45) inhibited growth compared to treatments 1 (control), 2 and 4 (P < 0.05). Cell density under this treatment was always lower than the numbers recorded through the rest of treatments (Fig. 4a). Although the cell density in treatment 3 was maintained below the treatments growth 1, 2 and 4, the difference was not significant (P values > 0.05).

In the August experiment, the effect of the treatments on the cell density of *Cyclotella* spp. depended on the day on which the effect was evaluated (ANOVA: treatments: F= 3.11, P = 0.05; day: F= 47.1, P < 0.001; treatment x day: F= 3.10, P < 0.001). Differences in cell density were detected on day 14, when treatment 5 inhibited the growth compared to treatments 1, 2, and 4 (P values < 0.02). Even though no significant differences were detected on the following days, the treatments with leaf litter extract had lower cell densities than the controls (treatment 1 and 2) after day 35 (Fig 4b).

In October, treatments did not significantly affect the *Skeletonema subsalsum* cell density (ANOVA: F= 2.59, P = 0.09) and this did not depend on the day (ANOVA: F= 0.36, P = 0.99). In spite of this, on day 42 cell density in treatment 3 (extract from day 2) was higher than in treatments 1 and 2 (controls). On day 49, cell density in treatments 3 and 4 was also higher (Fig. 4c). Cell density in treatment 5 (extract from day 45) was always lower than that the rest.

Discussion

Extract obtained from decomposing *Rhizophora mangle* leaves, showed that leaching is a very important process during the first phase of degradation. Leaching has been identified as the main way nutrients are released during the early stages of decomposition (Davis III *et al.*, 2003). The high concentration of N compounds rapidly recorded during this experiment (on day 2) show that a quick mineralization occurred, therefore microbial action was important during the decomposition process (Cundell *et al.*, 1979). The concentration of FPAS was the result of leaching, which went on during the entire decomposition process. Although FPAS comprise different phenolic compounds (APHA, 1998), it is indicative of high tannins concentrations in the leaf mangrove released into the water. The P-PO$_4^{3-}$ dynamics could not be attributed to the decomposing litter as our control had similar concentrations however, varying P-PO$_4^{3}$concentrations suggested that this nutrient $^{-}$was used by microbial activity through accumulation in litter or rapid recycling (Davis III *et al.*, 2003; Gürel *et al.*, 2005).

Effects on primary productivity resulted in both stimulatory and inhibitory effects depending on the concentrations of extracts, thus indicating that mangrove leachate is a mixture of stimulatory and inhibitory substances as well. A stimulatory effect in PP using runoff from mangrove forest has been observed at Términos Lagoon (Rivera-Monroy *et al.*, 1998), but in high quantities the PP is inhibited. Similar responses have been found in experiments on dinoflagellates and diatoms photosynthetic activity (Prakash & Rashid 1968; Prakash *et al.*, 1973), the stimulatory effect may be associated to moderate concentration of humic substances, whereas the inhibitory effect of humic substances contained in water with a high concentration of organic matter can be attributed to their ability to chelate trace metals and phosphate (Jackson & Hecky 1980), or by increasing toxicity through stimulating absorption of heavy metals such as lead, as demonstrated experimentally by Sánchez-Marín & Beiras (2014).

In addition, the stimulatory-inhibitory effect detected in PP experiments suggests that the metabolism of the phytoplankton species present during the months when the experiments were carried out did affect the responses (Table 2). Each species has a different threshold for carrying out biological functions such as photosynthetic activity and growth (Bonilla *et al.*, 1998). The results of the PP experiments indicate that in May the phytoplankton community was composed of species sensitive to high concentrations of FPAS, which inhibited PP. In August, species were more sensitive to the addition of the nutrients that stimulated PP and depressed it when high concentration of FPAS occurred. In October species were tolerant to high concentrations of FPAS.

The effect on phytoplankton growth showed that the species composition was important in the response; the only significant effect (growth suppression) was detected for the extract with a high FPAS concentration. The species responses indicated that *Chaetoceros muelleri* var *subsalsum* is more sensitive to high FPAS concentrations than *Cyclotella* spp (complex with *C. cryptica, C. meneghiniana*) or *Skeletonema subsalsum*. *Cyclotella* spp. was sensitive to high FPAS concentrations in its early growth stage, but later it was able to tolerate them. The *S.subsalsum* growth was not affected significantly by any treatment, but its positive response in treatments 3 and 4 indicate that this species might be able to tolerate moderate concentrations of FPAS. These responses support the idea that tolerance to organic compounds in natural systems can determine the dynamics of phytoplankton communities (Herrera-Silveira & Ramírez-Ramírez, 1996; Aké-Castillo & Vázquez, 2008).

The results of this study demonstrate the effects of mangrove extracts on two different metabolic functions acting at two different time scales: photosynthetic rate in the short term, and growth in the mid term. Photosynthetic activity is a basic function that determines growth, so the stimulatory and inhibitory effects on this process are reflected in the species' growth. Extracts of leaf litter can enhance PP by supplying nutrients, but if the FPAS concentration is too high there is no stimulatory effect and growth can be stopped. The negative effect of the extract on PP in May decreased the growth of *Chaetoceros*. The antagonistic stimulatory and inhibitory effect on PP may be reflected in the adjustment of *Cyclotella*'s metabolism observed during its growth in August. The lack of a significant effect on PP in October may reflect the lack of significant effect on the *S. subsalsum* growth. Although all the responses were observed in the diatom group, other microalgae are known to respond in a similar way (Prakash & Rashid, 1968; Heil, 2005; Sánchez-Marín&Beiras, 2014).

These results support the observations made during a one-year study of phytoplankton dynamics in Sontecomapan Lagoon, where *C.cryptica* and *C. meneghiniana* were associated with a low FPAS concentrations gradient, whereas *S.subsalsum* was associated with moderate FPAS concentrations (Aké-Castillo & Vázquez, 2008). Although *C.muelleri* var *subsalsum* did not account for more than 3% of the total cell density in that study, this species can potentially grow in moderate FPAS concentrations.

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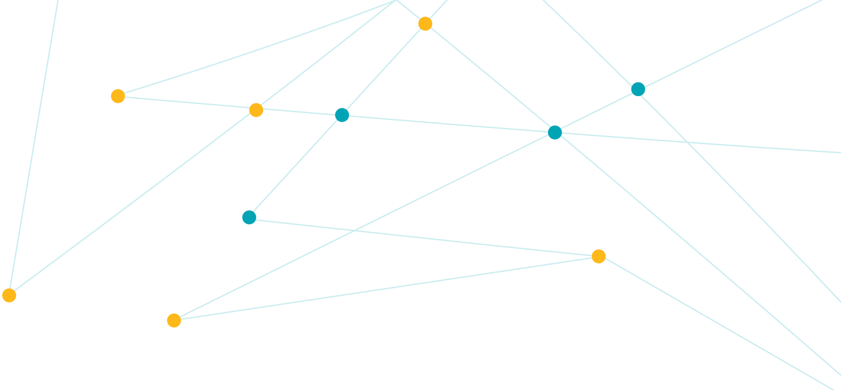

PART 5

Secondary producers

Biomass, size structure and trophic compartments of the metazooplankton in the Sontecomapan Lagoon (Veracruz, Mexico)

M. I. Benítez-Díaz Mirón, M. E. Castellanos-Páez, G. Garza-Mouriño, M. J. Ferrara-Guerrero & M. Pagano

Biomass, size structure and trophic compartments of the metazooplankton in the Sontecomapan Lagoon (Veracruz, Mexico)

M. I. Benítez-Díaz Mirón*, M. E. Castellanos-Páez*, G. Garza-Mouriño*, M. J. Ferrara-Guerrero** & M. Pagano***

ABSTRACT: In this chapter, based on re-analysis of previous published data, we study the biomass, trophic component and size structure of the zooplankton in the Sontecomapan Lagoon, in order to better evaluate its trophic status and assess its possible incidence on food transfers, eutrophication and terminal productivity in this lagoon. Our results show very low values of zooplankton biomass and zooplankton/phytoplankton ratio, compared to literature data, revealing an overall low food-transfer at the basis of the food chain. The size-structure is in average dominated by small organisms which suggests a predation impact from fish on larger organisms (calanoid copepods). This impact was particularly important during the north wind season perhaps in relation with the breeding cycles of important fish species such as *Cathorops aguadulce*. However, tidal inputs of large-sized coastal marine organisms (e.g. *Paracalanus aculeatus*, *Temora turbinata* and *Centropages velificatus*)

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seems to compensate this "erosion" of large zooplankton in the region close to the communication channel. Our data analyses confirm the top-down effect on zooplankton, but also show that bottom-up forces affect phytoplankton (NH_4 limitation) and zooplankton (phytoplankton limitation), meanwhile the top-down control by herbivorous zooplankton on phytoplankton is very low ($<10\%$ of the phytoplankton stock day^{-1}).

KEYWORDS: Zooplankton biomass, Trophic structure, Tropical coastal lagoon.

Introduction

Tropical coastal lagoons like Sontecomapan, are often considered as hot spots for biodiversity and are among the most productive ecosystems in the world (Mitsch & Gosselink, 1993; Day *et al.*, 2012), but they are also very sensitive to human activities and climate variability (Cloern *et al.*, 2016). Their high productivity combined with favorable hydrodynamic conditions, as well as the presence of sheltered areas (as mangroves) make these ecosystems good nursery areas for many fish species (Esteves *et al.*, 2008). Fisheries of these lagoon environments are often key sectors for the economy of local population. However, the fishery potential is highly dependent on planktonic productivity that is, in itself, essential to support the diversity of higher trophic levels. The phytoplankton, first pelagic primary producer, forms the basis of the trophic network. Zooplankton is then a major link between lower and higher trophic levels hence supporting their biodiversity (Rose *et al.*, 2010).

Zooplankton organisms often represent essential prey for larvae, crustaceans and fish juveniles, and become a key for their recruitment (Munk, 1997; Østergaard *et al.*, 2005). Furthermore, in lagoon and coastal marine ecosystems the zooplankton includes large proportions of early larval stages of many benthic and nektonic organisms, some having high commercial value, and are thus key for their recruitment (Archambault & Bourget, 1999). When composed mainly of herbivorous organisms, zooplankton exert a top-down control on primary production and then is a key for controlling accumulation of phytoplankton and eutrophication (Sommer & Sommer, 2006; York *et al.*, 2014). Then, to evaluate the fishery potential of lagoon ecosystems, it is essential to evaluate the biomass and size structure of the different trophic compartments of the planktonic ecosystem (Jeppesen *et al.*, 2000).

Fishing represents a major consumptive heritage use of the coastal lagoons along the Gulf coast of Mexico which produce around 90,000 tons annually (Smardon, 2006). In the Sontecomapan Lagoon, fishing activity takes place both at sea and the lagoon with a 289 boat fleet. It targets mainly three species of fish (*Mugil cephalus*, *Arius melanopus*, *Cathorops aguadulce*, *Eugerres* sp.), crabs (*Callinectes* sp., *Mennipe mercenari*), and prawn (*Litopenaeus setiferus*). The ichthyofauna of the lagoon is very rich, with 52 fish species that belong to 24 families and 41 genera reported by Rodríguez-Varela *et al.* (2010), however, Castillo-Rivera & Lara-Domínguez (this issue) more recently reported 115 species corresponding to 26 orders, 41 families and 84 genera.

In a previous work (Benítez-Díaz Mirón *et al.*, 2014) we showed that transparency, salinity, and food availability were important drivers for zooplankton abundance, composition and diversity. We also discussed that the low zooplankton density found in the lagoon, compared to other eutrophic coastal lagoons, could be attributed to the combined effects of high water exchanges, low depth and high transparency, favoring instability and vulnerability to UV-effects and/or to visual predation.

In this chapter, focused on the biomass and size structure of zooplankton, we completed this analysis, in order to better evaluate the trophic status of the planktonic ecosystems and assess the possible incidence on food transfers, eutrophication and terminal productivity.

Materials and Methods

Three sampling surveys, covering a 10-station network with different characteristics (Figure 1) were performed in March (26 to 29), June (11 to 14) and November (19 to 22), 2010 corresponding to dry, rainy and north wind seasons respectively. During each survey the ten stations were sampled once for environmental variables, bacterial abundance, chlorophyll concentration and zooplankton. All sampling and measurements were performed during daylight hours (between 9:00 and 16:00 h). The sampling strategy, and the measurements techniques for the abiotic (transparency, pH, salinity, temperature, nutrients) and biotic (bacteria biomass and chlorophyll pigments), variables are detailed in Benítez-Díaz Mirón *et al.* (2014).

The zooplankton was collected using a cylinder-conical net (64 μm in mesh opening size, 30 cm in mouth diameter and 1 m in length) and the samples were preserved with 4% formalin.

Zooplankton taxa identification and enumeration were performed as in Benítez-Díaz Mirón *et al.* (2014).

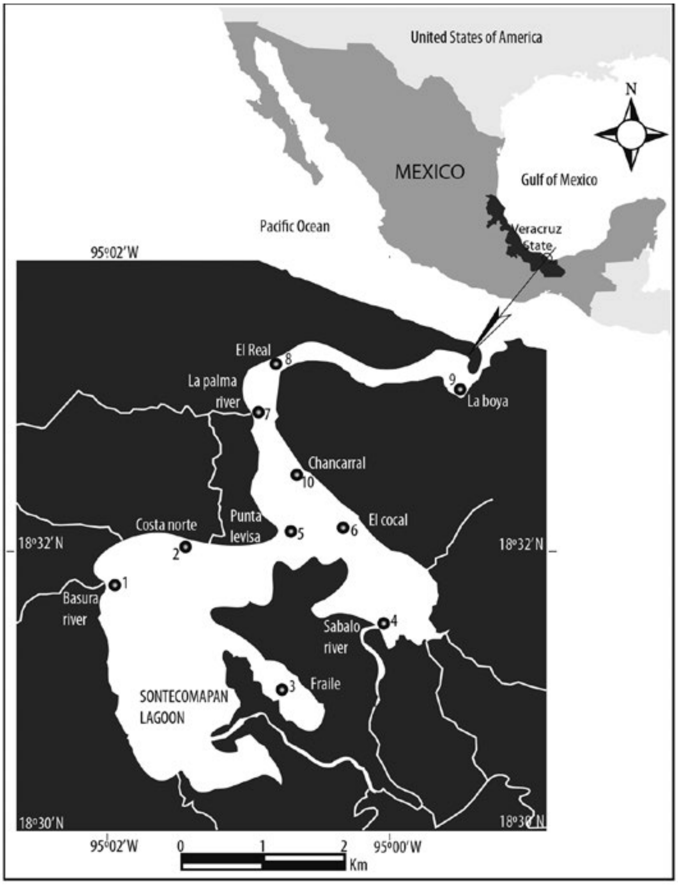

Figure 1. Sontecomapan Lagoon with the location and number of each sampling station.

Individual size and weight of zooplankton organisms and estimation of zooplankton biomass

The biovolume (μm^3) of rotifers was determined by measuring 20 individuals of each species, using the formulae described by Ruttner-Kolisko (1977). The biomass was then estimated assuming a relative density of 1.0. For species that appeared less frequently, such that the total of necessary individuals for measurements were not attained, the corresponding values of biomass were obtained from Dumont *et al.* (1975), Sarma *et al.* (2008) and Basinska *et al.* (2010).

The individual sizes of the other zooplankton taxa were computed from literature: summarized data for copepod species in Razouls *et al.* (2005-2017), mean size values of the other taxa, from Pagano & Saint-Jean (1994; lagoon Ebrié, Ivory Coast). Then the individual weight (as carbon, dry weight and wet weight units) were computed using length-weight relationships from the literature: Mauchline (1998) for calanoid copepods, Chisholm & Roff (1990) for other copepods, and Davis & Wiebe (1985) for other zooplankton taxa.

The individual size, biovolume and weight for each taxon are summarized in Table 1.

To assess the spectrum size, we expressed the size of all organisms with an equivalent spherical diameter computed from the wet weight values and assuming a relative density of 1.

The biomass of each taxon (expressed as mg C m^{-3}) was obtained multiplying the abundance values from Benítez-Díaz Mirón *et al*. (2014) by the individual carbon weights.

Zooplankton taxa diets and constitution of trophic categories

Using common data from literature the zooplankton taxa were grouped into three categories according to their feeding modes (Table 1):

- Detritus - feeders included Poecilostomatoid copepods (genera *Oncaea* and *Corycaeus*), harpacticoid copepods (genera *Canuella, Tisbe,* and undetermined), polychaete larvae and chironomid larvae.
- Predators / carnivorous included cyclopoid copepods from freshwater origin belonging to the genera *Mesocyclops* and *Thermocyclops*, chaetognaths, hydrachnids, and other water mites.
- Herbivorous-omnivorous (suspension feeders) included all other zooplankton taxa.

Estimation of zooplankton carbon demand and grazing impact

Zooplankton carbon demand (ZCD) of suspension feeding zooplankton was computed based on estimates of biomass and ration:

$$\text{ZCD (mgC m}^{-3}\text{ d}^{-1}) = \text{Ration } B_{zoo}$$

where B_{zoo} is the biomass of zooplankton in mgC m^{-3}, and Ration is the amount of food consumed per unit of biomass, calculated as:

$$\text{Ration (d}^{-1}) = (gz + r) / A$$

where *gz* is the growth rate, *r* is the weight specific respiration and A is assimilation efficiency; *gz* was calculated following Zhou *et al.* (2010)

Table 1. Mean values of individual size (as equivalent spherical diameter, ESD, and total length), volume and weight (as wet weight, dry weight and carbon weight) of the zooplankton taxa recorded. The second column indicates the trophic group (based on literature knowledge): H=herbivorous / omnivorous, D=detritivorous, C=carnivorous

Taxa	Trophic group	ESD μm	Length μm	Volume μm^3	Wet weight μg ind^{-1}	Dry weight μg ind^{-1}	Carbon weight μg ind^{-1}
ROTIFERA							
Brachionus plicatilis Müller, 1786	H	103		0.57	0.571	0.057	0.026
Brachionus angularis Gosse, 1851	H	62		0.13	0.127	0.013	0.006
Brachionus dimidiatus Bryce, 1931	H	145		1.59	1.591	0.159	0.072
Keratella americana Carlin, 1943	H	74		0.22	0.216	0.022	0.010
Keratella quadrata (Müller, 1786)	H	101		0.53	0.533	0.053	0.024
Platyias quadricornis (Ehrenberg, 1832)	H	182		3.16	3.165	0.316	0.142
Plationus patulus (Müller, 1786)	H	87		0.35	0.347	0.035	0.016
Euchlanis incisa Carlin, 1939	H	104		0.60	0.597	0.060	0.027
Euchlanis dilatata Ehrenberg, 1832	H	109		0.68	0.677	0.068	0.030
Lecane bulla (Gosse, 1851)	H	70		0.18	0.176	0.018	0.008
Lecane quadridentata (Ehrenberg, 1830)	H	86		0.33	0.327	0.033	0.015
Lecane lunaris (Ehrenberg, 1830)	H	62		0.13	0.127	0.013	0.006
Lecane nana (Murray, 1913)	H	57		0.09	0.095	0.009	0.004
Lepadella (Lepadella) donneri Koste, 1972	H	74		0.21	0.212	0.021	0.010
Lepadella (Lepadella) patella (Müller, 1773)	H	64		0.14	0.139	0.014	0.006
Mytilina ventralis (Ehrenberg, 1830)	H	184		3.26	3.262	0.326	0.147
Mytilina bisulcata Luks, 1912	H	107		0.65	0.647	0.065	0.029
Synchaeta oblonga Ehrenberg, 1832	H	222		5.76	5.761	0.576	0.259
Synchaeta bicornis Smith, 1904	H	217		5.37	5.365	0.537	0.241
Trichotria tetractis Ehrenberg, 1830	H	106		0.63	0.630	0.063	0.028
Testudinella patina Hermann, 1783	H	90		0.39	0.388	0.039	0.017
CLADOCERA							
Chydorus sp.	H	416	446	37.59	37.592	3.759	1.692
Ceriodaphnia sp.	H	201	360	4.28	4.283	0.428	0.193
Penilia avirostris Dana, 1849	H	295	613	13.50	13.500	1.350	0.608
COPEPODA							
Non identified nauplii	H	133	156	1.24	1.244	0.124	0.056
Acartia tonsa Dana, 1852	H	436	800	43.29	43.295	4.329	1.948
Centropages velificatus Oliveira, 1947	H	1203	1700	912.18	912.179	91.218	41.048
Paracalanus aculeatus Giesbrecht, 1888	H	754	1100	224.20	224.202	22.420	10.089
Phaenna sp.	D	792	2000	259.91	259.915	25.991	11.696
Temora turbinata Dana, 1852	H	896	1500	376.46	376.459	37.646	16.941
Pseudodiaptomus sp.	H	558	784	90.74	90.739	9.074	4.083
Mesocyclops spp.	C	390	846	30.98	30.976	3.098	1.394
Oithona nana Giesbrecht, 1892	H	264	600	9.67	9.672	0.967	0.435
Oithona sp.	H	264	600	9.67	9.672	0.967	0.435
Corycaeus sp.	D	433	800	42.37	42.369	4.237	1.907
Oncaea venusta Philippi, 1843	D	536	1100	80.53	80.527	8.053	3.624
Euterpina acutifrons Dana, 1852	D	361	600	24.55	24.546	2.455	1.105
Canuella sp.	D	503	1000	66.81	66.806	6.681	3.006
Tisbe sp.	D	503	1000	66.81	66.806	6.681	3.006
unidentified Harpacticoid sp1	D	320	500	17.17	17.171	1.717	0.773
unidentified Harpacticoid sp2	D	503	1000	66.81	66.806	6.681	3.006
OTHER HOLOPLANKTON							
Ostracods	H	594	500	109.62	109.620	32.886	5.919
Nematoda	H	502	2000	66.10	66.097	6.610	2.313
Appendicularia	H	283	1122	11.90	11.900	1.190	0.536
Chaetognatha	C	334	2847	19.43	19.428	1.748	0.680
Hidroarachnida	C	131	500	1.17	1.169	0.222	0.093
Watermite	C	104	400	0.59	0.586	0.111	0.047
MEROPLANKTON							
Polychaeta larvae	D	256	1000	8.80	8.800	0.880	0.308
Polychaeta juveniles	D	96	362	0.46	0.458	0.046	0.016
Cirripeda larvae	H	289	337	12.60	12.600	1.260	0.567
Chironomid larvae	D	502	2000	66.10	66.097	6.610	2.313
Gasteropod larvae	H	240	156	7.20	7.200	0.720	0.324

Table 1. (Continuation)

Taxa	Trophic group	ESD µm	Length µm	Volume µm³	Wet weight µg ind⁻¹	Dry weight µg ind⁻¹	Carbon weight µg ind⁻¹
Bivalve larvae	H	132	120	1.20	1.200	0.120	0.054
MICRONEKTON							
Decapod larvae	C	547	2000	85.56	85.564	16.26	6.85
Fish larvae	C	1229	2500	971.19	971.189	174.81	76.72

$$g\,(w,T,C_a) = 0.033\left(\frac{C_a}{C_a + 205e^{-0.125T}}\right)e^{0.09T}w^{-0.06}$$

as a function of sea water temperature (T, °C), food availability (C_a, mgC m^{-3}, estimated from Chl-*a*), and individuals weight (w, mgC).

Following Nival *et al.* (1975) we considered a constant value of $A = 0.7\ d^{-1}$

Respiration, r, was estimated from gz using an empirical relationship established for tropical lagoon zooplankton by Pagano & Saint-Jean (1994):

$$\text{Ln } r = 0.134 \ln gz - 0.340$$

We compared ZCD to the phytoplankton stock, converted to carbon assuming a classical C:Chl-*a* ratio of 50:1 to estimate the potential clearance of phytoplankton by copepods.

Data analysis

In the results, all mean values are presented with indication of standard deviation (mean ± SD).

Two-way ANOVAs were performed to test the effects of sampling survey (dry season, rainy season and north wind season), stations on the biotic and abiotic parameters and on zooplankton. We used linear model after checking for normal distribution of each variable using Saphiro-Wilk test. Tukey's post-hoc test of honest significant difference (HSD) was also performed to compare the mean group values.

Stepwise multiple regression analyses were conducted to explain the variability in zooplankton and phytoplankton biomasses. Relationships were tested between these biomasses and variables describing main environmental conditions (temperature, salinity, oxygen), food resources (nutrients for phytoplankton and phytoplankton for zooplankton) and potential consumers (herbivorous zooplankton for phytoplankton, and fish and decapod larvae for zooplankton).

The spatial and seasonal variability of biomass percentages of zooplankton groups or biomass percentages of zooplankton size classes were tested using principal component analyses (PCA). In each analysis zooplankton and phytoplankton biomass, temperature and salinity were considered as additional variables. To increase homoscedasticity, all the data were transformed (log + 1) before analysis.

ANOVA and regression analyses were performed with Statistica v6 software. PCAs were performed with PRIMER 6 software.

Results

Taxonomic and trophic composition of the zooplankton biomass

Zooplankton biomass varied between 0.1 and 230 mgC m^{-3} with a very high outlier value (230 mg m^{-3}) at station 4 during the dry season linked to a zooplankton patch (mostly due to the cyclopoid copepod *Oithona nana*) (Fig. 2), thus this value was discarded from the statistical data analysis. The mean biomass was significantly higher during the dry season than during the rainy and north wind seasons (ANOVA, $p=0.018$; Fig. 2) whereas no significant difference was found between stations despite a very high mean value at station 6 (Cocal) (Fig. 2) due to the maximal value (35 mg C m^{-3}) recorded during the dry season. The zooplankton biomass was strongly dominated by copepods except in some occasions where other holoplankton (station 1, dry season, stations 7 and 9, rainy season) or meroplankton (stations 2 and 3, dry season, stations 2 and 5, north wind season) became very abundant and even dominant. Cladocerans and rotifers represented a very low fraction of the biomass, and almost disappeared during the north wind season together with other holoplankton, therefore, during the north wind season the zooplankton biomass was quite exclusively composed of copepods and meroplankton organisms.

The comparison of phytoplankton biomass with the biomass of the main zooplankton trophic groups revealed a very unbalanced trophic chain, with a very strong biomass decrease between the first two trophic levels (phytoplankton and herbivorous); in most cases, except at station 4 during the dry season, due to the patch of *Oithona nana* (Fig. 3). The mean herbivorous/phytoplankton ratio was 1.5% ± 0.28% with most values <1%. Detritivores represented up to 45% of the zooplankton biomass, with a mean percentage of 7.9 ± 2.2%. Predaceous zooplankton was always <1% of the biomass and were absent in many cases.

Biomass size-structure

The zooplankton biomass was dominated by small organisms of the size-class <200 μm, representing up to 97% of the total biomass (mean = 33 ± 5%; Fig. 4). The largest size-class (>600 μm) was the second in terms of importance (0 to 95% of the biomass; mean = 19.9 ± 5.2%), whereas the intermediate class (300-400 μm) was the less representative (0-13%; mean = 1.9 ± 0.6%). This resulted roughly in a bimodal size-structure with two peaks, for small and large organisms. This bimodal size-distribution disappeared during the north wind season, with a significant increase of the smallest class (<200μm) and a significant decrease of the largest one (>600 μm), as compared to the two other seasons (ANOVA, p=0.008 and 0.016 respectively). In stations close to the communication with the ocean (stations 7, 8 and 9) the size-distribution was shifted towards large organisms with a significant reduction of the smallest class (<200 μm) and a significant increase of the largest one (>600 μm) as compared with the other stations (ANOVA, p=0.003 and 0.045 respectively).

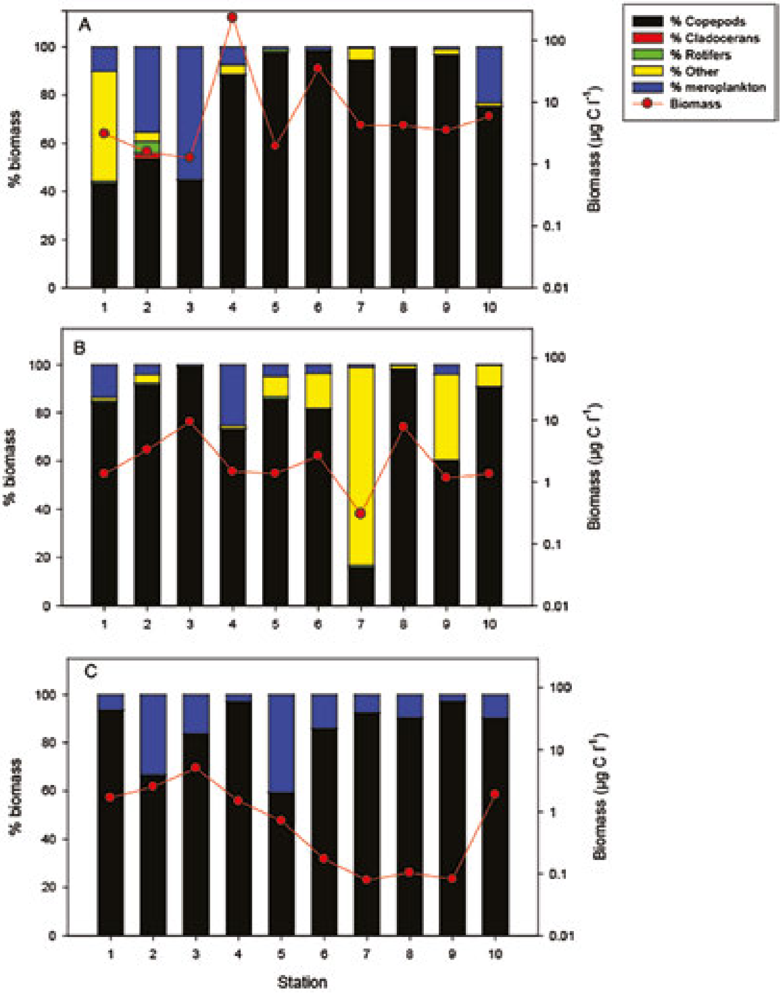

Figure 2. Variations for total zooplankton biomass and for biomass percentages of the main zooplankton taxonomic groups during the dry (A), rainy (B) and north wind (C) seasons.

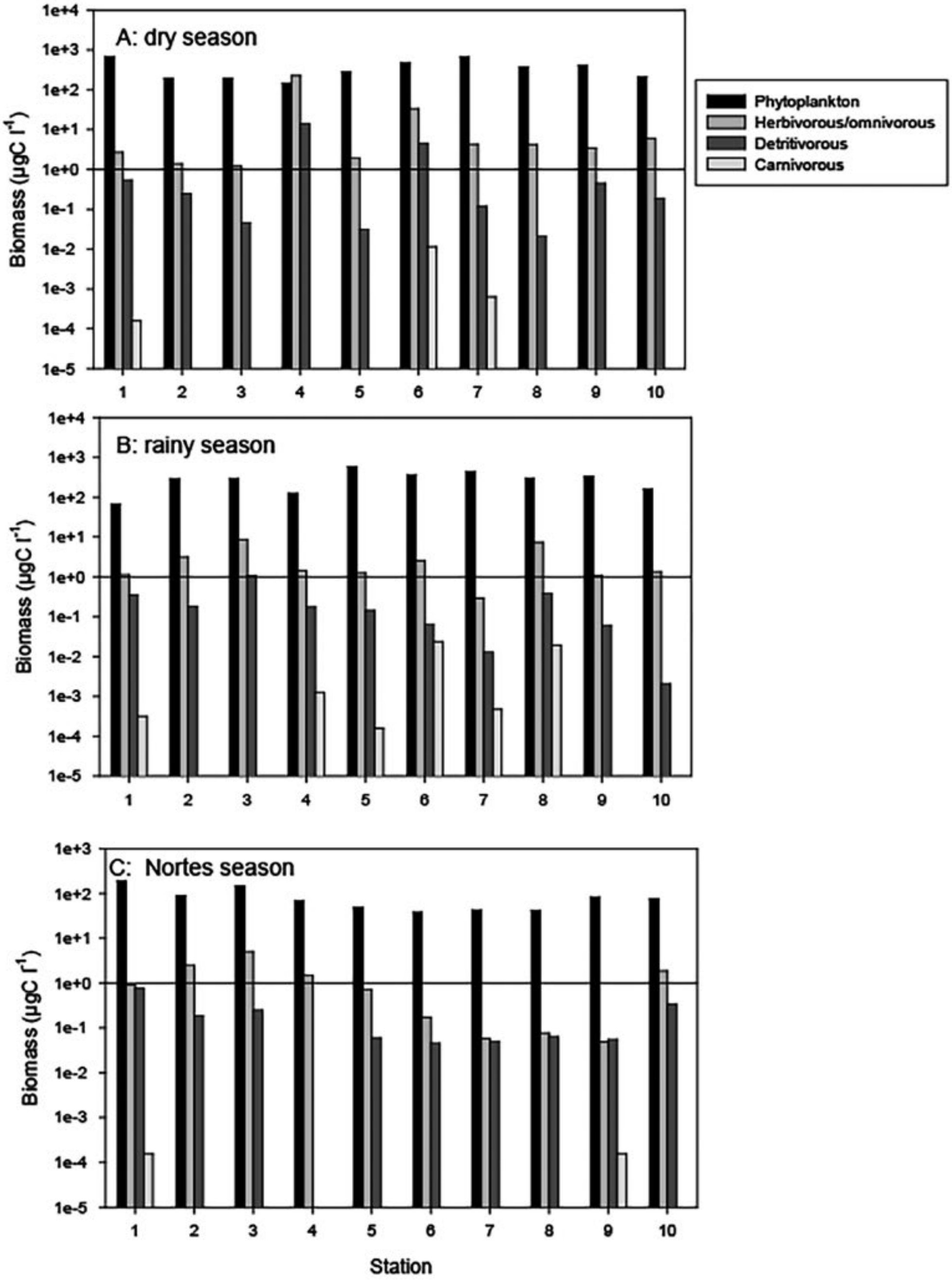

Figure 3. Variations for phytoplankton biomass and for the main zooplankton trophic groups during the dry (A), rainy (B) and north wind (C) seasons.

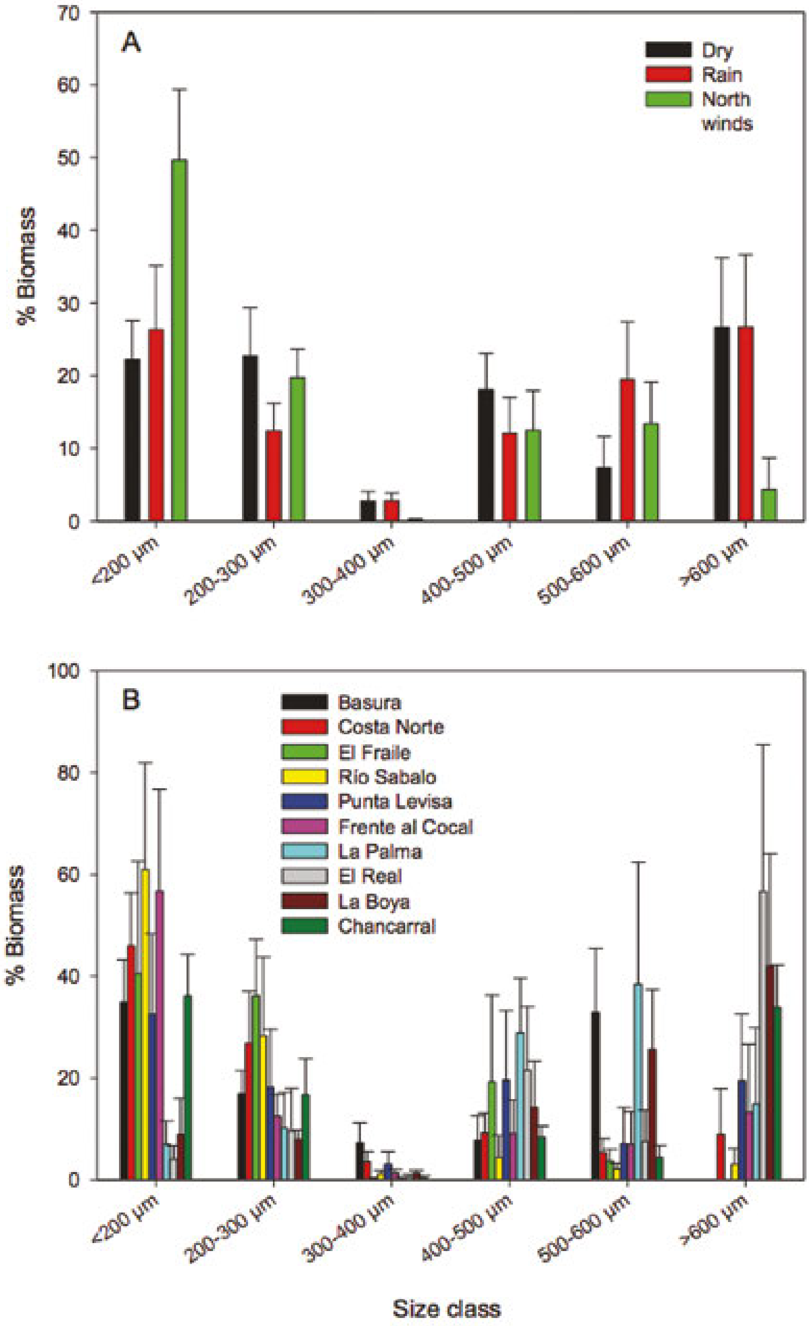

Figure 4. Variation for biomass percentage of the six zooplankton size-classes: mean values per season (A) and sampling station (B).

Zooplankton grazing impact on phytoplankton

If we exclude the abnormally very high value at station 4 during the dry season, the zooplankton grazing pressure varied between 0.1 and 46 μgC l^{-1} d^{-1} (mean = 4.9 ± 1.6 μgC l^{-1} d^{-1}) which represented 0.1 to 9.8% of the phytoplankton stock removed daily (mean = 2.0 ± 0.4% d^{-1}; Fig. 5). The grazing pressure was significantly higher during the dry season than during the north wind season (ANOVA, p=0.042) but did not differ significantly between sampling stations. It was significantly correlated to the phytoplankton biomass (r=0.585; p=0.001) suggesting that bottom-up control prevailed.

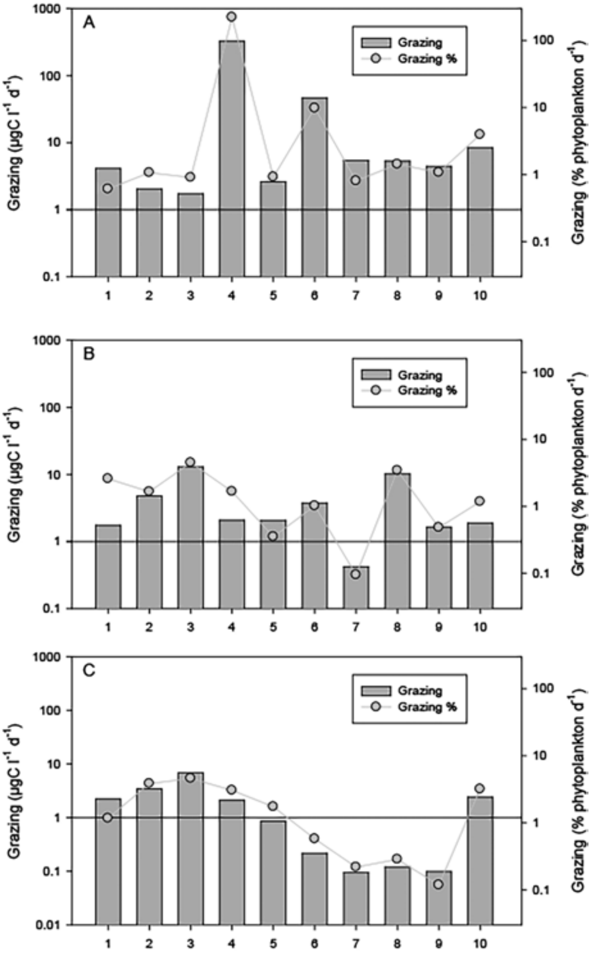

Figure 5. Variations for zooplankton grazing and % grazing impact during the dry (A), rainy (B) and north wind (C) seasons.

Multiple regression analysis for Phytoplankton and Zooplankton biomass

Table 2. Equation parameters of the multiple linear regression models, significances and partial correlation coefficients

	Beta			p	Partial correlation
Zooplankton biomass, $R^2 = 0.048$; $p<0.0072$					
Intercept	1.005	±	5.072	0.845	
Temperature	-1.448	±	3.639	0.694	-0.083
Salinity	0.027	±	0.219	0.903	0.026
Phytoplankton	0.699	±	0.185	0.001	0.619
Fish larvae	-0.886	±	0.382	0.029	-0.436
Decapod larvae	0.120	±	1.292	0.927	0.019
Phytoplankton biomass, $R^2 = 0.49$; $p<0.028$					
Intercept	-1.427	±	0.665	0.044	
Temperature	0.288	±	0.216	0.197	0.279
Salinity	0.034	±	0.249	0.892	0.030
NH_4	2.618	±	1.024	0.018	0.487
Nox	0.581	±	0.402	0.163	0.301
PO_4	-0.153	±	0.349	0.665	-0.095
N/P	-0.595	±	0.396	0.148	-0.312
Herbivorous	0.122	±	0.170	0.481	0.155

The multiple regression model for zooplankton ($R^2 = 0.48$; $p<0.0072$) shows that the variability of the zooplankton biomass is positively related to phytoplankton and negatively related to fish larvae whereas temperature and salinity do not enter significantly into the model. This suggests coupled bottom-up (phytoplankton) and top-down (with fish-larvae as predator index) controls of the zooplankton biomass.

The multiple regression model for phytoplankton ($R^2 = 0.49$; $p<0.028$) suggests bottom-up control of phytoplankton biomass with NH_4^+ contributing significantly ($p=0.018$), whereas top-down control (with herbivorous biomass as a proxy) has no significant effect (Table 2).

Principal component analyses (PCA)

The first factorial plane of PCA on biomass percentages of zooplankton taxonomic and trophic groups explained 65% of the variance of which 37.5% were attributed to the first axis and 27.5% to the second (Fig. 6). The first axis opposes most sampling points of the dry season, characterized by high zooplankton and phytoplankton biomasses and by high percentage of non-copepod holoplankton (others), to the sampling points of the north wind season, characterized by lower plankton biomass and importance of meroplankton and detritivorous organisms.

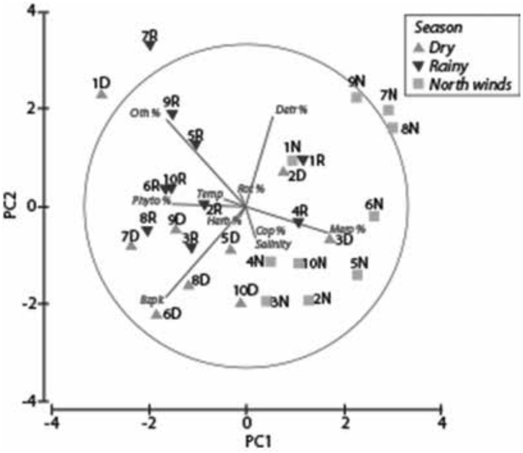

Figure 6. PCA on biomass percentages of zooplankton groups and trophic categories with zooplankton and phytoplankton biomass, temperature and salinity as additional variables: plot of vector variables and stations with color-code according to seasons.

The first factorial plane of PCA on biomass percentages of zooplankton size classes explained 63% of the variance of which 40.1% were attributed to the first axis and 23.0% to the second (Fig. 7). The first axis opposes almost all sampling points of the north wind season, characterized by abundance of smaller size-classes (<300 μm), to sampling points of the two other surveys (Dry and Rainy), characterized by larger sizes and higher total biomass. The second axis mainly shows spatial opposition during the north wind season between stations close to the ocean (stations 7, 8 and 9) characterized by larger size classes (500-600 μm) and stations of the central part of the lagoon (stations 2, 3, 4, 5 and 6) characterized by small organisms (100-200 μm).

Discussion

Low zooplankton biomass in Sontecomapan compared to other coastal lagoons

Simultaneous data on phyto- and zooplankton biomass of coastal lagoons are rather scarce, thus it is difficult to compare our data to that from other sites of the world. However, the zooplankton biomass in Sontecomapan (0.1 to 34 mg C m^{-3}; mean = 3.5 mg C m^{-3}) appears very low compared to literature data. From several studies we could summarize in Table 3, Sontecomapan is the poorest in terms of zooplankton biomass despite the fact that its phytoplankton level is in the range of the other studies. Zooplankton biomass reported in this study is one magnitude lower than the second poorest lagoon in the list (Ebrié lagoon, Ivory Coast; Pagano & Saint Jean, 1994) and more than 100 times lower than those reported in NW Mediterranean lagoons (Lam Hoai & Rougier, 2001; Badosa *et al.*, 2007). Also, it is far poorer than the other Mexican coastal lagoons for which zooplankton biomass were reported, either from the Atlantic coast (Alvarado lagoon; Cruz-Escalona *et al.*, 2007) or the Pacific coast (Huizache-Caimanero Lagoon; Zetina-Rejón *et al.*, 2003).

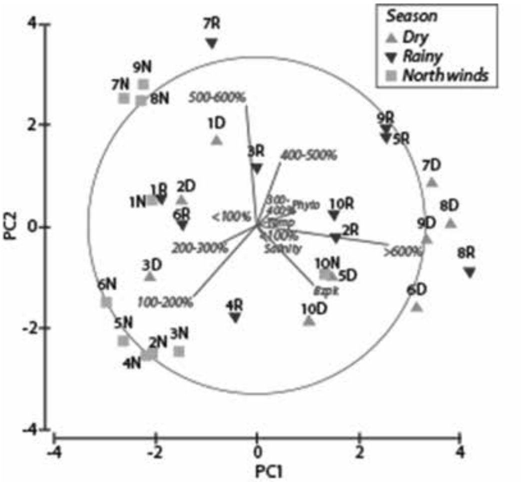

Figure 7. PCA on biomass percentages of zooplankton size classes with zooplankton and phytoplankton biomass, temperature and salinity as additional variables: plot of vector variables and stations with color-code according to seasons.

Another characteristic of Sontecomapan is the very low zooplankton/phytoplankton biomass ratio (1.4%), the lowest reported in Table 3. This low ratio may arise from (i) poor (or inefficient) transfer between the first trophic levels in the pelagos and/or (ii) strong predation pressure by zooplanktophagous organisms. The second hypothesis seems realistic due to high water transparency, favoring visual predation for example from juvenile fish forms (Benítez-Díaz Mirón *et al.*, 2014). Besides, this low biomass ratio is comparable to the one reported by Pagano & Saint Jean (1994) in another Atlantic tropical lagoon (Ebrié lagoon; W Africa) where it was shown that the low zooplankton biomass compared to phytoplankton was mostly due to strong predation pressure by an important mysid (*Rhopalophthalmus africana*) population (Kouassi *et al.*, 2006).

Zooplankton biomass size-structure in Sontecomapan suggests top-down control by fish

The size spectrum is considered as a key indicator of the zooplankton assemblages dynamics (Krupica *et al.*, 2012). It also reflects the status of the zooplankton community in relation with trophic bottom-up and top-down forcing (Hall *et al.*, 1976; Pace, 1986), and has been shown to be a good indicator of the fish community structure (Mills *et al.*, 1987). Any changes in zooplankton size-structure, may thus reflect changes in the ecosystem structure and functioning in relation with environmental context modification, driven by climatic change and/or anthropogenic pressure (Brucet *et al.*, 2010). In Sontecomapan, the biomass size-structure is, in average, dominated by small organisms in the size-class of <200 μm (up to 97% of the total biomass, with mean of 33 ± 5%), including mostly rotifer species. This dominance of small organisms may reflect strong predation pressure from larvae or juvenile fish on larger zooplankton organisms, particularly in the 300-600 μm size range, which includes calanoids like *Acartia tonsa,* and harpacticoid copepods like *Euterpina acutifrons* and *Tisbe* spp. (Table 1). Such top-down control by fish on Sontecomapan zooplankton is also suggested by the results from the multiple linear-regression analysis showing a significant negative relationship between zooplankton biomass and fish larvae. The mullet *Mugil cephalus* well represented in the lagoon may exert a strong predation pressure on these copepods as its larvae eat zooplankton almost exclusively (Eda *et al.*, 1990) and its younger stages are known to prey on copepods near the sediment during the day (Eggold & Motta, 1992), when copepods migrate close to the bottom. Besides, mugilids have been shown to play a key role in shallow lakes by removing sediment (and indirectly nutrients) and large zooplankton, most likely influencing the dominance of smaller forms like rotifers (Oltra *et al.*, 2001; Blanco *et al.*, 2003). Catfish *Cathorops aguadulce*, well represented in Sontecomapan lagoon as well as in other coastal lagoon of the Gulf of Mexico, was also shown to spawn in different habitats of the lagoons using the seagrass/mangrove system as a nursery area. This suggests an important larval density of *Cathorops aguadulce* in Sontecomapan, probably exerting strong predation on zooplankton. High predation may be also exerted by their juvenile and adult stages which showed great percentages of copepods in their diet (87% and 47% of total prey numbers respectively) at Téminos Lagoon, Mexico (Yáñez-Arancibia & Lara-Domínguez, 1988).

In this study, we observed spatial and temporal changes on the zooplankton size-structure in Sontecomapan. In terms of seasonal variations, the dominance of small organisms was more accentuated during the north wind season mainly due to depletion of the largest size-class (>600 μm) compared to the other seasons (Fig. 4). This seasonal variation in zooplankton size-structure may be linked to fish breeding cycles, leading to seasonal variation of the predation pressure from fish larvae on large calanoid copepods such as *Paracalanus aculeatus*, *Temora turbinata* and *Centropages velificatus*, which make the bulk of the largest zooplankton size-class (Table 1). For example, it was shown in Términos Lagoon (Mexico) that *Arius melanopus* carried seasonal migrations for reproduction and feeding with a peak spawning period just before the north wind season (Yáñez-Arancibia & Lara-Domínguez, 1988). Spatially, the dominance of small organisms was permanent in the central and southern part of the lagoon, but disappeared in the channel communicating with the ocean (stations 7, 8 and 9), where the size-distribution was shifted towards large organisms with a significant increase of the largest size-class ($>600\mu$m) as compared with the other stations. This may correspond to variations of zooplankton composition linked to marine water influence, with rather stable salinity conditions in the central and southern part of the lagoon contrasting with tidal influence in the northern part driving inputs of marine coastal species as *Paracalanus aculeatus*, *Temora turbinata* and *Centropages velificatus* in the channel and rotifers collapse due to high salinity pulses.

Trophic controls of phyto- and zooplanktons

If a top-down control on zooplankton by fish predation is suggested by biomass ratio analysis and the size-structure (see discussion above), our regression analysis also shows clearly that the zooplankton biomass is positively related to phytoplankton suggesting a bottom-up control linked to food (phytoplankton) limitation. Such food limitation, despite the rather high chlorophyll level (Table 3) may result from inedibility of a large part of phytoplankton. In Sontecomapan during the study period, high concentration of chlorophyll c with importance of *c1* and *c2* forms (Benítez-Díaz Mirón *et al.*, 2014) suggested large proportion of dinoflagellates and diatoms, from which several species may be toxic to zooplankton (Granéli & Turner, 2006), also, blooms of diatoms (*Skeletonema* spp., *Cyclotella* spp. and *Chaetoceros holasticus*) and dinoflagellates (*Peridinium* spp., *Prorocentrum cordatum*) have been reported (Aké-Castillo & Vázquez, 2008).

The multiple regression model for phytoplankton suggests a bottom-up control of phytoplankton biomass linked to nitrogen limitation (NH_4^+) as also suggested by the overall low N/P nutrient ratio in the lagoon (6.4 ± 1.5), much lower than the Redfield ratio. Aké-Castillo & Vázquez (2008) also pointed out the importance of bottom-up effects on phytoplankton in Sontecomapan with clear seasonal and spatial variations linked to organic matter mineralization introduced by mangrove litter.

Table 3. Mean and range values comparison of phytoplankton and zooplankton biomass with literature data on other coastal lagoons

Name	Location	Phytoplankton (P) mg C m^{-3}			Zooplankton (Z) mg C m^{-3}			Z/P (%)	Reference
		mean	min	max	mean	min	max	mean	
Thau	France (NW Mediterranean)	**150**	100	250	**1000**	205	3550	**667**	**1**
La Pletera	Spain (NW Mediterranean)	**790**	214	2239	**498**	54	1994	**63**	**2**
Ochkeul	Tunisia (SW Mediterranean)	**19**	1	34	**69**			**367**	**3**
Ebrié	Ivory Coast (SE Atlantic)	**2100**	776	3160	**36**	8	80	**2**	**4**
Huizache	Mexico (NW Atlantic)	**885**			**265**			**30**	**5**
Alvarado	Mexico (NE Pacific)	**207**			**184**			**89**	**6**
Sontecomapan	Mexico (NW Atlantic)	**250**	38	674	**3.5**	0.1	34.4	**1.4**	**This study**

References: 1 = Lam Hoai & Rougier (2001), 2 = Badosa *et al.* (2007), 3 = Casagranda & Boudouresque (2010), 4 = Pagano & Saint (1994a)

On the other hand, top-down control by herbivorous organisms on phytoplankton would be non-significant as shown by the regression analysis. Besides, our grazing estimates show that the grazing pressure from herbivorous zooplankton on phytoplankton is rather low, representing only 0.1 to 9.8% of the phytoplankton stock removed daily.

Conclusion

In Sontecomapan, the low zooplankton abundance is related to strong predation by larval and juvenile fish, while the food selectivity of herbivorous zooplankton would lead to a low exploitation of the phytoplankton production, and thus to reduced zooplankton stocks. This should foster the sedimentation of ungrazed dead cells and the accumulation of organic matter at the bottom that could lead to the lagoon's silting up and anoxia events. This tendency also favored by the use of fertilizers and pesticides, and associated eutrophication, would cause a degradation of the ecosystem structure and functioning, however, the high rate of water exchange with the ocean and the resulting low water renewal time in the lagoon should preserve the lagoon from these degrading trends, with the exception of the confined areas that must be surveyed and monitored with priority.

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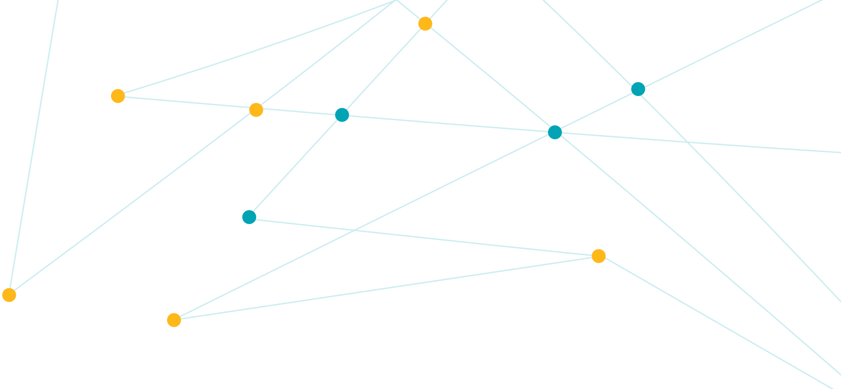

PART 6

Higher trophic levels

Fish species richness in the Sontecomapan Lagoon, Veracruz: A historic review

Manuel Castillo-Rivera & Ana Laura Lara-Domínguez

Fish community structure and its relationship with environmental behavior in Sontecomapan Lagoon, Los Tuxtlas Biosphere Reserve, Veracruz, Mexico

Arturo Aguirre-León, Silvia Díaz-Ruiz & Marissa Guillén Castrillo

Fish species richness in the Sontecomapan Lagoon, Veracruz: A historic review

Manuel Castillo-Rivera* & Ana Laura Lara-Domínguez**

Abstract: The objective of the present study is focused on a comprehensive literature review of Sontecomapan Lagoon's fish fauna, in order to have a better estimate of its species richness. According to previous studies, in the Sontecomapan Lagoon 115 species have been historically recorded, although two of them may correspond to misidentifications (*Cathorops melanopus* by *Cathorops aguadulce,* and *Membras vagrans* by *Membras martinica*). At least for 17 cases the taxonomic status has changed, the most relevant being *Hypanus sabinus*, *Ctenogobius boleosoma*, *Gobionellus oceanicus*, *Mayaheros urophthalmus*, *Pseudoxiphophorus bimaculatus*, and *Microphis lineatus*. The species level is uncertain for two cases, *Tilapia* sp. (in the present study considered as *Oreochromis* sp.) and *Poeciliopsis* sp. In both cases, the species to which these records can correspond is proposed. All species listed correspond to 26 orders, 41 families and 85 genera. Despite having only about 8.9 km^2 of surface, the lagoon is one of the most species-rich estuarine ecosystems in the southern Gulf of Mexico, with approximately 13% of the total fish species recorded for this area. Finally, according to the nonparametric methods for the estimation of species richness (Mao Tau, Chao 2, Jackknife 1, Jackknife 2, and Bootstrap), the total number of species expected for the lagoon could be between 119 and 127 species.

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KEYWORDS: Synonymy, Misspellings, Species composition changes, Species accumulation curves, Caribbean Province

Introduction

Over one-half (32,000 to 33,300 species and counting) of the world's living vertebrates are fish (Nelson *et al.*, 2016; Froese & Pauly, 2018) and from these, about 40% occur in shallow tropical waters (Moyle & Cech, 2004). In recent surveys, a total of 1,541 fish species have been recorded from the Gulf of Mexico (from Florida Keys to northwestern Cuba). This represents 64.3% of the species occurring in the Western Central Atlantic Region (McEachran, 2009). The Gulf of Mexico can be divided in three biogeographical subregions (McEachran, 2009; McEachran & Fechhelm, 2005). The eastern subregion extends from the Florida Keys and western Cuba to Pensacola, Florida, which is the most species-rich subregion, with about 1,300 species. The northwestern subregion extends from Pensacola, Florida, to Cabo Rojo (Tamiahua Lagoon), Veracruz, and is second in species richness, with about 1,100 species. The southern subregion extends from Cabo Rojo, to Cabo Catoche, Yucatan, and is third place with about 900 species.

Also, according to McEachran & Fechhelm (2005) and Briggs & Bowen (2012), two biogeographical provinces of the Warm Western Atlantic Region converge in the Gulf, the Carolina Province with warm-temperate biota, that occupies the area north of the tropical boundaries between Cape Romano, Florida, and Cape Rojo (similar to the limits of the eastern and northwestern subregions), and the Caribbean Province with tropical and subtropical biota, which extends from Cabo Rojo to Cabo Catoche, in Yucatan (same boundaries of the southern subregion, Figure 1).

Many coastal fish species (which can vary between 50-95%) use estuaries permanently or during some period of their lifecycle (Able, 2005), so the study of these ecosystems is vital for knowledge about fish diversity. In the southwestern Gulf of Mexico (Caribbean province), there are about 30 major estuarine systems, of which 19 are located in Veracruz, 4 in Tabasco, 3 in Campeche and 4 in Yucatan (Contreras, 2010). In most of them, fish species compositions have been studied, thus for the state of Veracruz 82 species have been recorded in the Pueblo Viejo lagoon (the northernmost case) (Castillo-Rivera *et al.*, 2002, 2010, 2011), 136 in the Tamiahua lagoon (Franco-López & Chávez-López, 1992; Díaz-Ruiz *et al.*, 2003), 179 in the Tuxpan-Tampamachoco estuarine system (Kobelkowsky, 1985; Pérez-Hernández & Torres-Orozco, 2000), 22 in the Chica-Grande lagoon system (Aguirre-León *et al.*, 2014), 107 in the Camaronera-Alvarado system (Chávez-López *et al.*, 2005) and 36 in the Ostión lagoon (Bozada & Chávez, 1986). For the State of Tabasco, there are records in the Carmen-Machona-Redonda lagoon system with 81 species (Reséndez-Medina & Kobelkowsky, 1991) and at the lower basin of the Grijalva-Usumacinta riverine system with 46 species (Arévalo-Frías & Mendoza-Carranza, 2015). In the State of Campeche, for the Sabancuy estuary, 33 species have been recorded (González-Solís & Torruco, 2013), and at Términos lagoon up to 154, although in recent years it has been noted a reduction in species richness to about 105 species (Ramos-Miranda *et al.*, 2005). Finally, for the State of Yucatan 56 fish species have been reported in the Celestun lagoon (Arceo-Carranza *et al.*, 2010), 81 for the "Bocas de Dzilam" lagoon (Arceo-Carranza & Vega-Cendejas, 2009) and 81 species for the Rio Lagartos (Vega-Cendejas & Hernández, 2004) (Fig. 1).

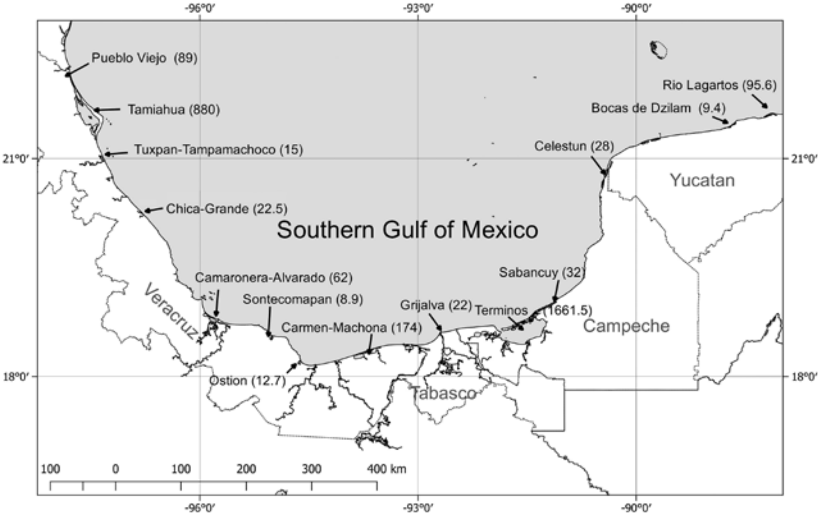

Figure 1. Geographic location of the main estuarine systems in the southern Gulf of México (Caribbean Province). The approximate surface (km^2) of each one is also indicated in parentheses.

Particularly for Sontecomapan Lagoon there are historical reports of its fish fauna, with sampling periods during 1975-1979 (Reséndez-Medina, 1983), 1984-1987 (Fuentes-Mata *et al.*, 1989), 1985-1995 (Fuentes-Mata & Espinosa-Perez, 1997) and more recently during 2005-2006 (Rodriguez-Varela *et al.*, 2010). In addition, there are studies comparing the fish fauna of several estuarine systems in the state of Veracruz which also provide information on this lagoon fish species richness (Lara-Domínguez *et al.*, 2011; Abarca-Arenas *et al.*, 2012).[1]

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[1] Note of Editors: In a very recent study, Aguirre-León *et al.* (this issue) analyze for Sontecomapan, the variations of fish community structure in relation to environmental factors, but this study was not taken into account in the present review.

In all these studies, the species richness reported for the system did not exceed 92. Therefore, the objective of the present study is focused on a comprehensive literature review of the fish fauna of the Sontecomapan Lagoon, in order to have a better estimate of its species richness.

Methodological procedures

The present study was conducted through a comprehensive literature review, mainly based on field studies that were directly carried out in this system (Reséndez-Medina, 1983; Fuentes-Mata *et al.*, 1989; Fuentes-Mata & Espinosa-Pérez, 1997; Rodríguez-Varela *et al.*, 2010), as well as on two comparative checklists (Lara-Domínguez *et al.*, 2011; Abarca-Arenas *et al.*, 2012). The name of every species recorded in each of these studies is presented as originally written (Appendix) and its consistency was checked according to Froese & Pauly (2018), to detect problems of synonyms (each of two or more scientific names of the same rank used to denote the same taxon), probable misidentifications (the incorrect assignment of a specimen to a taxon) and misspellings (spelling incorrectly the scientific name of a species). Based on the above, the total number of species that has been historically reported for the Sontecomapan Lagoon was determined. The systematic sequence of orders and families follows the new fish classification of Nelson *et al.* (2016), which implies major changes in some taxa, principally at order level (in the present case, nine new orders, previously covered by only two).

The Jaccard index (for presence/absence data) was used to assess the similarity degree of species among the different studies (Magurran, 2004). Then, a similarity matrix was generated and subjected to an UPGMA (unweighted pair-group method with arithmetic mean) in order to generate a similarity dendrogram (McGarigal *et al.*, 2000) using MVSp. (Multivariate Statistical Package, version 3.1; Kovach, 2005).

Since the number of species counted in a biodiversity study is usually a biased underestimate of total species richness (because many rare species are not detected), a species accumulation curve was employed to provide an estimate of the total fish richness in the system (Magurran, 2004) using data from each study as samples. For this purpose, the nonparametric estimators of Mao Tau, Chao 2, Jackknife 1, Jackknife 2, and Bootstrap were calculated using EstimateS software (Colwell, 2013).

General description of previous studies in the system

The first formal study on the fish fauna in the system was developed by Reséndez-Medina (1983), and was conducted from 1975 to 1980, with 15 collection sites using a wide range of fishing gear (seine net, dip net, cast net and gill net), as well as a Chem Fish collector. In this study 26 families, 45 genera and 52 species were recorded (Appendix).

Fuentes-Mata *et al.* (1989), with surveys conducted between 1984 and 1987, reported new records in accordance with previous studies, using a beach seine and recording 18 families, 30 genera and 31 species. In a similar survey, between 1985 and 1995, Fuentes-Mata & Espinosa-Pérez (1997), a study of the natural history of Los Tuxtlas, reported 32 families, 60 genera and 83 species (Appendix).

More recently, during 2005-2006, Rodríguez-Varela *et al.* (2010) selected 17 sampling sites using stratified sampling. Fish larvae and juveniles were caught using a Renfro net and adults using a "rowing-boat", with a mesh aperture of ¼ inch. These authors recorded 24 families, 41 genera and 52 species (Appendix).

Finally, Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012) surveys, recorded the fish species composition for many estuarine systems of Veracruz, which include extensive information from theses, symposia and institutional internal reports. Thus, for the Sontecomapan Lagoon Lara-Domínguez *et al.* (2011) recorded 33 families, 65 genera and 91 species, whereas Abarca-Arenas *et al.* (2012) recorded 34 families, 66 genera and 92 species (Appendix).

Taxonomic changes in recorded species

The taxonomic status of some species previously recorded in the Sontecomapan Lagoon have changed to current date. These changes are listed below, following the sequence of orders and families proposed by Nelson *et al.* (2016) and, when necessary, species are sorted alphabetically within each family.

Dasyatis sabina. Reséndez-Medina (1983), Lara-Domínguez *et al.* (2011), and Abarca-Arenas *et al.* (2012) recorded this species for the system, which is a name accepted by many other authors. However, according to recent studies on molecular analyses, combined with comprehensive morphological investigations of the Dasyatidae family (Last *et al.*, 2016a; 2016b), the valid name for this taxon is *Hypanus sabinus*.

Astyanax fasciatus. Fuentes-Mata & Espinosa-Pérez (1997) recorded this species, however Miller *et al.* (2009: 172) note that is no longer acceptable to believe that *Astyanax fasciatus* (restricted to Rio Sao Francisco, Brazil) occurs as far north as Mexico. Based on molecular analyses, morphological characters and patterns of geographic distribution of the genus *Astyanax* species (Strecker *et al.*, 2004; Nelson, 2006; Ornelas-García *et al.*, 2008; Miller *et al.*, 2009; Hausdorf *et al.*, 2011; Schmitter-Soto, 2016.), it is considered in the present study that this record could correspond to *Astyanax aeneus* (Günther, 1860), which is spread south of the Trans-Mexican Volcanic Belt, from Papaloapan river, southward to at least Costa Rica.

Arius aguadulce. Although Reséndez-Medina (1983) recorded this species, it has been widely recognized that this name is a synonym of the valid name *Cathorops aguadulce* (Marceniuk & Menezes, 2007; Froese & Pauly, 2018).

Arius felis. This species is recorded by Reséndez-Medina (1983), but it has been long recognized as a synonym of *Ariopsis felis* (Linnaeus, 1766) (Acero, 2002; Froese & Pauly, 2018) and with this name it is reported in Sontecomapan by Fuentes-Mata & Espinosa-Pérez (1997), Rodríguez-Varela *et al.* (2010), Lara-Domínguez *et al.* (2011), and Abarca-Arenas *et al.* (2012).

Arius melanopus = *Cathorops melanopus*. Fuentes-Mata & Espinosa-Pérez (1997) recorded the *Arius melanopus* species, whereas Rodríguez-Varela *et al.* (2010) and Lara-Domínguez *et al.* (2011) recorded the *Cathorops melanopus* species. In this sense, it has been noted that *Arius* (or *Cathorops*) *melanopus* does not occur in Mexico and their records frequently are misidentifications of *Cathorops aguadulce* (Meek, 1904) (Marceniuk & Betancur-R, 2008; Miller *et al.*, 2009; Froese & Pauly, 2018). *Cathorops aguadulce* is distinguished from all congeners by the fleshy papillae intercalated with gill rakers on the first two gill arches (*vs*. papillae absent in the remaining species, except in *C. kailolae*) (Marceniuk & Betancur-R, 2008; Marceniuk *et al.*, 2012). Moreover, the recognized current distribution for *C. aguadulce* is from Panuco river basin, to Izabal Lake, in Guatemala, whereas *C. melanopus* is restricted to the freshwaters of Motagua river (in Guatemala and probably Honduras), and is commonly recognized as the only species of *Cathorops* described for the entire Caribbean portion of Central America (Ferraris, 2007; Marceniuk & Betancur-R, 2008; Miller *et al.*, 2009). Although there has been an apparent resistance in Mexico to recognize these facts, especially by the ecologists, the species *C. aguadulce* is properly recorded in the Sontecomapan Lagoon by Fuentes-Mata *et al.* (1989), Fuentes-Mata & Espinosa-Pérez (1997), Tenorio-Colín *et al.* (2010), and Abarca-Arenas *et al.* (2012).

Rhamdia quelen. Recorded by Abarca-Arenas *et al.* (2012) as a member of the Pimelodidae family, however, for a long time the Rhamdiinae and Heptapterinae subfamilies have been recognized within the Heptapteridae family (Bockmann & Guazzelli, 2003; Nelson 2006; Nelson *et al.*, 2016).

Gobionellus boleosoma. Recorded by Reséndez-Medina (1983), but based on phylogenetic analysis of osteological and cephalic lateralis characters, Pezold (2004a; 2004b) formally reassigns the genus to *Ctenogobius*. Thus, the current valid name for this species is *Ctenogobius boleosoma* (Jordan & Gilbert, 1882). Despite this, recent records for the Sontecomapan Lagoon still retain the incorrect name of *G. boleosoma* (Rodríguez-Varela *et al.*, 2010; Lara-Domínguez *et al.*, 2011; Abarca-Arenas *et al.*, 2012). Moreover, based on phylogenetic works (Thacker, 2009; Betancur-R *et al.*, 2013; Nelson *et al.*, 2016), this species, along with other recorded in the Sontecomapan Lagoon, as *Evorthodus lyricus*, *Gobioides broussonnetii*, and *Gobionellus oceanicus* (formerly included in Gobionellinae subfamily), are now included within Oxudercidae (= Gobionellidae from some authors), in the Gobiiformes order.

Gobionellus hastatus. Fuentes-Mata *et al.* (1989) recorded this species. However, based on morphological and allozymic analysis, Pezold & Grady (1989) placed *Gobionellus hastatus,* Girard, 1858 and *Gobionellus gracillimus* Ginsburg, 1953, in the synonymy of *Gobionellus oceanicus* (Pallas, 1770) (Pezold, 2004b). In the same way as in the previous case, although *G. oceanicus* has been recognized as valid for a long time, for the Sontecomapan Lagoon, Rodríguez-Varela *et al.* (2010), Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012) persisted in the use of the invalid name of *G. hastatus*.

Cichlasoma fenestratum. Reséndez-Medina (1983) and Fuentes-Mata & Espinosa-Pérez (1997) recorded *Cichlasoma fenestratum* (Günther, 1860), which corresponds to the original description of *Chromis fenestrata* Gunther, 1860. Later, this was recognized as *Vieja fenestrata* (Günther, 1860) by some authors (Kullander, 2003; Miller *et al.*, 2009; Mercado-Silva *et al.*, 2011) and recorded in the system with this name by Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012). Although studies on molecular systematics of the genus *Vieja* (López-Fernández *et al.*, 2010; McMahan *et al.*, 2010) proposed the name *Paraneetroplus fenestratus* for this taxon, recent studies rediagnose it as *Vieja fenestrata* (McMahan *et al.*, 2015). This species, along with other recorded in the Sontecomapan Lagoon, as *Cichlasoma urophthalmum*, *Cichlasoma geddesi*, *Rocio octofasciata*, *Theraps irregularis*, and *Oreochromis* sp. (Cichlidae family), are now included within the order Cichliformes (Nelson *et al.*, 2016).

Cichlasoma urophthalmus. There are numerous problems of species discrimination within the Cichlidae family and some of the most enigmatic cases include '*Cichlasoma*' *urophthalmus*. Indeed, the generic allocation of this species has been uncertain for some time (Kullander, 2003; Concheiro-Pérez *et al.*, 2007; Říčan *et al.*, 2008; López-Fernández *et al.*, 2010). Although this name has been recognized by many authors (Kullander, 2003; Miller *et al.*, 2009; Page *et al.*, 2013) and reported by Rodríguez-Varela *et al.* (2010) for the system, according to recent studies on the Middle American cichlid fishes (Říčan *et al.*, 2016), it has been proposed *that Mayaheros urophthalmus* be the valid name of the species.

Rocio octofasciatus. The species *R. octofasciatus* was recorded by Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012). However, since Schmitter-Soto (2007), the current valid name for this species is *Rocio octofasciata* (Regan, 1903).

Tilapia sp. An unidentified species of the *Tilapia* genus was recorded by Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012), but this record can be very controversial. According to Schmitter-Soto (Miller *et al.*, 2009), there has been a nomenclatural decision to assign species of the genus *Oreochromis* to species of the *Tilapia* genus, which goes against the recent use and both genera can be consistently separated (Sodsuk & McAndrew, 1991). Thus, in this case, the first problem is to know to which genus corresponds this record. Amador-del Ángel & Wakida-Kusunoki (2014) consider that in southwestern Mexico there are only four introduced species from the Cichlidae family (all of African origin), three of them within the *Oreochromis* genus (*O. aureus, O. niloticus*

niloticus, and *O. mossambicus*), and one within the *Tilapia* genus (*T. rendalli*) [According to Froese & Pauly (2018) the current valid name for the latter species is *Coptodon rendalli* (Boulenger, 1897)]. Page *et al.* (2013) recognize in Mexico the same three introduced species for the genus *Oreochromis*, but another for the genus *Tilapia* (*T. zillii*) [Also according to Froese & Pauly (2018) the current valid name for this species is *Coptodon zillii* (Gervais, 1848)]. In the same way, in the checklist of freshwater fish from Veracruz reported by Mercado-Silva *et al.* (2011: Appendix VIII.40), only *O. aureus, O. niloticus niloticus* and *O. mossambicus* are recorded (species of the *Tilapia* genus are not reported). Finally, Fuentes-Mata & Espinosa-Pérez (1997) for Catemaco lake and Escondida lagoon (both very near to Sontecomapan Lagoon) recorded *Oreochromis niloticus*. According to the above, in the present study the record of *Tilapia* sp. is considered as *Oreochromis* sp., which most likely corresponds to *Oreochromis niloticus* (Linnaeus, 1758).

Blennius nicholsi. Reséndez-Medina (1983) recorded the species *Blennius nicholsi*, but for a long time, this has been recognized as a synonym for the current valid name *Lupinoblennius nicholsi* (Tavolga, 1954) (Froese & Pauly, 2018). With this valid name, Fuentes-Mata & Espinosa-Pérez (1997), Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012) recorded this species for the system. The family of this species (Blenniidae) is currently included in the Blenniiformes order (Nelson *et al.*, 2016).

Archomenidia sallei. This species was recorded by Reséndez-Medina (1983) within the *Archomenidia* genus, but currently this species is recognized within the *Atherinella* genus (Chernoff, 1986; Miller *et al.*, 2009), and its valid name is *Atherinella sallei* (Regan, 1903). Reséndez-Medina (1983) also assigns this species to the Atherinidae family. However, the New World silversides that had formerly been recognized as a subfamily of Atherinidae, were formally removed from this family and were reassigned in Atherinopsidae (Dyer & Chernoff, 1996; Dyer, 2006; Nelson *et al.*, 2016). Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012) recorded this species for the system with the correct family and species names.

Membras vagrans. Rodríguez-Varela *et al.* (2010) recorded this species, but it can also be controversial. Despite that it is a valid name (Castro-Aguirre & Espinosa-Pérez, 2006; Castro-Aguirre *et al.*, 1999; Froese & Pauly, 2018), according to Hoese & Moore (1998), reports of *Membras vagrans* (Goode & Bean, 1879) refer to *Membras martinica* (Valenciennes, 1835). *Membras vagrans* is generally found in the deeper; more saline portions of the bays and out into the Gulf as far as 15 miles offshore. Similarly, Chernoff (1986) consider this species as *M. martinica*. In this way, in many studies, *M. martinica* is recorded in the Gulf of Mexico, but not *M. vagrans* (Robins & Ray, 1986; Hoese & Moore, 1998; Chernoff, 2002; McEachran & Fechhelm, 2005; McEachran, 2009; Mercado-Silva *et al.*, 2011; Page *et al.*, 2013). On the other hand, in the lists of fish species in some coastal lagoons of Veracruz, as Tamiahua (Franco-López & Chávez-López, 1992; Díaz-Ruiz *et al.*, 2003), Mandinga (Lara-Domínguez *et al.*, 2011), and Camaronera-Alvarado (Chávez-

López *et al.*, 2005), *M. vagrans* is recorded, but not *M. martinica*. This probably involves an identification problem, mainly related to the number of anal soft rays, which according to Castro-Aguirre *et al.* (1999) and Castro-Aguirre & Espinosa-Pérez (2006), it is the main characteristic to differentiate both species. However, meristic and morphometric variation within a single species may be associated with environmental variation, and differences between size classes (Chernoff, 1982), in part, the problem is also that Rodriguez-Varela *et al.* (2010) recorded both species. Indeed, this seems to be the only report from southwestern Gulf of Mexico (Caribbean province) which records both species in sympatry. Anyway, in the present study it was decided to recognize the record of *M. vagrans* for the system, but it is clear that further analysis is necessary not only to determine the presence of *M. vagrans* in this system, but also to validate all records reported for coastal systems in the southwestern Gulf of Mexico.

Hyporhamphus roberti. This species is recorded by Fuentes-Mata *et al.* (1989), which is a name accepted by other authors (Cervigón *et al.*, 1992; Collette, 2002; Page *et al.*, 2013). Within this species, two subspecies are recognized (Collette, 2002), *Hyporhamphus roberti hildebrandi* Jordan & Evermann, 1927 and *Hyporhamphus roberti roberti* (Valenciennes, 1847). The latter taxon is recognized as a valid name by Froese & Pauly (2018) and with this name, the species is recorded for the system by Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012).

Strongylura notata. Fuentes-Mata *et al.* (1989), Fuentes-Mata & Espinosa-Pérez (1997), Rodríguez-Varela *et al.* (2010), Lara-Domínguez *et al.* (2011), and Abarca-Arenas *et al.* (2012) recorded this species for the system which is a name accepted by other authors (Robins & Ray, 1986; Castro-Aguirre *et al.*, 1999; Collette, 2002; McEachran, 2009; Page *et al.*, 2013). As with the previous species, two subspecies are recognized (Collette, 2002): *Strongylura notata forsythia* Breder, 1932 and *Strongylura notata notata* (Poey, 1860). The latter is recognized as the valid name by Froese & Pauly (2018).

Tylosurus crocodilus crocodilus. Fuentes-Mata & Espinosa-Pérez (1997) recorded this species and the name is accepted by other authors (Cervigón *et al.*, 1992; Collette, 2002). However, Froese & Pauly (2018) consider that this name is a synonym of *Tylosurus crocodilus* (Péron & Lesueur, 1821), which is also used by other authors (Robins & Ray, 1986; Castro-Aguirre *et al.*, 1999; McEachran, 2009; Page *et al.*, 2013), including the original record for the system by Fuentes-Mata *et al.* (1989).

Heterandria bimaculata. Reséndez-Medina (1983), Fuentes-Mata & Espinosa-Pérez (1997), Lara-Domínguez *et al.* (2011), and Abarca-Arenas *et al.* (2012) recorded this species for the system, however, based on available phylogenetic information (Morales-Cazan & Albert, 2012), species of "*Heterandria*" were reassigned to the genus *Pseudoxiphophorus*, and the valid name proposed for this species is *Pseudoxiphophorus bimaculatus*.

Poeciliopsis sp. An unidentified species of the *Poeciliopsis* genus was recorded by Lara-Domínguez *et al.* (2011) & Abarca-Arenas *et al.* (2012). According to Miller (1975), Miller *et al.* (2009), and Mercado-Silva *et al.* (2011), this record may correspond to two species, *Poeciliopsis catemaco* Miller, 1975, with distribution range in the Atlantic slope, in Catemaco lake and its outlet near the lake, the Papalopan river basin, Veracruz; and *Poeciliopsis gracilis* (Heckel, 1848), with a distribution also in the Atlantic slope, from a 20 km stream, north of Cardel city, southward into the Coatzacoalcos river basin, Veracruz (Miller *et al.*, 2009). Given this uncertainty, in the present study, this record is considered as was originally proposed.

Oostethus brachyurus lineatus = *Oostethus (B.) lineatus* = *Microphis brachyurus lineatus*. This species is recorded as *Oostethus brachyurus lineatus* by Reséndez-Medina (1983) and as *Oostethus* (*B.*) *lineatus* by Fuentes-Mata & Espinosa-Pérez (1997). However, Dawson (1984) proposed *Oostethus* as a subgenus of *Microphis*, and it is currently considered as the genus accepted for this species. Meanwhile, Rodríguez-Varela *et al.* (2010), Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012) recorded *Microphis brachyurus lineatus* for the system. But, according to Froese & Pauly (2018), all of these names are synonyms of the valid name *Microphis lineatus* (Kaup, 1856); although this species is still referred in recent literature as *Oostethus lineatus*, *Oostethus brachyurus lineatus* or *Microphis brachyurus lineatus* (Fritzsche & Vincent, 2002; Ferraris, 2003).

Eucinostomus lefroyi. In comparative studies of several estuarine systems from Veracruz, Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012) recorded this species for the system. Although the name is accepted by many authors (Robins & Ray, 1986; Hoese & Moore, 1998; Gilmore & Greenfield, 2002; McEachran & Fechhelm, 2005; McEachran, 2009; Page *et al.*, 2013), the genus of this species has not been fully defined (Castro-Aguirre *et al.*, 1999). According to Böhlke & Chaplin (1993) this species should be recognized within the genus *Ulaema*, and Froese & Pauly (2018) consider that *Eucinostomus lefroyi* is synonym of *Ulaema lefroyi* (Goode, 1874). It is with this name that is recognized in the present study, which was also registered in the system by Fuentes-Mata *et al.* (1989), Fuentes-Mata & Espinosa-Pérez (1997), and Rodríguez-Varela *et al.* (2010).

Aluterus schoepfi. This species is recorded by Fuentes-Mata *et al.* (1989) within the Balistidae family, but it is currently included within the Monacanthidae family. The separation of Monacanthidae as a family distinct from Balistidae (formerly as a subfamily of the latter), has been long recognized (Berry & Vogele, 1961). Moreover, the original report has a misspelling problem, the correct name for this species is *Aluterus schoepfii*.

List of species historically recorded in the Sontecomapan Lagoon

According to previous analyzes, the total number of species historically recorded for the Sontecomapan Lagoon is 115, although two of them could correspond to misidentifications (*Cathorops melanopus* for *Cathorops aguadulce*, and *Membras vagrans* for *Membras marti-*

nica). All species listed correspond to 26 orders, 41 families and 85 genera. Of these 115 species, 37 were reported in five studies, 30 in four, 14 in only three, 20 were exclusive for two, and 14 were recorded in only one study (Table 1). From these 14 species, six (*C. urophthalmus*, *M. vagrans*, *M. beryllina*, *S. guachancho*, *L. rhomboides*, and *S. testudineus*) were recorded only by Rodríguez-Varela *et al.* (2010), five (*A. fasciatus*, *A. alvarezi*, *S. plumieri*, *C. arenarius*, and *B. capriscus*) were recorded exclusively by Fuentes-Mata & Espinosa-Pérez (1997), two (*O. gomesii* and *A. schoepfii*) were recorded only by Fuentes-Mata *et al.* (1989), and only one (*R. quelen*) was recorded exclusively by Abarca-Arenas *et al.* (2012). All species recorded by Reséndez-Medina (1983) and Lara-Domínguez *et al.* (2011) were considered in at least one of the remaining four studies.

Table 1. List of species historically recorded for the Sontecomapan Lagoon according to studies by Reséndez-Medina (1983), Fuentes-Mata *et al.* (1989), Fuentes-Mata & Espinosa-Pérez (1997), Rodríguez-Varela *et al.* (2010), Lara *et al.* (2011) and Abarca-Arenas *et al.* (2012). The sequence of orders and families follows Nelson *et al.* (2016) and the species are sorted alphabetically within each family. All fish names according to Froese & Pauly (2018), indicating the number of studies (1–5) in which each species was recorded. New orders (a), new families (b) and probable misidentifications (c) are also indicated.

Order/Family	Species	Order/Family	Species
1 Myliobatiformes		**16 Carangiformes**[a]	
Dasyatidae	*Hypanus sabinus* (Lesueur, 1824)^{3}	Carangidae	*Carangoides bartholomaei* (Cuvier, 1833)^{2}
2 Elopiformes			*Caranx crysos* (Mitchill, 1815)^{2}
Elopidae	*Elops saurus* Linnaeus, 1766^{5}		*Caranx hippos* (Linnaeus, 1766)^{4}
Megalopidae	*Megalops atlanticus* Valenciennes, 1847^{3}		*Caranx latus* Agassiz, 1831^{5}
3 Albuliformes			*Chloroscombrus chrysurus* (Linnaeus, 1766)^{3}
Albulidae	*Albula vulpes* (Linnaeus, 1758)^{2}		*Hemicaranx amblyrhynchus* (Cuvier, 1833)^{4}
4 Anguilliformes			*Oligoplites saurus* (Bloch & Schneider, 1801)^{5}
Ophichthidae	*Myrophis punctatus* Lütken, 1852^{3}		*Selene setapinnis* (Mitchill, 1815)^{5}
	Ophichthus gomesii (Castelnau, 1855)^{1}		*Selene vomer* (Linnaeus, 1758)^{4}
5 Clupeiformes			*Trachinotus falcatus* (Linnaeus, 1758)^{2}
Engraulidae	*Anchoa hepsetus* (Linnaeus, 1758)^{5}	**17 Istiophoriformes**[a]	
	Anchoa mitchilli (Valenciennes, 1848)^{5}	Sphyraenidae	*Sphyraena barracuda* (Edwards, 1771)^{5}
	Cetengraulis edentulus (Cuvier, 1829)^{4}		*Sphyraena guachancho* Cuvier, 1829^{1}
Clupeidae	*Brevoortia gunteri* Hildebrand, 1948^{4}	**18 Pleuronectiformes**	
	Harengula jaguana Poey, 1865^{4}	Paralichthyidae	*Citharichthys abbotti* Dawson, 1969^{4}
	Opisthonema oglinum (Lesueur, 1818)^{4}		*Citharichthys macrops* Dresel, 1885^{3}
6 Characiformes			*Citharichthys spilopterus* Günther, 1862^{5}
Characidae	*Astyanax aeneus* (Günther, 1860)^{1}		*Cyclopsetta fimbriata* (Goode & Bean, 1885)^{3}

Table 1. (Continuation)

Order/Family	Species	Order/Family	Species
7 Siluriformes		Achiridae	*Achirus lineatus* (Linnaeus, 1758)[5]
Ariidae	*Ariopsis felis* (Linnaeus, 1766)[5]		*Gymnachirus texae* (Gunter, 1936)[3]
	Bagre marinus (Mitchill, 1815)[4]	Cynoglossidae	*Symphurus plagiusa* (Linnaeus, 1766)[2]
	Cathorops aguadulce (Meek, 1904)[4]	**19 Syngnathiformes**[a]	
	Cathorops melanopus (Günther, 1864)[3,c]	Syngnathidae	*Microphis lineatus* (Kaup, 1856)[5]
Heptapteridae	*Rhamdia quelen* (Quoy & Gaimard, 1824)[1]		*Syngnathus floridae* (Jordan & Gilbert, 1882)[2]
8 Batrachoidiformes			*Syngnathus louisianae* Günther, 1870[2]
Batrachoididae	*Opsanus beta* (Goode & Bean, 1880)[3]		*Syngnathus scovelli* (Evermann & Kendall, 1896)[5]
9 Gobiiformes[a]		**20 Scombriformes**[a]	
Eleotridae	*Dormitator maculatus* (Bloch, 1792)[5]	Trichiuridae	*Trichiurus lepturus* Linnaeus, 1758[4]
	Eleotris pisonis (Gmelin, 1789)[2]	**21 Perciformes**	
	Gobiomorus dormitor Lacepède, 1800[5]	Centropomidae	*Centropomus ensiferus* Poey, 1860[3]
	Guavina guavina (Valenciennes, 1837)[2]		*Centropomus parallelus* Poey, 1860[5]
Oxudercidae[b]	*Ctenogobius boleosoma* (Jordan & Gilbert, 1882)[4]		*Centropomus pectinatus* Poey, 1860[5]
(= Gobionellidae)	*Evorthodus lyricus* (Girard, 1858)[5]		*Centropomus poeyi* Chávez, 1961[4]
	Gobioides broussonnetii Lacepède, 1800[5]		*Centropomus undecimalis* (Bloch, 1792)[5]
	Gobionellus oceanicus (Pallas, 1770)[4]	Gerreidae	*Diapterus auratus* Ranzani, 1842[5]
Gobiidae	*Bathygobius soporator* (Valenciennes, 1837)[5]		*Diapterus rhombeus* (Cuvier, 1829)[5]
10 Mugiliformes			*Eucinostomus argenteus* Baird & Girard, 1855[4]
Mugilidae	*Mugil cephalus* Linnaeus, 1758[4]		*Eucinostomus gula* (Quoy & Gaimard, 1824)[4]
	Mugil curema Valenciennes, 1836[5]		*Eucinostomus melanopterus* (Bleeker, 1863)[5]
	Mugil trichodon Poey, 1875[3]		*Eugerres plumieri* (Cuvier, 1830)[5]
11 Cichliformes[a]			*Gerres cinereus* (Walbaum, 1792)[4]
Cichlidae	*Cichlasoma geddesi* (Regan, 1905)[2]		*Ulaema lefroyi* (Goode, 1874)[5]
	Mayaheros urophthalmus (Günther, 1862)[1]	Polynemidae	*Polydactylus octonemus* (Girard, 1858)[4]
	Oreochromis sp[2]	Serranidae	*Epinephelus adscensionis* (Osbeck, 1765)[5]
	Rocio octofasciata (Regan, 1903)[2]	Haemulidae	*Conodon nobilis* (Linnaeus, 1758)[2]
	Theraps irregularis Günther, 1862[2]		*Pomadasys crocro* (Cuvier, 1830)[5]
	Vieja fenestrata (Günther, 1860)[4]	Lutjanidae	*Lutjanus apodus* (Walbaum, 1792)[4]
12 Blenniiformes[a]			*Lutjanus griseus* (Linnaeus, 1758)[5]
Blenniidae	*Lupinoblennius nicholsi* (Tavolga, 1954)[4]		*Lutjanus jocu* (Bloch & Schneider, 1801)[5]
13 Atheriniformes		**22 Scorpaeniformes**	
Atherinopsidae	*Atherinella alvarezi* (Díaz-Pardo, 1972)[1]	Scorpaenidae	*Scorpaena plumieri* Bloch, 1789[1]
	Atherinella sallei (Regan, 1903)[3]	**23 Acanthuriformes**[a]	
	Membras martinica (Valenciennes, 1835)[4]	Sciaenidae	*Bairdiella chrysoura* (Lacepède, 1802)[5]
	Membras vagrans (Goode & Bean, 1879)[1,c]		*Bairdiella ronchus* (Cuvier, 1830)[5]
	Menidia beryllina (Cope, 1867)[1]		*Cynoscion arenarius* Ginsburg, 1930[1]

Table 1. (Continuation)

Order/Family	Species	Order/Family	Species
14 Beloniformes			*Cynoscion nebulosus* (Cuvier, 1830)[2]
Hemiramphidae	*Hemiramphus brasiliensis* (Linnaeus, 1758)[4]		*Micropogonias undulatus* (Linnaeus, 1766)[4]
	Hyporhamphus roberti roberti (Valenciennes, 1847)[3]		*Umbrina coroides* Cuvier, 1830[2]
	Hyporhamphus unifasciatus (Ranzani, 1841)[4]	**24 Spariformes**[a]	
Belonidae	*Strongylura marina* (Walbaum, 1792)[5]	Sparidae	*Archosargus probatocephalus* (Walbaum, 1792)[5]
	Strongylura notata notata (Poey, 1860)[5]		*Archosargus rhomboidalis* (Linnaeus, 1758)[4]
	Tylosurus crocodilus (Péron & Lesueur, 1821)[2]		*Lagodon rhomboides* (Linnaeus, 1766)[1]
15 Cyprinodontiformes		**25 Lophiiformes**	
Poeciliidae	*Belonesox belizanus* Kner, 1860[4]	Antennariidae	*Antennarius striatus* (Shaw, 1794)[2]
	Gambusia sexradiata Hubbs, 1936[3]	**26 Tetraodontiformes**	
	Poecilia catemaconis Miller, 1975[2]	Balistidae	*Balistes capriscus* Gmelin, 1789[1]
	Poecilia mexicana Steindachner, 1863[5]	Monacanthidae	*Aluterus schoepfii* (Walbaum, 1792)[1]
	Poecilia sphenops Valenciennes, 1846[5]	Tetraodontidae	*Sphoeroides testudineus* (Linnaeus, 1758)[1]
	Poeciliopsis sp[2]		
	Pseudoxiphophorus bimaculatus (Heckel, 1848)[4]		
	Xiphophorus hellerii Heckel, 1848[4]		

Classification changes at higher levels

In the present study, systematic sequence of orders and families of the new fish classification of *Fishes of the World* (Nelson *et al.* 2016) was adopted, which differs in many respects from the previous editions. For the present case, at family level, the main change was the recognition of Oxudercidae (= Gobionellidae from some authors), based on phylogenetic studies (Thacker, 2009; Nelson et al, 2016), which includes the members of several former subfamilies of the Gobiidae. At order level, Gobiiformes were formerly treated as suborder Gobioidei within the Perciformes order (Nelson, 2006), but according to molecular evidence and some morphological characters (Betancur-R. *et al.*, 2013; Nelson *et al.*, 2016), it is now recognized as a new order, including Eleotridae, Oxudercidae and Gobiidae families. The Cichlidae family (formerly in Labroidei suborder, Perciformes order) is currently in Cichliformes. The family Blenniidae, formerly treated in the Blennioidei suborder (Perciformes order), is placed in Blenniiformes. The Sphyraenidae and Trichiuridae families had been placed in the Scombroidei suborder (Perciformes order) by Nelson (2006), but are now placed in Istiophoriformes and Scombriformes, respectively. The seven families formerly treated as Syngnathoidei suborder (included Syngnathidae) within the Gasterosteiformes order by Nelson (2006) are now placed in the Syngnathiformes order. The Carangidae, Sciaenidae, and Sparidae families were formerly treated as Percoidei suborder within the Perciformes order, and now placed in the new Carangiformes, Acanthuriformes and Spariformes orders respectively. Finally, the phylogenetic position of some orders like Pleuronectiformes and Lophiiformes have drastically changed.

Species similarity among studies

According to the analysis of similarity between the species' composition of each study (Table 2), it can be observed that similarity values vary widely from 0.012 to 0.968. In part, this variability may be due to the different fishing gear used, as well as to changes in species composition, or at least, to changes abundance. In addition, this variability can be influenced by the characteristics of each report. Thus, the lower similarities (≤0.295) are among Fuentes-Mata *et al.* (1989) and the remaining five studies (Figure 2). This is due to two reasons: the first one is that Fuentes-Mata *et al.* (1989) did not list all species caught, reporting only those considered as new records (31 in total); the second, is that comparative studies of Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012) ignored the Fuentes-Mata report. In this sense, it is particularly interesting that studies of Fuentes-Mata *et al.* (1989) and Fuentes-Mata & Espinosa-Pérez (1997) show a low similarity (0.295), with five species (*O. gomesii*, *E. pisonis*, G. hastatus [= *G. oceanicus*], H. *roberti*, and *A. schoepfii*) recorded in the first study, but not in the second, although both sampling periods concurred during 1985-1987.

Table 2. Species similarity (above the diagonal) and shared species (below the diagonal) among the different lists of species reported by: Reséndez-Medina (1983), Fuentes-Mata *et al.* (1989), Fuentes-Mata & Espinosa-Pérez (1997), Rodríguez-Varela *et al.* (2010), Lara-Domínguez *et al.* (2011), and Abarca-Arenas *et al.* (2012).

	1983	1989	1997	2010	2011	2012
1983		0.012	0.500	0.351	0.554	0.565
1989	1		0.295	0.203	0.196	0.206
1997	45	26		0.421	0.657	0.651
2010	27	14	40		0.444	0.426
2011	51	20	69	44		0.968
2012	52	21	69	43	90	

The species composition reported by Rodríguez-Varela *et al.* (2010) also tends to show low similarities with the other studies (Fig. 2), with a particularly low similarity (0.351) between this study and that of Reséndez-Medina (1983). Although both studies recorded a total of 52 species, these only shared 27. In part, this may be because the sampling periods have the greatest separation in years (1980-2005), which can mean drastic changes in the composition or relative abundance of species in 25 years.

In contrast, the checklists of Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012) tend to show higher similarities. This is mainly because they are comparative studies, which take into account previous studies on the lagoon.

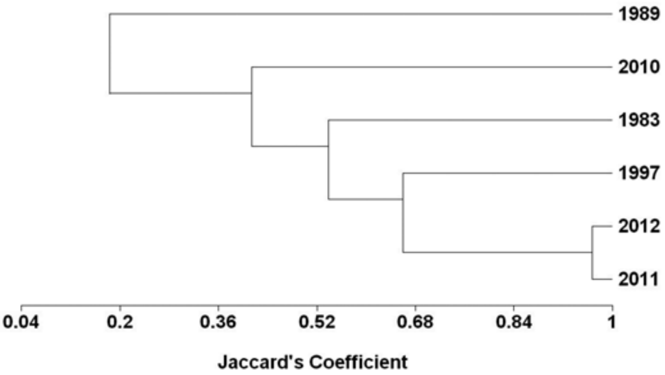

Figure 2. Similarity dendrogram (Jaccard, UPGMA) from the species lists reported by Reséndez-Medina (1983), Fuentes-Mata *et al.* (1989), Fuentes-Mata & Espinosa-Pérez (1997), Rodríguez-Varela et al. (2010), Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012).

Expected number of fish species for the Sontecomapan Lagoon

The number of species reported in the studies carried out in situ varies from 31 to 83, with an accumulated of 103 species recorded, from which at least, 38 were common in all four studies (Reséndez-Medina, 1983; Fuentes-Mata *et al.*, 1989; Fuentes-Mata & Espinosa-Pérez, 1997; Rodríguez-Varela *et al.*, 2010). The two comparative checklists report between 91 and 92 species, from which 90 are common (Lara-Domínguez *et al.*, 2011; Abarca-Arenas *et al.*, 2012). However, the number of species counted in these studies may be an underestimation of the total species richness (due to fishing gear limitations and the fact that many rare species are not detected).

Thus, considering the total of species recorded in approximately 25 years, the total number of expected species according to each of the nonparametric estimators used was: Mao Tau = 115, Chao 2 = 118.61, Jackknife 1 = 126.67, Jackknife 2 = 125.33, and Bootstrap = 121.71. The species accumulation curves for the six studies considered using these nonparametric estimators are shown in Figure 3.

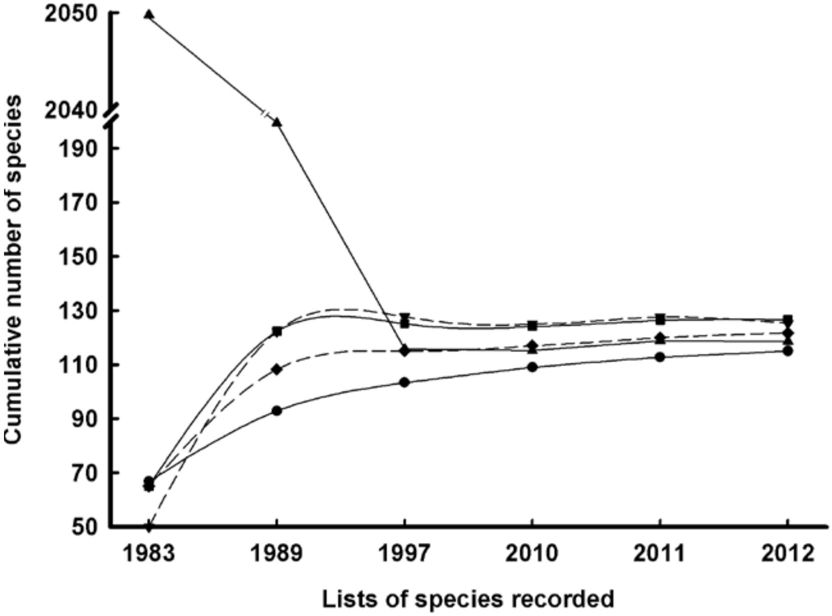

Figure 3. Species accumulation curves considering the species lists (as samples) reported by Reséndez-Medina (1983), Fuentes-Mata et al. (1989), Fuentes-Mata & Espinosa-Pérez (1997), Rodríguez-Varela et al. (2010), Lara-Domínguez et al. (2011), and Abarca-Arenas et al. (2012). Nonparametric estimators used were: Mao Tau (—•—), Chao 2 (—▲—), Jackknife 1 (—■—), Jackknife 2 (–▼–), and Bootstrap (–♦–).

The main ecological interpretation of these results is that the estimates of nonparametric methods are not very different from the total number of species recorded for the system (maximum difference of roughly 12 species) thus, implying that this historic review results in a good estimate of the richness of fish species for this system. Largely, this is because different studies in the system have covered a wide range of developmental stages, including larvae and juveniles (e.g. Rodríguez-Varela *et al.* 2010), as well as a wide range of fishing methods (e.g. Reséndez-Medina 1983).

Concluding remarks

Considering the previous information for the Sontecomapan Lagoon, 115 species have been recorded, two of which may correspond to misidentifications (*Cathorops melanopus* and *Membras vagrans*). At least for 17 cases the taxonomic status has changed, with

Hypanus sabinus (recorded as *Dasyatis sabina*), *Ctenogobius boleosoma* (recorded as *Gobionellus boleosoma*), *Gobionellus oceanicus* (recorded as *Gobionellus hastatus*), *Mayaheros urophthalmus* (recorded as *Cichlasoma urophthalmus*), *Pseudoxiphophorus bimaculatus* (recorded as *Heterandria bimaculata*) and *Microphis lineatus* (recorded as *Oostethus brachyurus lineatus, Oostethus (B.) lineatus and Microphis brachyurus lineatus*) being the most relevant. Although the genus level is known, the species level is uncertain for two cases, *Tilapia* sp. (in the present study considered as *Oreochromis* sp.) and *Poeciliopsis* sp. For both, the species to which these taxa can correspond is proposed. The generic allocation of one species (*Eucinostomus lefroyi* or *Ulaema lefroyi*) seems to be a case in which taxonomists did not end up agreeing. Considering the new fish classification by Nelson *et al.* (2016), the *Ctenogobius boleosoma, Evorthodus lyricus, Gobioides broussonnetii* and *Gobionellus oceanicus* species are now included in the new family Oxudercidae (= Gobionellidae), and 11 of the 41 recorded families are now included in nine new orders.

Despite having an approximately 8.9 km^2 surface, the lagoon is one of the most species-rich estuarine ecosystems in the southern Gulf of Mexico (Caribbean province), with approximately 13% of the total fish species recorded in this area. In relation to the similarity analysis, fish composition showed lower values (<0.6) among most studies in the Sontecomapan Lagoon, even in those studies involving the same authors and similar collection dates. Finally, according to nonparametric methods for species richness estimation, the total number of species expected for the lagoon could be between 119 and 127 species.

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Appendix

Total list of historically recorded taxa for the Sontecomapan Lagoon, as originally reported by Reséndez-Medina (1983), Fuentes-Mata *et al.* (1989), Fuentes-Mata & Espinosa-Pérez (1997), Rodríguez-Varela *et al.* (2010), Lara-Domínguez *et al.* (2011) and Abarca-Arenas *et al.* (2012). Taxa follows the sequence of families proposed by Nelson *et al.* (2016). According to Froese & Pauly (2018), changes in taxonomic status (1), misspellings (2) and probable misidentifications (3) are also indicated.

Especies	1983	1989	1997	2010	2011	2012	Especies	1983	1989	1997	2010	2011	2012
Dasyatis sabina[1]	x				x	x	*Xiphophorus hellerii*	x		x		x	x
Elops saurus	x		x	x	x	x	*Carangoides bartholomaei*					x	x
Megalops atlanticus	x				x	x	*Caranx crysos*		x	x			
Albula vulpes					x	x	*Caranx hippos*	x		x		x	x
Myrophis punctatus	x				x	x	*Caranx latus*	x		x	x	x	x
Ophichthus gomesii		x					*Chloroscombrus chrysurus*					x	x
Anchoa hepsetus		x	x	x	x	x	*Chloroscombrus chysurus*[2]				x		
Anchoa mitchilli	x		x	x	x	x	*Hemicaranx amblyrhynchus*		x	x		x	x
Cetengraulis edentulus	x		x		x	x	*Oligoplites saurus*	x		x	x	x	x
Brevoortia gunteri					x	x	*Selene setapinnis*		x	x	x	x	x
Brevoortia guntheri[2]		x	x				*Selene vomer*	x		x		x	x
Harengula jaguana	x		x		x	x	*Trachinotus falcatus*		x	x			

Continuation

Especies	1983	1989	1997	2010	2011	2012	Especies	1983	1989	1997	2010	2011	2012
Opisthonema oglinum	x		x		x	x	*Sphyraena barracuda*		x	x	x	x	x
Astyanax fasciatus[1]			x				*Sphyraena guachancho*				x		
Ariopsis felis			x	x	x	x	*Citharichthys abbotti*	x				x	x
Arius felis[1]	x						*Citharichthys abboti*[2]			x			
Cathorops aguadulce		x	x			x	*Citharichthys macrops*			x		x	x
Arius aguadulce[1]	x						*Citharichthys spilopterus*	x		x	x	x	x
Cathorops melanopus[3]				x	x		*Cyclopsetta fimbriata*					x	x
Arius melanopus[1]			x				*Cyclopssetta fimbriata*[2]			x			
Bagre marinus	x		x		x	x	*Achirus lineatus*		x	x	x	x	x
Rhamdia quelen						x	*Gymnachirus texae*			x		x	x
Opsanus beta				x	x	x	*Symphurus plagiusa*		x	x			
Dormitator maculatus	x		x	x	x	x	*Microphis brachyurus lineatus*[1]				x	x	x
Eleotris pisonis		x		x			*Oostethus brachyurus lineatus*[1]	x					
Gobiomorus dormitor	x		x	x	x	x	*Oostethus (B.) lineatus*[1]			x			
Gobionellus boleosoma[1]	x			x	x	x	*Syngnathus floridae*					x	x
Guavina guavina		x	x				*Syngnathus louisianae*					x	x
Evorthodus lyricus	x		x	x	x	x	*Syngnathus scovelli*	x		x	x	x	x
Gobioides broussonnetii		x	x				*Trichiurus lepturus*	x		x		x	x
Gobioides broussonneti[2]				x	x	x	*Centropomus ensiferus*			x		x	x
Gobionellus hastatus[1]		x		x	x	x	*Centropomus parallelus*	x		x	x	x	x
Bathygobius soporator			x	x	x	x	*Centroposus pectinatus*		x	x	x	x	x
Bathygobius soporator soporator[1]	x						*Centropomus poeyi*	x		x		x	x
Mugil cephalus	x		x		x	x	*Centropomus undecimalis*	x		x	x	x	x
Mugil curema	x		x	x	x	x	*Diapterus auratus*	x		x	x	x	x
Mugil trichodon			x		x	x	*Diapterus rhombeus*	x		x	x	x	x
Cichlasoma geddesi					x	x	*Eucinostomus argenteus*		x	x		x	x
Cichlasoma urophthalmus[1]				x			*Eucinostomus gula*		x	x		x	x
Cichlasoma fenestratum[1]	x		x				*Eucinostomus melanopterus*	x		x	x	x	x
Vieja fenestrata					x	x	*Eugerres plumieri*	x		x	x	x	x
Rocio octofasciatus[2]					x	x	*Gerres cinereus*		x	x		x	x
Theraps irregularis					x	x	*Ulaema lefroyi*		x	x	x		

Continuation

Especies	1983	1989	1997	2010	2011	2012	Especies	1983	1989	1997	2010	2011	2012
Tilapia sp[3]					x	x	*Eucinostomus lefroyi*[1]					x	x
Lupinoblennius nicholsi			x		x	x	*Epinephelus adscensionis*		x	x	x	x	x
Blennius nicholsi[1]	x						*Conodon nobilis*			x	x		
Atherinella alvarezi			x				*Pomadasys crocro*	x		x	x	x	x
Atherinella sallei					x	x	*Polydactylus octonemus*	x		x		x	x
Archomenidia sallei[1]	x						*Lutjanus apodus*	x		x		x	x
Membras martinica			x	x	x	x	*Lutjanus griseus*	x		x	x	x	x
Membras vagrans[3]				x			*Lutjanus jocu*		x	x	x	x	x
Menidia beryllina				x			*Scorpaena plumieri*			x			
Hemiramphus brasiliensis		x	x		x	x	*Bairdiella chrysoura*				x	x	x
Hyporhamphus roberti roberti					x	x	*Bairdiella chrysura*[2]		x				
Hyporhamphus roberti[1]		x					*Bairdiella chrysora*[2]			x			
Hyporhamphus unifasciatus	x		x		x	x	*Bairdiella ronchus*	x		x	x	x	x
Strongylura marina	x		x	x	x	x	*Cynoscion arenarius*			x			
Strongylura notata[1]		x	x	x	x	x	*Cynoscion nebulosus*		x	x			
Tylosurus crocodilus		x					*Micropogonias undulatus*	x		x		x	x
Tylosurus crocodilus crocodilus[1]			x				*Umbrina coroides*		x	x			
Belonesox belizanus	x			x	x	x	*Archosargus probatocephalus*	x		x	x	x	x
Gambusia sexradiata	x				x	x	*Archosargus rhomboidalis*	x		x		x	x
Heterandria bimaculata[1]	x		x		x	x	*Lagodon romboides*[2]				x		
Poecilia catemaconis					x	x	*Antennarius striatus*					x	x
Poecilia mexicana	x		x	x	x	x	*Balistes capriscus*			x			
Poecilia sphenops		x	x	x	x	x	*Aluterus schoepfi*[2]		x				
Poeciliopsis sp					x	x	*Sphoeroides testudineus*				x		

Fish community structure and its relationship with environmental behavior in Sontecomapan Lagoon, Los Tuxtlas Biosphere Reserve, Veracruz, Mexico

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Abstract: The spatial and temporal variations of the fish community and their relationship with the physicochemical parameters in the Sontecomapan Lagoon System, Veracruz, Mexico, were analyzed. The physicochemical parameters of the surface and bottom water (salinity, temperature, dissolved oxygen and pH), as well as depth and transparency were recorded, and fishes were collected with shrimp trawls at 10 sampling stations for October 2014, February and October 2015, February 2016. To estimate spatial and temporal statistical differences between environmental variables and community parameters (diversity, richness, evenness, density, biomass and average weight), an analysis of variance was used. Cluster analysis was used to determine physicochemical similarity between sampling sites to define habitats. Canonical correspondence analysis (CCA) was performed to determine fish associations and their correlation with the environmental factors of the system. The spatial and temporal variations of the physicochemical parameters allowed to define three environments in the system, the first includes stations 1 to 4 with marine influence, the second includes stations 5 to 7 as a mixing zone and the third, stations 8 to 10 with fluvial influence. A total of 3,018 individuals with a total weight of 27,500 g were collected. We identified 18 families, 28 genera and 34 species, 2 of these are new records for the lagoon. The dominant species were *Diapterus rhombeus* and *Cathorops aguadulce*. Species diversity and richness declined from the

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marine inlet to the system header and the evenness trend was the inverse. The density and biomass were higher in the inner system while the inverse occurred for average individual weight. Temporally, diversity and abundance were the highest in February and October 2015. CCA showed that the environmental factors that determined the spatial and temporal composition and distribution of the fish community were salinity, temperature, dissolved oxygen, pH, depth and transparency.

Keywords: Physicochemical similarity, Diversity, Abundance, Dominance, Fish-habitat relationships

Introduction

Coastal lagoons and estuaries are transitional environments between land and sea permanently or temporarily connected by channels that allow exchange between organisms, nutrients and water. Tidal exchange and rivers discharge produce distinctive spatial and temporal variations that characterize them with complex environmental and ecological processes (Snedden *et al.*, 2013). These systems have a high diversity of invertebrates (mollusks and crustaceans) and vertebrates (birds, mammals and fish) that intensively use these environments for feeding, breeding, and protection (Lara Domínguez *et al.*, 2011; Díaz-Ruiz *et al.*, 2012).

Because they have a wide range of habitats, high biological diversity and significant primary productivity, however lagoon-estuarine coastal ecosystems of the Gulf of Mexico have characteristics of their own and a complex ecological structure, (Day *et al.*, 2013). Their productivity is due to the subsidies from rivers and tidal exchange, and from vegetation (mangrove forests, marsh grass, others such a freshwater wetlands) that either surrounds them or is submerged (seagrasses, macroalgae), in turn, these communities determine the volume of secondary production (Díaz-Ruiz *et al.*, 2003, Lara-Domínguez *et al.*, 2011).

Lagoons along the coast of Veracruz cover a 1166 km^2 area (7.4% of the national total). Among the most important ones, from North to South, are Pueblo Viejo, Tamiahua, Tampamachoco, Chica-Grande, San Agustín, Salada, Verde, El Llano, El Farallón, La Mancha, Mandinga, Alvarado, Sontecomapan and El Ostión (Lara-Domínguez *et al.*, 2011; Peresbarbosa, 2005).

Sontecomapan Lagoon, located to the South, is part of the mountain chain of Los Tuxtlas. This system belongs to the Biosphere Reserve of the same name and was decreed on November 23, 1998 (SEMARNAT, 2006). Also, it was declared a Ramsar site in 2004 (Ramsar Site 1342, FIR, 2004) because of the conservation degree of its mangroves and later worldwide acknowledged as a Biosphere Reserve within the “Man and Biosphere” UNESCO program (MAB) in October, 2006 (MAB, 2010).

Currently, ecological knowledge about fish communities in Sontecomapan Lagoon System is limited since no previous study has quantified their diversity and abundance. Therefore, it is priority to analyze current information on structure and function to better understand their ecological behavior and regional economic value.

Background

From late seventies to present, research has been conducted on fish that inhabit the lagoon systems in Veracruz, however, the tendency has been to contrast the changes in species composition through different years (Kobelkowsky, 1991; Pérez-Hernández & Torres-Orozco, 2000; Bedia-Sánchez & Franco-López, 2008; Lara-Domínguez *et al.*, 2011; Abarca-Arenas *et al.*, 2012). The work of Lara-Domínguez *et al.* (2011) summarizes that, for the different coastal environments in Veracruz, a total of 101 families have been recorded, including 478 species. Other contributions to this synthesis, regarding the available literature about Veracruz fish, are from undergraduate or graduate theses, technical reports, conferences and of some scientific articles published. Altogether, these papers comprise a total of 117 references where, only 4 correspond to the Sontecomapan Lagoon; therefore, it is necessary to gather more ecological information that allows a better comparison between systems and fish communities on the coast of Veracruz.

However, there is some progress on this subject as shown by the various publications for the State coastal systems. Such is the case of works by Chávez-López & Franco-López (1992), Chávez-López *et al.* (2005) and Franco-López *et al.* (2012) for the Alvarado Lagoon where different ecological aspects of the system, such as development and the environment regional impact on the fish community are analyzed, summarizing a record of more than 76 species. In Tamiahua Lagoon, Díaz-Ruiz *et al.* (2003) characterized the correlation of fish associations —including 32 families, 62 genera and 83 species— with environmental variables. Similarly, Castillo-Rivera *et al.* (2003, 2005, 2011) studied the lagoon system of Pueblo Viejo, analyzing spatial-temporal diversity and abundance patterns of the fish community —30 families, 52 genera and 66 species— in relation to hydro-meteorological variables, considering daytime and nychthemeral scales. For the Chica-Grande lagoon system, Aguirre-León *et al.* (2014) analyzed the environmental heterogeneity and its relationship with the diversity and fish community abundance, recording 14 families, 20 genera and 22 species, while Mendoza (2015) performed a temporary ecological assessment of the fish community in La Mancha Lagoon, emphasizing environmental processes that determine their behavior; 25 families, 37 genera and 50 species were identified. Also, Juárez (2015) analyzed in Ostión Lagoon, south of Veracruz, the spatial-temporal variations in structure and function of the fish community with relation to habitat dynamics, in this system; 19 families, 23 genera and 34 species were recorded.

Currently, ichthyological research for Sontecomapan Lagoon System is scarce, most of it is only a list of species. The oldest study is from Reséndez (1983) who related this lagoon's fish composition and hydrological data, the author recorded 26 families, 43 genera and 52 species; later, Fuentes-Mata *et al.* (1989) added new records for this specific system. Nava (2005) made a fish catalog for this location, including 18 families, 29 genera and 37 species. Rodríguez-Varela *et al.* (2010) analyzed the system fish fauna within two climatic seasons, recording 24 families, 42 genera and 52 species. Lara-Domínguez *et al.* (2011) summarized a total of 91 species for this system. The most recent publication that adds information on coastal fish in Veracruz is from Abarca-Arenas *et al.* (2012), which makes a synthesis of at least three decades and report for Sontecomapan a total of 33 families, 64 genera and 92 species including marine, estuarine and freshwater. Finally, Castillo-Rivera and Lara-Domínguez (this issue), from a comprehensive literature review on the fish fauna of the lagoon revealed that 115 species have been historically recorded but pointed out possible misidentifications and changes in taxonomic status.

Considering the ecological information scarce about the fish community structure in Sontecomapan Lagoon, the objectives of this study are: to analyze the variation of physicochemical parameters and define habitats within the system; determine species composition and types of community inhabitants; define trophic levels based on the species feeding habits; determine dominant species on the fish community; analyze diversity, richness, evenness, density, biomass and average weight in the system, and determine which physicochemical factors condition the fish assemblages' behavior.

Materials and Methods

Study Area

Sontecomapan Lagoon is in the western of the Gulf of Mexico (018° 32' N, 095° 02' W) and is part of the Los Tuxtlas region in southern Veracruz (Fig. 1). The system is roughly 12 km long and 1.5 km wide with a surface area of 8.9 km^2 and a perimeter of 36.5 km (FIR, 2004). It is shallow, with sediments ranging from sands to silts. It is permanently connected to the sea through an inlet of about 5.5 m depth called "The Bar of Sontecomapan" (Contreras, 1993). The lagoon has a considerable water input from rivers and streams, the most important are: Coxcoapan, Yuhualtijapan, La Palma and El Sábalo. The most significant streams are: Los Pollos, Sontecomapan, El Fraile, La Boya and Basura (Fig. 1). The climate is warm and wet, with an annual rainfall of 4500 mm. The rainy season is from June to September and the dry season from March to May (García, 2004). The driest month is May and the peak of the rainy season is from August to November. The movement of cold and dry air masses from the north affects the area from October to February causing cold fronts that are locally called “Nortes” (Soto-Esparza & Giddings, 2011). There are different aquatic environments in the system, such as the surrounding mangrove vegetation in brackish and low salinity areas mostly with *Rhizophora mangle* and *Avicennia germinans* and submerged vegetation represented by *Ruppia maritima* (Contreras, 1993; Carmona-Díaz *et al.*, 2004).

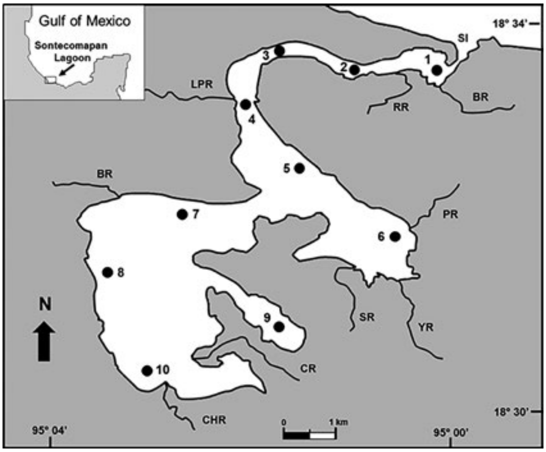

Figure 1. Sontecomapan Lagoon System, Veracruz. The location of sampling stations (•), the main physiographic features of the study area and rivers draining into the system are noted. Palma River (LPR), Basura River (BR), Chuniapan River (CHR), Coxcoapa River (CR), Sábalo River (SR), Yuhualtijapan River (YR), Pollos River (PR) Sontecomapan Inlet (SI).

Field and laboratory activities

Field surveys were conducted in the months of October 2014, February and October 2015 and February 2016. A network of 10 sampling stations was set for collecting fish (Fig. 1), they were placed considering the marine inlet and tidal channel, sediment type, submerged and emergent vegetation, and fresh water flow and contribution to the system. Catches were obtained trhough a 5.0 m long shrimp trawl, with a 2.5 m work opening and a 1.8 cm mesh. This was done for two days every month, sampling the first day and taking a replicate the second. Two 10-minute trawls at an average speed of two knots were made at each station —one per working day— for a total of 20 hauls per month. The trawling time corresponds to a sampling area of 1500 m^2. The fish were stored in labeled plastic bags and fixed with a solution of formaldehyde at 10% for transport to the laboratory.

At each sampling site, salinity, temperature and dissolved oxygen in the water column (surface and bottom levels) were measured with a YSI85 multiparametric meter, pH with a Hanna meter, transparency of the water column with a Secchi disc, depth with a digital depth probe, and environment temperature with a pocket weather meter.

Taxonomic identification of the species was carried out using identification sheets and descriptions based on the work of Reséndez (1981a, 1981b), Castro-Aguirre *et al.* (1999), Fischer (1995) and Miller (2009), each species was weighed with a digital Ohaus balance (accuracy 0.01 g) and measured using a conventional ichthyometer in cm for the record.

The community inhabitants types were determined using the Yáñez-Arancibia (1986) criteria, who classifies species as: a) Estuarine-lagoon permanent inhabitant (E), b) Marine euryhaline species (ME), c) Marine stenohaline species (MES), d) Freshwater species (F). The trophic category of species was defined per published information on each species feeding habits, with the following criteria: a) First order consumers, mainly planktivorous fish that consume phytoplankton and zooplankton, b) Second order consumers, omnivorous fish, that include vegetables and detritus in their diet, c) Third-order consumers, fish that are exclusively carnivorous, and their vegetables and detritus consumption is accidental (Yáñez-Arancibia,1986). The species dominance was determined by using the Importance Value Index (IVI=%N+%W+%F) as the sum of percentage in number plus the species weight plus the frequency of their occurrence (De la Cruz-Agüero, 1994).

Community ecological parameters were estimated: diversity (H') (Shannon & Weaver, 1963), species richness (D) (Margalef, 1969) and evenness (J)' (Pielou, 1966), as well as density (ind/m^2), biomass (g/m^2) (Brower &Zar, 1977) and average fish weight (g/ind) (Aguirre-León *et al.*, 1998).

Statistical analysis

Spatial (between sampling stations) and temporal (monthly) statistical differences of community's environmental variables and ecological parameters were estimated with an analysis of variance. Homogeneity of variance between groups was previously evaluated using the Levene test (1960). Logarithmic transformations were made when necessary. When variances were homogenous, an ANOVA F-test was used, otherwise the Kruskal-Wallis H test (K-W) was used (Zar, 1999). A significance level of $p < 0.05$ was considered for all these tests (Zar, 1999; Hair *et al.*, 2001).

Based on the spatial-temporal values of the physicochemical parameters, habitats were defined using a cluster analysis to determine environmental similarity between stations. The methods used were Euclidean distance coefficient and the clustering algorithm was the weighted average technique (Hair *et al.*, 2001).

To determine associations of fish and its correlation with environmental factors, a canonical correspondence analysis (CCA) was performed (Ter Braak & Verdonschot, 1995). This was done with an array of abundance of the species per station and month, and an array of average values of the physiochemical parameters. The CCA was done with version 3.22 of the Multivariate Statistical Package (MVSP).

Results

Spatial and temporal variation of environmental parameters

The analysis of physicochemical data indicated that the system has a very dynamic spatial behavior related to its size and geomorphology. Environmental parameters depend on the seawater through its small inlet and freshwater from the rivers (Fig. 2). The highest average surface salinity (21.4±11.5) was recorded at station 1 (Fig. 2a) and the lowest (2.4±2.3) in station 10, with significant statistical differences between stations, K-W (9,80) H=31.08, *P*=0.0003. At the bottom, the highest average (29.4±4.5) was recorded at station 3 (Fig. 2a) and the lowest (7.7±5.2) at station 10, with significant differences between stations, K-W (9,80) H =41.20, *P*<0.0001. There also were differences between surface and bottom (Table 1). The highest average temperatures at the surface (28.2°C±2.9) and bottom (28.8±2.9) were recorded at station 9 (Fig. 2b), and the lowest (26.4 ±3.5 and 26.5 ±3.6), at station 1. There were no significant differences between stations, neither in surface, bottom or between these two levels (*P*>0.05) (Table 1). The highest average dissolved oxygen on the surface (5.8mg/L±2.0) and bottom (4.7±1.8) was recorded at station 9 (Fig. 2c), while the lowest values were those from stations 1 (4.7 ±1.5) and 3 (4.8 ±1.3). There were no significant differences between stations, neither on the surface or at the bottom (*P*>0.05), however, there were differences between these levels (Table 1). The highest average pH values at the surface (7.8±0.3m) and bottom (7.9±0.4) were recorded at station 1 (Fig. 2d), whereas the lowest were at stations 10 (7.3±0.34) and 8 (7.0±0.38). There were significant statistical differences for pH between surface stations, ANOVA (9,70) F=2.46, *P*=0.0167 and between bottom stations, ANOVA (9,70) F=3.50, *P* =0.0013; however, there were no differences between levels (Table 1). The largest average depth (3.46m±1.1) was recorded at station 2 (Fig. 2e) and the smallest (1.15 ±0.15) at station 9, with significant differences between stations: K-W (9,80) H=49.74, *P*<0.0001 (Table 1). The highest average transparency of the water column (1.82m±0.6) was recorded at station 1 (Fig. 2e) and lowest at station 6 (0.78±0.25) and station 8 (0.78±0.15), with significant differences between stations, K-W (9,80) H=36.76, *P*<0.0001 (Table 1).

Regarding temporal variation (Fig. 2), physicochemical variables indicated changes between months. Highest average salinity at the surface (16.6ups±13.8) and bottom (21.6±12.9) was recorded in February 2016 (Fig. 2f), while the lowest surface (5.3±6.6) and bottom (15.9±9.3) values were in October 2014, with significant differences only for surface data, K-W (3,80) H =36.17, *P*=0.00001. Highest average temperatures at the surface (29.9°C±1.5) and bottom (31.2±1.7) were recorded in October (Fig. 2g) and the lowest (24.2 ±1.5 and 23.8±1.6 respectively), in February (2015) thus reflecting monthly temperature variation. There were significant differences for surface, ANOVA (3,76) F

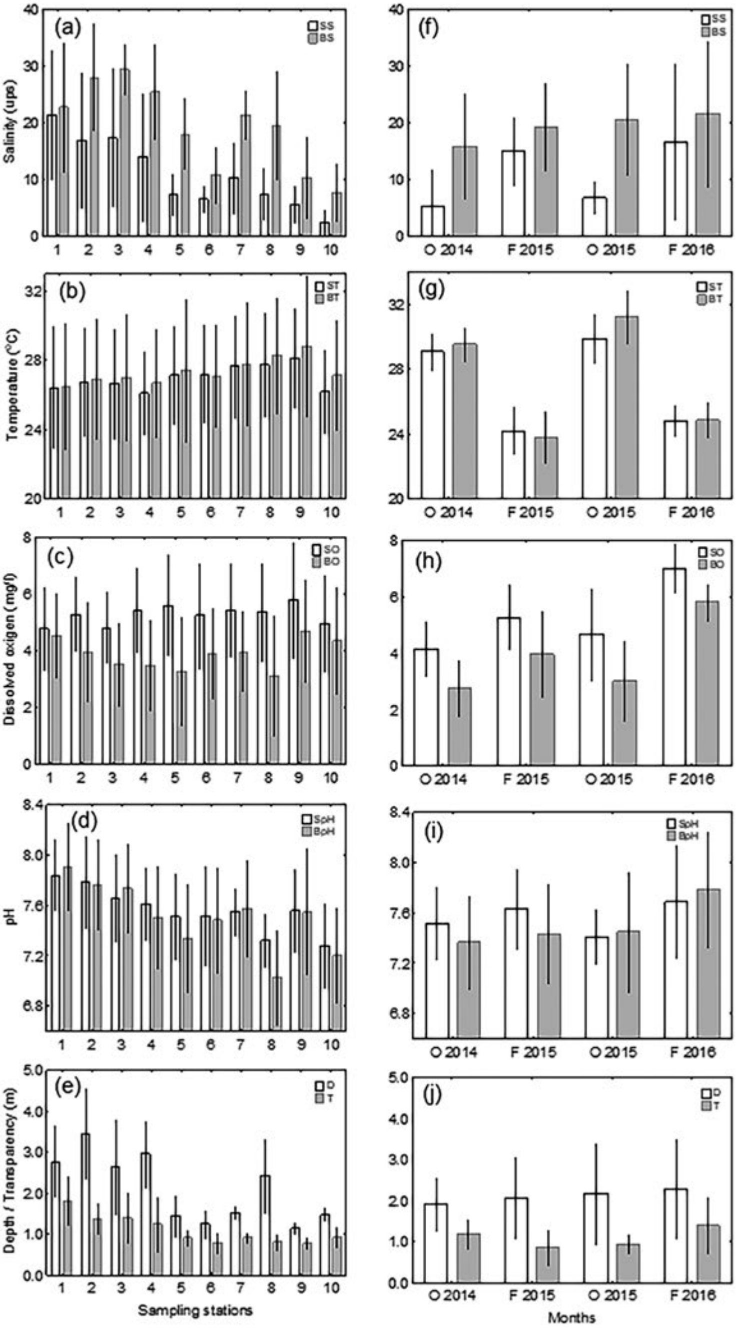

Figure 2. Spatial and temporal average values (± 1 SD) of the physicochemical parameters in the Sontecomapan Lagoon System. For each environmental variable, the surface (S) and bottom (B) values are shown.

=101.39, *P*<0.0001, and bottom values, ANOVA (3,76) F =132.62, *P*<0.0001. Average dissolved oxygen concentrations at the surface (6.9mg/L±0.9) and bottom (5.8±0.7) were higher in February 2016 (Fig. 2h), and lower (4.2±1.0 and 2.7±1.0, for surface and bottom respectively), in October 2014, with significant differences for surface ANOVA (3,76) F=21.22, *P*<0.0001, and bottom, ANOVA (3,76) F=25.76, *P*<0.0001. Highest average pH values at the surface (7.7±0.4) and bottom (7.8±0.5) were recorded in February 2016 (Fig. 2i), while the lowest values were recorded in October 2015 for the surface (7.4±0.2), and October 2014 for the bottom (7.4±0.4), with significant differences between month for both surface ANOVA (3,76) F=2.85, *P*=0.043, and bottom, ANOVA (3,76) F=3.80, *P*=0.013. Average depth of the system ranged from 1.2±2.3m in February 2016 (Fig. 2j) to 1.9±0.7 in October 2014, with no significant differences between months (*P*>0.05). Highest average value for transparency (0.7±1.4m) was in February 2016 (Fig. 2j) and the lowest (0.9±0.4) in February 2015, with significant differences between months, K-W (3,80) H=19.79, *P*=0.0002.

Based on the spatial-temporal variation of physicochemical parameters, cluster analysis indicated three groups or habitats (Fig. 3a), the first one went from station 1 to 4, which grouped the marine inlet and the tidal channel of the lagoon where water exchange with the adjacent continental shelf is important. (Fig. 3b). The second group consists of stations 5, 7 and 8 located in the middle section of the lagoon where mixing processes of water from the sea and river drainage happen. The third group includes stations 6, 9 and 10, which corresponds to areas that are mostly influenced by freshwater such as El Sábalo, El Fraile and the mouth of Arroyo Chuniapan (Fig. 3a and 3b). Salinity and pH from surface and bottom, as well as depth and transparency were the defining parameters for these environments since they indicated significant statistical differences at both spatial and temporal scales (Fig. 3, Tables 1 and 2).

Fish composition

A total of 3,018 individuals with a total weight of 27,500.1 g were captured during the study for an overall of 18 families, 28 genera and 34 species identified (Table 3). The most represented family was Gerreidae with 6 species, *Diapterus rhombeus, Diapterus auratus, Eucinostomus argenteus, Eucinostomus gula, Eucinostomus melanopterus* and *Eugerres plumieri*.

Some of the 34 species were caught in just a month. Such is the case for *Bardiella ronchus, Eleotris pisonis* and *Polydactylus octonemus* in October, 2014. During February, 2015, was *Mugil curema* and *Scorpaena plumieri*. In contrast, the period with most species was October, 2015 when *Caranx crysos, Chloroscombrus chrysurus, Eucinostomus argenteus, Gobiomorus dormitor, Lutjanus analis, Lutjanus synagris, Selene vomer, Sphoeroides splengeri* and *Syn-*

odus foetens were recorded, whereas for February 2016 only *Mayaheros urophthalmus* was captured. Documented as new records for Sontecomapan Lagoon were *Synodus foetens and Sphoeroides splengeri.*

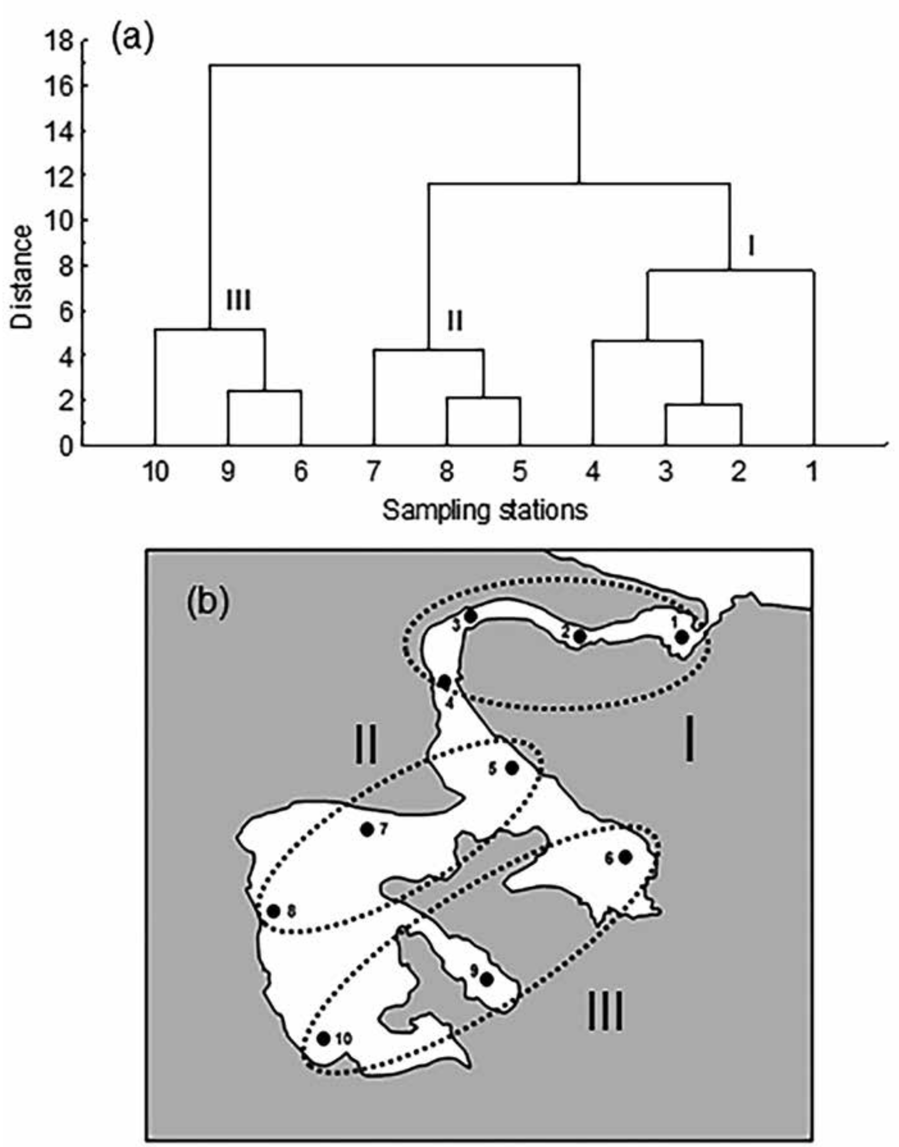

Figure 3. (a) Station groups with similar environmental characteristics in the Sontecomapan Lagoon System. (b) Spatial distribution of the three habitats defined in the system.

Table 1. Average values (± 1 SD) and statistical estimators of the physicochemical variables between surface and bottom for Sontecomapan Lagoon System, Veracruz

Parameter	Surface	Bottom	ANOVA Model
Salinity	10.9 ±9.6	19.4 ±10.2	H (1,160) =2773, P<0.0001**
Temperature	26.9 ± 2.9	27.4 ±3.4	F (1,158) =0.553, P =0.458
Dissolved oxygen	5.28 ±1.6	3.89 ±1.7	F (1,158) =28.39, P<0.0001**
pH	7.56 ±0.3	7.51 ±0.4	F (1,158) =0.731, P =0.393
Depth	2.1 ±1.1		H (9, 80) =49.74, P <0.0001*
Transparency	1.1 ±0.5		H (9, 80) =36.76, P <0.0001*

* statistical differences between sampling stations
** parameters with significant statistical differences between levels

Table 2. Average values (± 1 SD) and statistical estimators of the physicochemical variables for each station groups in the Sontecomapan Lagoon System, Veracruz.

Parameter	Group 1	Group 2	Group 3	ANOVA Model
Surface salinity	17.4 ±11.47	8.3 ±4.99	4.8 ±3.14	H (2, 80) =18.92, *P*<0.0001*
Bottom salinity	26.45 ± 8.81	19.6 ±6.92	9.6 ±5.83	F (2, 77) =34.81, *P*<0.0001*
Surface temperature	26.4 ±2.96	26.5 ±2.83	27.1 ±2.73	F (2, 77) =0.97, *P* =0.382
Bottom temperature	26.7 ±3.33	27.8 ±3.57	27.7 ±3.42	F (2, 77) =0.79. *P* =0.454
Surface oxygen	5.1 ±1.34	5.5 ±1.66	5.3 ±1.84	F (2, 77) =0.44, *P* =0.645
Bottom oxygen	3.9 ±1.56	3.5 ±1.8	4.3 ±1.73	F (2, 77) =1.55, *P* =0.217
Surface pH	7.7 ±0.32	7.5 ±0.26	7.4 ± 0.36	F (2, 77) =6.65, *P* =0.002*
Bottom pH	7.7 ±0.38	7.3 ±0.45	7.4 ±0.45	F (2, 77) =7.42, *P* = 0.001*
Depth	3.0 ±1.0	1.8 ±0.75	1.3 ±0.26	H (2, 80) =42.17, *P*<0.0001*
Transparency	1.5 ±0.6	0.9 ±0.18	0.8 ±0.22	H (2, 80) =32.10, *P*<0.0001*

Group 1: stations 1, 2, 3 and 4. Group 2: stations 5, 7 and 8. Group 3: stations 6, 9 and 10
* Parameters with significant statistical differences between groups.

Table 3. List of the fish species collected at the Sontecomapan Lagoon System, Veracruz

Species	Abbreviations (AB)	Individuals Number	Weight (g)	Number (%)	Weight (%)	Occurrence (%)	Community inhabitant	Trophic level	IVI (%)
*Diapterus rhombeus***	Dr	2070	8140.3	68.59	29.60	13.16	ME	1	111.3
*Cathorops aguadulce***	Ca	369	9290.8	12.23	33.78	11.84	E	3	57.9
Citharychthys spilopterus	Cs	87	602.4	2.88	2.19	9.65	ME	3	14.7
Eucinostomus melanopterus	Em	49	394.9	1.62	1.44	9.65	ME	1	12.7
Diapterus auratus	Da	128	872.0	4.24	3.17	8.33	ME	1	15.7
Eugerres plumieri	Ep	58	1776.4	1.92	6.46	6.58	ME	1	15.0
Centropomus parallelus	Cp	40	1038.0	1.33	3.77	7.02	ME	1	12.1
Sciades felis	Sf	63	1319.2	2.09	4.80	5.70	ME	2	12.6
Achirus lineatus	Al	29	119.3	0.96	0.43	5.26	ME	2	6.7
Anchoa mitchilli	Am	21	32.9	0.70	0.12	2.19	ME	1	3.0
Hypanus sabinus	Hs	3	1449.9	0.10	5.27	1.32	ME	3	6.7
Eucinostomus gula	Eg	14	96.9	0.46	0.35	1.75	ME	1	2.6
Opsanus beta	Ob	3	743.7	0.10	2.70	1.32	ME	2	4.1
Polydactylus octonemus	Po	12	180.0	0.40	0.65	1.32	ME	2	2.4
Eucinostomus argenteus	Ear	9	21.7	0.30	0.08	1.32	ME	1	1.7
Lutjanus apodus	Lap	3	52.9	0.10	0.19	1.32	ME	3	1.6
Mugil curema	Mc	6	232.4	0.20	0.85	0.88	ME	3	1.9
Centegraulis edentulus	Ce	19	45.2	0.63	0.16	0.88	ME	2	1.7
Gobionellus hastatus	Gh	7	146.5	0.23	0.53	0.88	ME	1	1.6
Centropomus undecimallis	Cu	2	104.9	0.07	0.38	0.88	ME	3	1.3
Synodus foetens	Sfo	2	90.5	0.07	0.33	0.88	ME	3	1.3
Trinectes maculatus	Tm	6	28.3	0.20	0.10	0.88	ME	2	1.2
Caranx crysos	Cc	3	54.3	0.10	0.20	0.88	ME	2	1.2
Selene vomer	Sv	2	12.4	0.07	0.05	0.88	ME	2	1.0
Hemicaranx amblyrynchus	Ha	2	3.9	0.07	0.01	0.88	ME	2	1.0
Chloroscombrus chrysurus	Cch	2	3.4	0.07	0.01	0.88	ME	2	1.0
Scorpaena plumieri	Sp	1	286.9	0.03	1.04	0.44	MES	3	1.5
Bairdiella ronchus	Br	2	130.0	0.07	0.47	0.44	ME	2	1.0
Gobiomorus dormitor	Gd	1	100.6	0.03	0.37	0.44	E	3	0.8
Lutjanus analis	La	1	82.8	0.03	0.30	0.44	ME	3	0.8
Sphoeroides spengleri	Ss	1	28.1	0.03	0.10	0.44	MES	2	0.6
Mayaheros urophthalmus	Mur	1	14.0	0.03	0.05	0.44	F	2	0.5

Table 3. (Continuation)

Species	Abbreviations (AB)	Individuals Number	Weight (g)	Number (%)	Weight (%)	Occurrence (%)	Community inhabitant	Trophic level	IVI (%)
Lutjanus synagris	Ls	1	4.5	0.03	0.02	0.44	MES	1	0.5
Eleotris pisonis	Epi	1	0.5	0.03	0.00	0.44	E	1	0.5

Abbreviations (AB) used in figure 6 are shown (CCA). Community inhabitants: ME=Marine euryhaline, MES=Marine stenohaline, E=estuarine-lagoon permanent inhabitant, F=freshwater. Trophic Level: 1=planktivorous, 2=omnivores, 3=carnivorous. IVI=Importance Value Index.

** system's dominant species

Community inhabitants

From the species collected in the system, 27 (79.4%) are Marine euryhaline (ME) with *D. rhombeus* being the most abundant of this category. The group of Marine stenohaline species (MES) has 3 species (8.8%), *L. synagris*, *S. plumieri* and *S. splengeri* whereas *C. aguadulce*, *G. dormitor* and *E. pisonis* are estuarine-lagoon permanent inhabitant (E) and only one species, *M. urophthalmus*, is from freshwater (F) (Table 3).

Trophic categories

Within the trophic category of first order consumers (planktivorous), 11 species (33%) were found; 13 species (37%) are second order consumers (omnivores), and 10 species (30%) are third order consumers (exclusively carnivores) (Table 3).

Dominant Species

During this study, two species were established as dominant in the system based on their Importance Value Index (IVI), *D. rhombeus* (IVI=111.3%) and *C. aguadulce* (IVI=57.9%). These two-species accounted for 80.8% (2,439 individuals) of numerical abundance and 63.4% (17,431.0 g) of the total catch of the community (Table 3).

Spatial and temporal variation of diversity

In general, the spatial analysis of H' and D showed a tendency with lower values towards the marine inlet and higher inside the system, the value of J' had an inverse trend (Fig. 4). The average number of species ranged from 2 in station 1, to 8 in stations 9 and 10; the highest average diversity H' (1.43 ±0.34) was recorded at station 5 (Fig. 4a) and the lowest (0.71 ±0.5) at station 1. Richness D (Fig. 4b) was higher in stations 5 (1.59 ±0.22) and 9 (1.58 ±0.26) and the lowest in station 3 (1.03 ±0.96). The evenness J' (Fig. 4c) was higher (0.81 ±0.07) at station 4 and the lowest (0.44 ±0.15) in station 8. The behavior of the three indices is related to a low number of species and the dominance of some of them in the sampling stations. There were no significant differences between sites for any of these indices ($P > 0.05$).

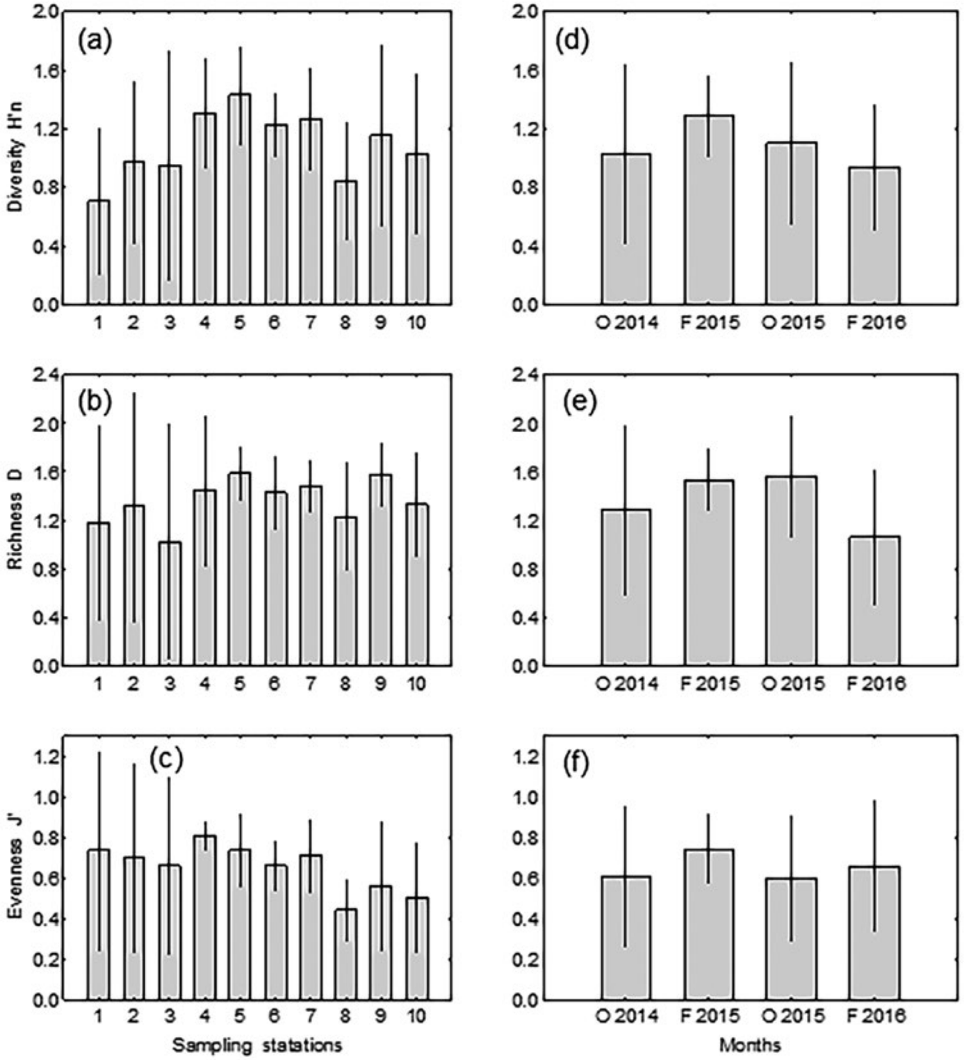

Figure 4. Spatial and temporal average values (± 1 SD) of the diversity indices H', D and J' for the fish community in the Sontecomapan Lagoon System.

The temporal variation of H', D and J' is shown in Figure 4. The total number of species changed between sampling periods: 17 in October, 2014; 16 in February, 2015; 25 in October, 2015; and 14 in February, 2016. The highest average H' (1.28 ±0.28) occurred in February, 2015 (Fig. 4d) and the lowest (0.94 ±0.43) in February, 2016. D index was the highest (1.56 ±0.543) in October, 2015 (Fig. 4e) and the lowest (1.1 ±0.51) in February, 2016. Maximum average J' (0.74 ±0.17) was in February 2015 (Fig. 4f), while the minimum values occurred in October, 2014 (0.61 ±0.35) and 2015 (0.60 ±0.31). Both H' and D average monthly values tend to be low. There were no significant differences in the average values of these indices between months (P>0.05).

Spatial and temporal variation of abundance

Figure 5 shows values of fish density (ind/m^2), biomass (g/m^2) and average weight (g/ind). Therefore, the highest average value in density (0.096 ind/m^2 ±0.12) was recorded at station 10 and the lowest at stations 1 (0.0005 ±0.0006) and 2 (0.001 ±0.001) (Fig. 5a), with significant statistical differences between stations, K-W (9,40) H=29.07, P=0.0006. Species that showed higher density were *D. rhombeus, C. aguadulce* and *D. auratus*. The highest average biomass (0.45 g/m^2 ±0.27) was recorded in station 10 and the lowest (0.014 ±0.02) in station 2 (Fig. 5b), with significant statistical differences between sites, K-W (9,40) H=29.67, P =0.0049. Species with higher biomass were *C. aguadulce* and *D. rhombeus*, and, to a lesser amount *E. plumieri*, *Hypanus sabinus*, *Sciades felis* and *Centropomus parallelus*. Highest average weight (34.2 g/ind ±36.3) was obtained at station 1 and the lowest at station 7 (8.3 ±5.4) (Fig. 5c), showing an inverse relation between this parameter and the other two. Species with the highest average weight were *H. sabinus*, *S. plumieri* and *Opsanus beta*. This parameter showed no significant differences between sites (P> 0.05).

Throughout this study, the highest average density (0.055 ind/m^2 ±0.08) was recorded in October, 2015 and the lowest (0.012 ±0.015) in February, 2016 (Fig. 5d), the rest of the time, it was also low. The highest average biomass and weight (0.22 g/m^2 ± 0.13 and 22.16 g/ind ±21.8, respectively) were recorded in February, 2015 (Fig. 5f) and the lowest (0.15 g/m^2 ±0.16 and 12.1 g/ind ±6.6) in February, 2016. There were no significant differences between months for any of these parameters (P>0.05).

Fish-habitat analysis

The ordination diagram (Fig. 6) shows the relationship between species, stations and environmental variables as well as the relative importance of each environmental variable that determines the composition and distribution of fish in the community. The CCA showed that 71.9% of the total variance was explained by the relationship between the abundance of species and environmental variables, with eigenvalues of 0.348 (axis 1) and 0.105 (axis 2). CCA explained 55.2% of the variation for species-environment axis 1, indicating that the most important factors in the distribution of species are transparency (r=0.80), salinity at the bottom (r=0.65), depth (r=0.57) and pH at the bottom (r=0.45). Species that showed a high correlation with axis 1 were, *Citharychthys spilopterus*, *E. plumieri* and *E. melanopterus* from the euryhaline species, *L. synagris*, *S. plumieri* and *S. splengeri* from the stenohaline, and *E. pisonis* among estuarine fish. Low and high abundances of these species were associated to areas with higher and intermediate depths (stations 2, 3, 4, 5, 6, 7) with higher salinity and transparency.

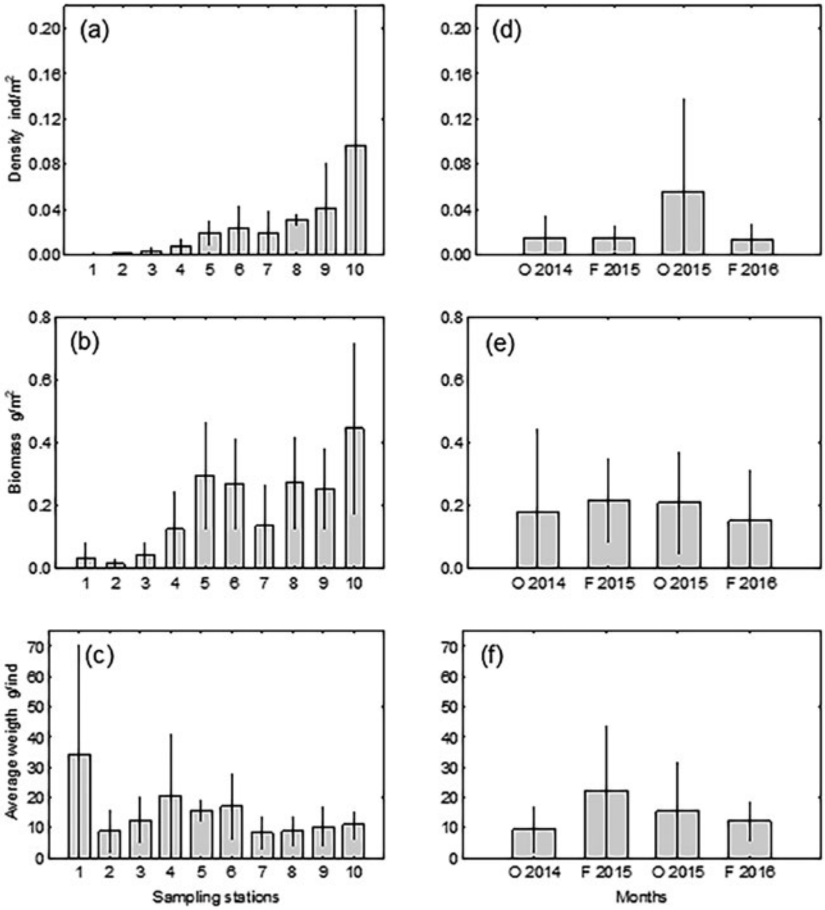

Figure 5. Spatial and temporal average values (± 1 SD) in density, biomass and average weight for the fish community in the Sontecomapan Lagoon System.

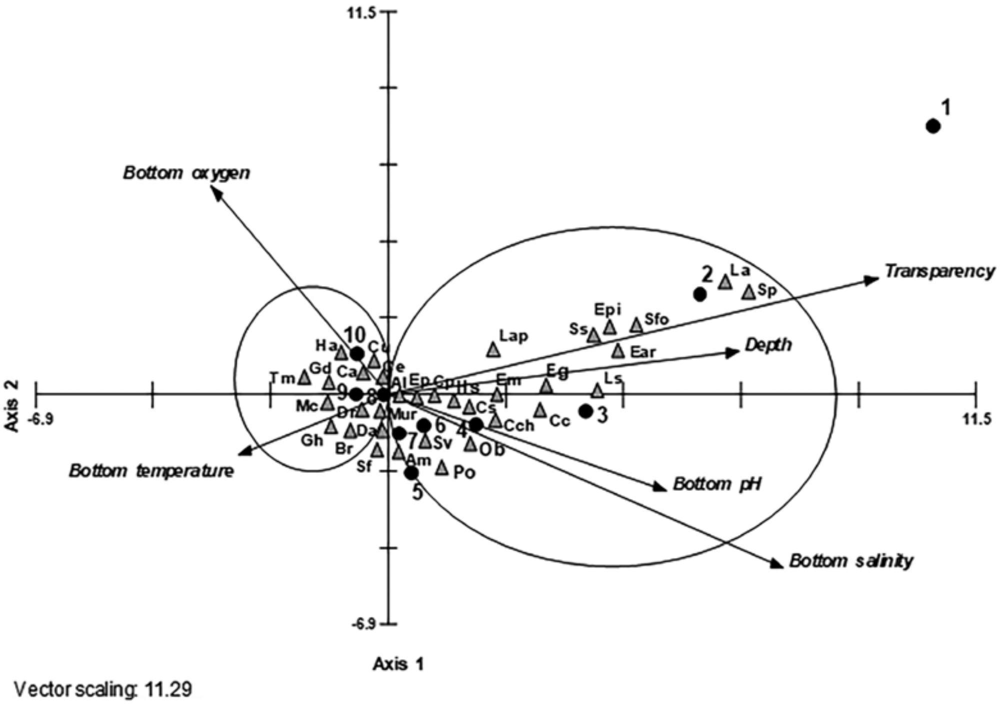

Figure 6. Canonical Correspondence Analysis Plot of species abundance, environmental variables and sampling stations in the Sontecomapan Lagoon System. Arrow length and direction indicate the relative importance of environmental variables. Species abbreviations (AB) are listed in Table 3.

Axis 2 explained 16.68% of species-environment variation with bottom dissolved oxygen (r=0.48) and bottom temperature (r=-0.14) as the most important factors that explain the distribution of species along this axis. Euryhaline species that showed high correlation with axis 2 were *D. rhombeus*, *D. auratus* and *S. felis,* the estuarine species were *C. aguadulce* and *G. dormitor,* and *M. urophtalmus* from freshwater. The abundance of these species was intermediate and high, strongly related to temperature and also associated to shallow areas (stations 8, 9 and 10) with lower salinity and high oxygen levels.

Discussion

The variability of physicochemical parameters in Sontecomapan Lagoon System is conditioned by its shape and size, by sea water input through the marine inlet and tidal channel from the Gulf of Mexico, and by freshwater from rivers and streams that flow into the lagoon. Salinity had a spatial gradient with higher values close to the sea channel and lower near the system header. This was previously described by Martínez (1987), Morán (1994), Castellanos (2002) and Rodríguez-Varela *et al.* (2010) who indicated high values in the inlet and the tidal channel, and lower into the system. The highest average salinities were recorded in February, 2015 (rainfall 58.5 mm; CONAGUA, 2015) and 2016 (rainfall 55.0 mm; CONAGUA, 2016) due tu reduced precipitation and increased tide effect into the system, while the lowest were found in October, 2014 (rainfall 660.5 mm; CONAGUA, 2014) and 2015 (rainfall 872.3 mm; CONAGUA, 2015). The variation in salinity was thus closely related to the climatic changes of this coastal region (Morán, 1994; Castellanos, 2002; and Rodriguez-Varela *et al*., 2010). In this study, the data showed a vertical stratification of the whole lagoon within the analyzed months.

Water temperature showed little variation throughout the system, with differences of 1.1°C to 1.3°C between recording stations and between surface and bottom. Martínez (1987), Castellanos (2002) and Rodríguez-Varela *et al.* (2010) also recorded an average fluctuation of ±1°C due to solar radiation heating the water column of this shallow system. In relation to the thermal change, the highest values were in October, 2014 and 2015 and the lowest in February 2015, and 2016 when the reduction of air temperature extends to the lagoon. Aguirre-León *et al.* (1998) in a coastal system in Campeche and Castellanos (2002) in Sontecomapan Lagoon, both described similar conditions.

Dissolved oxygen showed little spatial variation in the system, however, higher values were recorded at the surface and lower at the bottom. This stratified distribution was related to the salt wedge described above and associated with the tide and local winds effect that exchange oxygen in the water column (Morán,1994). Castellanos (2002) reported high dissolved oxygen values into the system, due to photosynthetic activity of submerged vegetation in shallow areas. Monthly changes were observed, with the lowest values recorded in October, 2014 and 2015 and the highest in February, 2015 and 2016. This has a relation with the thermal variations described above and the frequency and intensity of the "nortes" wind that increase oxygen exchange in the water column (De la Lanza, 1994).

Depth and transparency showed a similar spatial trend, with higher values in the marine inlet and tidal channel (stations 1-4) and lower towards the head of the system (stations 8-10). Currently the Sontecomapan Lagoon System has an excess of sediment, due to the channel narrowness in its marine inlet and to the accumulation of deposits that are stream-carried from its eroded banks as a result of animal breeding and agriculture (SEMARNAT, 2006). Monthly depth has not significant differences, however, the lowest was registered in October, 2014 and the highest in February, 2016. Transparency values were inverse to depth in October, 2014, February and October, 2015. Both parameters are influenced by the intensity of precipitation and the "Nortes" phenomena; Zamora (2002), Castellanos (2002) and Rodríguez-Varela *et al.* (2010) also observed this behavior.

The cluster analysis of the system's environmental data showed that salinity, pH, depth and transparency defined three environments or habitats with statistically significant differences between them. The spatial behavior of these three environments remained unchanged in the analyzed months. Morán (1994), García-Cubas & Reguero (1995) and Castellanos (2002) described a similar environmental pattern according to the relationship between the distribution of habitats and the physicochemical variables, water movement in the lagoon, nutrient concentration or input from rivers and seasonality.

The fish community of the Sontecomapan lagoon has been under study since 1979 (Zamora, 2002), with most of the reports being only species listings without any additional ecological analysis, and no attempt to analyze why species composition has spatial-temporal changes. Therefore, there is no retrospective analysis of the change in the number of species at the community or any certainty over the past three decades. Although previous studies are available, they have been conducted with different sampling techniques, which results in different fish composition, so it is reasonable to assume that there is still no complete record of all species in the Sontecomapan system. The latest integrated species publication for this lagoon reported 92 (Abarca *et al.*, 2012) however, Fuentes-Mata *et al.* (1989) had reported 97; recently, Espinoza (2017) indicates that the historical record of fish in the system counts more than 100 species. Based on available literature on ichthyology and data from this research, *S. foetens* and *S. spengleri* species are new records for the Sontecomapan Lagoon System.

Yáñez-Arancibia & Nugent (1977) used the concept of the ecological role that community inhabitants play in a system and this concept was later expanded by criteria from Castro-Aguirre *et al.* (1999). In tropical coastal ecosystems, it is common to find the four types of community components found in Sontecomapan using these criteria. The ecological behavior of these species is due to physiological adaptations that allow them to use saline environments with large temperature changes and to move into the continental and marine environment as eury-thermohaline species (Able, 2005), such is the case of *D. rhombeus* that was the dominant species (69% abundance, IVI=111.3%) in the four months analyzed. Conversely, from an evolutionary perspective, there are stenohaline species that are only adapted to enter the ecosystem when salinity conditions only show slight changes and favor them. Generally, these are rare species (Cowan *et al.*, 2013) such as the stone fish *S. plumieri* (IVI=1.5%) and the puffer *S. spengleri* (IVI=0.6%), which were captured in the tidal channel near the marine inlet. Other species can remain throughout their life cycle and move across all the physical and chemical system gradients all year long, these so-called estuarine species occur in low proportion, as is the case of *C. aguadulce* (IVI=57.9%). Additionally, the species *M. urophthalmus* (IVI=0.5%) (*sensu lato Cichlasoma urophthalmus,* Rícan *et al.*, 2016) is a typical freshwater component captured in the habitat only under fluvial influence. However, in other coastal systems of the southern Gulf of Mexico and the Caribbean Sea, this species has been collected in brackish and marine environments, demonstrating its great adaptability to habitats other than their original (Miller, 2009). Studies that analyzed the proportion of community inhabitants for Sontecomapan Lagoon reported that most of the species were from the euryhaline component (Martínez, 1987; Zamora, 2002). For other systems, such as the Alvarado and Chica-Grande lagoons, there were also more euryhaline inhabitants and less stenohaline and estuarine ones (Sánchez, 2003; Aguirre-León *et al.*, 2014).

Based on the criteria proposed by Yáñez-Arancibia (1986), and the available information on feeding habits, we found that the ichthyofauna of Sontecomapan was represented by the three categories of defined consumers, where the number of species belonging to the second-order consumers (omnivores; n=13) was higher than the number of first-order (planktivorous; n=11) or third-order (carnivorous; n=10) consumers. However, the second-order has the lowest total abundance (total individual number= 143 and total weight= 2,682.0 g), compared to trophic level 1 (total individual number=2,398, total weight=12,525.0 g) and third-order (total individual number= 475, total weight= 12,294.0 g). Based on stomach contents, Alarcón (2007) determined, 6 first-order, 19 second-order and 10 third-order consumers and stated that the plant-derived organic matter, detritus and zooplankton are the main food sources that support the trophic structure of this system. In other coastal systems in the Gulf of Mexico, as Términos Lagoon, the most abundant orders were first and second (Álvarez-Guillén *et al.*, 1985). The same pattern of trophic categories was found for Raya Lagoon in Venezuela (Ramírez Villaroel, 1994). Likewise, Alvarez-Rubio *et al.* (1986), have pointed out that the spatial-temporal variation of the trophic levels is related to the species' ecological behavior, the environment they use and the selection of fishing nets. The same authors indicated that second-order species have a broad trophic spectrum, allowing them to be dominant (as is the case of *C. aguadulce* and *D. rhombeus* in Sontecomapan) since, because of their adaptive capabilities, they have different feeding options in different situations. Identifying what type of consumers use a coastal system could tell several aspects of the energy flow (food webs) such as producer-consumer and prey-predator relations, food availability and adaptations for consumption (Zárate-Hernández *et al.*, 2007; Hagy & Kemp, 2013). Another aspect of the trophic levels structure in a system is the size of the species, as it indicates what stage of their life cycle develops in the ecosystem. In Sontecomapan Lagoon, small sized of *D. rhombeus* (5.0 to 6.0 cm) to medium sized of *C. aguadulce* (16 to 18 cm) predominate, suggesting that habitats are used as nursery areas, as well as for feeding and growth. This ecological behavior has been well documented in tropical coastal systems (Yáñez-Arancibia *et al.*, 1985; Able, 2005; Cowan *et al.*, 2013; Hagy & Kemp, 2013).

The dominance analysis is a fundamental aspect in fish communities, since it allows to define the ecological importance of species through their interactions and habitats at a spatial-temporal scale (Sheridan & Minello, 2003; Yáñez-Arancibia *et al.,* 2007). In Sontecomapan Lagoon, previous fish research did not clearly analyze this aspect, so the system's natural variation of dominance reamained unknown, especially for species that are fishery resources or indicators of adverse environmental conditions (Aguirre-León *et al.*, 2010). The Importance Value Index (IVI) indicated that *D. rhombeus* and *C. aguadulce* were dominant in the system, the latter has been reported as *Cathorops melanopus* in most previous ichthyological studies in the Gulf of Mexico, and now it is recognized for Mexican coastal waters as a synonym of *C. aguadulce* (Marceniuk & Bentacour, 2008; Miller, 2009).

It is necessary to unify the numeric criteria upon which dominance is calculated throughout its latitudinal distribution and to report the specific fishing gear used to estimate it, however, each author has used different indicators and nets for this purpose. Martínez (1987) reported *D. rhombeus* as one of the most abundant species in the Sontecomapan system, while Castillo-Rivera *et al.* (2005), Pérez-Ponce (2007), Aguirre León *et al.* (2014) and Juárez Bautista (2015) indicate the dominance of *C. aguadulce* (*sensu lato Cathorops melanopus*) and *D. rhombeus* in several lagoon systems of the Gulf of Mexico. The same authors also reported *O. beta, C. spilopterus, C. parallelus., A. mitchilli, D. auratus, E. melanopterus, M. curema* and *A. lineatus* as dominant.

In other coastal States of the Gulf of Mexico, *C. aguadulce* has been reported as an abundant species in lagoons of Yucatan (Vega-Cendejas, 2004). Aguirre-León & Díaz-Ruiz (2006) found that *D. rhombeus* is also abundant in coastal systems from Campeche. Similarly, in other latitudes, *D. rhombeus*, is dominant in the Raya Lagoon in Venezuela (Ramírez, 1994). The wide distribution of these two species in coastal lagoons is due to the adaptive success that allows them to move, feed and reproduce within them (Cowan *et al.*, 2013).

Values for the H' and D indexes, showed similar spatial trends in the system and were generally higher on the connecting channel and middle part of the system where the tide influence and salinity is more pronounced, and temperature is warmer. The lagoon system has a geomorphology with a narrow marine inlet and a long tidal channel, this determines the exchange of water with the sea and species point of entry. In Chica-Grande lagoon in the north of Veracruz, Aguirre-León *et al.* (2014) observed similar behavior since it also has a sinuous tidal channel, and a tidal effect that allows the access of euryhaline and stenohaline species to the system. Other variables that influence diversity in coastal systems are depth, freshwater discharge and transparency associated with temporal precipitation. Also, in Pueblo Viejo Lagoon, Veracruz, Castillo-Rivera *et al.* (2005) established that the changes of fish diversity are determined by organic production and rainfall, which increases the availability of food and controls the immigration of species. In other tropical latitudes, it has also been established that the diversity and richness of the fish community is influenced by low transparency and changes in salinity that have a significant effect on their spatial-temporal distribution (Blaber, 1985; Cyrus & Blaber, 1987; Blaber *et al.*, 1995; Blaber, 2002). Spatially, evenness J' was opposite to H' and D. This trend was related to the low diversity near the marine inlet, however, the lowest values towards the system header were due to the abundance of species such as *C. aguadulce* and *D. rhombeus*, which behaved as dominant. Velázquez-Velázquez *et al.* (2008), Díaz-Ruiz *et al.* (2012) and Cowan *et al.* (2013) mention that evenness is inversely related to dominance, so that high values of dominance reveal low evenness values, as recorded for Sontecomapan.

There were no marked changes in the indexes between months, however, small H' changes were related to the evenness and dominance of the species in each month, as in February, 2015, where only *D. rhombeus* dominated and the average value of evenness was higher, in addition it was the month with the lowest average water temperature. Also, local changes in salinity and transparency influenced the monthly richness D of the system. Research from other lagoon systems of the Gulf of Mexico indicate that temporal values with highest diversity are also related to the input of nutrients from the rivers and an increase of primary production (Díaz-Ruiz *et al.*, 2003; Aguirre-León *et al.*, 2010; Arceo-Carranza *et al.*, 2010; Castillo-Rivera *et al.*, 2011).

The spatial variations of density and biomass were similar, with low values towards the marine inlet and higher values towards the head of the system. This trend was associated with the low diversity in the areas closest to the inlet, as noted in the previous section. In contrast, average weight showed an opposite trend, except in October, 2014 when values close to the marine inlet were higher due to the presence of rare big species such as S. *felis* (*sensu lato Ariopsis felis,* Marceniuk & Menezes, 2007), *S. plumieri* and *H. sabinus (sensu lato Dasyatis sabina,* Last *et al.*, 2016). Other features of the system that determine variation in abundance are, the spatial distribution of sediments (sandy in the sea mouth and silty in the header), the submerged vegetation (that functions as protection areas for growth and feeding), and spatial changes on physicochemical parameters (Martínez, 1987).

The highest average values of density, biomass and weight occurred in February and October, 2015 when *D. rhombeus* and *C. aguadulce* had the highest abundance due to their life cycles and the system's environmental characteristics. For Pueblo Viejo Lagoon, Castillo-Rivera *et al.* (2005) reported that lifecycles can be favored by organic production of the ecosystem when river discharge and runoff into the lagoon increase, providing allochthonous organic matter and nutrients to the system thus increasing the availability of food for consumer species. Several authors (Azevedo *et al.*, 2007; Aguirre-León *et al.*, 2010; Arceo-Carranza *et al.*, 2010; Lara-Domínguez *et al.*, 2011; Díaz-Ruiz *et al.*, 2006,

2012) have reported this seasonal pattern of abundance in the community related to environmental conditions and the seasonal migration of species linked to their lifecycles. In this sense, the species that contributed the most to the temporal changes in abundance at Sontecomapan Lagoon were *D. rhombeus, C. aguadulce, D. auratus, C. spilopterus, S. felis* and *E. plumieri*. These are mostly estuarine or euryhaline species which develop much of their life cycle in the lagoon system since both juveniles and pre-adult sizes were collected during this research. Particularly, the stripped mojarra *E. plumieri* is a marine species with commercial value in Sontecomapan and is often sold and consumed locally in the nearby town of Catemaco (personal observation from authors). However, biological and ecological knowledge of this and other fishery resources such as snooks of the genus *Centropomus* is scarce. Therefore, it is necessary to gather more information about the distribution, abundance and seasonality of fish community as soon as possible.

CCA results allowed to identify two groups of species related to the environmental system gradients. The first group included species of three community inhabitants (ME, E and MES), some of them were scarce and distributed in the first 4 sampling sites (stations 1 to 4). These sites correspond to the connection inlet and tidal channel of the system, where salinity, dissolved oxygen, transparency and depth were greater, thus the species *S. plumieri*, *S. spengleri* and *S. foetens* were collected in this habitat. The rest of this group's species had greater abundance and were associated mainly to sites 5, 6 and 7 with smaller values of the physicochemical variables previously indicated. A second group of species was closely related to higher salinity changes during the months of study, which tolerate conditions of low salinity, higher temperature, lower depth and transparency in areas of fluvial influence. There are abundant euryhaline and estuarine species such as *S. felis, C. spilopterus, C. aguadulce, D. rhombeus, D. auratus* and *E. plumieri*; freshwater species like *M. urophtalmus* were present as well. Both groups species use the system for protection, growth, feeding or maturation purposes in the "Nortes" season. Castillo *et al.* (2005), Mendoza *et al.* (2009), Arceo-Carranza *et al.* (2010), Díaz-Ruiz *et al.* (2012), Cowan *et al.* (2013) and Aguirre-León *et al.* (2014) reported a similar behavior of fish groups as observed in Sontecomapan for different costal lagoons of Mexico and the United States. It is known that in coastal lagoons fish use multiple habitats as a series of complex responses to environmental factors thus causing variations in species abundance and distribution. Seasonal migrations within and outside the system, as well as resident populations determined by environmental variables also cause these variations (Able, 2005; Cowan *et al.*, 2013; Velázquez-Velázquez *et al.*, 2008; Aguirre- León *et al.*, 2014).

Our analysis showed that the major changes in the distribution and composition of the fish community, are highly associated with changes in the hydrological behavior of the system that determines the variation of physicochemical variables (transparency, bottom salinity, pH, temperature and dissolved oxygen). In turn, this explains the distribution and migration patterns of fish during their lifecycles. Similar studies have emphasized that the variation of environmental factors such as salinity, temperature and depth determined by the rainy season and the entry of seawater have a significant effect on the distribution and abundance of fish in coastal ecosystems (Cyrus & Blaber, 1992; Barletta *et al.*, 2005; Pombo *et al.*, 2005; Cowan *et al.*, 2008; Velázquez-Velázquez *et al.*, 2008). Biological factors also play an important role in the distribution and abundance of species, in terms of availability changes in food and space during the year. The lifecycle of most fish in the coastal systems is typically estuarine, with seasonal migrations that concur with periods of high productivity and food availability (Díaz-Ruiz *et al.*, 2012; Aguirre-León *et al.*, 2014).

In Sontecomapan Lagoon the fish have adapted migration strategies and routes for the use of lagoon-estuarine habitats that offer protection and food for juveniles, facilitate spawning, and provide feeding of adult individuals both of marine and estuarine species linked to spatial-temporal changes of the environmental conditions. This latter has also been analyzed by Able (2005), Guevara *et al.* (2007), Kopp *et al.* (2007), Díaz-Ruiz *et al.* (2004, 2012), Zárate-Hernández *et al.* (2012) and Cowan *et al.* (2013). At the same time, biological information on species feeding, reproduction and spawning is required to know its relationship with the fish distribution and abundance in the lagoon (Aguirre-León *et al.*, 2010). Therefore, knowledge of the ecological structure of fish community and their interaction with environmental factors in the Sontecomapan lagoon is relevant, since there is no previous record of medium or long term studies in this system. Consequently, the information offered in this study is a starting point for further evaluations and a tool for monitoring the environmental status and ecological behavior with the purpose of preserving its biodiversity and protection status as an important area for breeding, feeding, protection and/or reproduction of fishery resources in this Biosphere Reserve that is also a Ramsar site.

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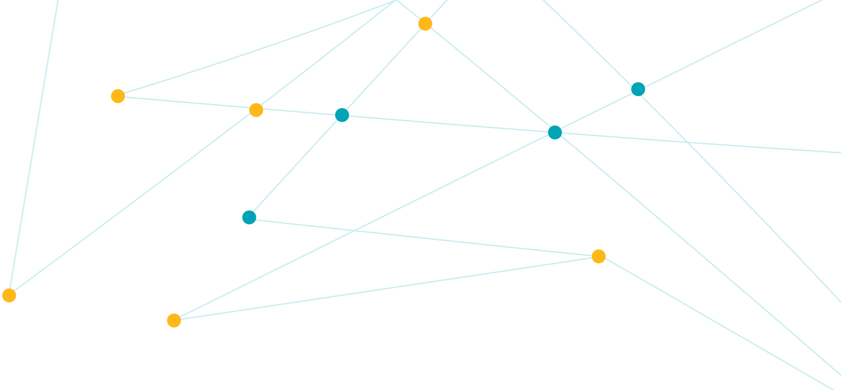

PART 7

Societal aspects

Organic and inorganic contamination

Evaluation anthropic fecal contamination through enteropathogenic genera identification in Sontecomapan coastal lagoon, Veracruz

Ruth Soto-Castor & Alfonso Esquivel-Herrera

Metal pollution in aquatic ecosystems: a case study on Sontecomapan coastal lagoon (State of Veracruz) considered as a reference of pristine ecosystems

Aurora González-Fierro & Guadalupe Ponce-Vélez

Conservation and restoration

Changing trends in mangrove the forest cover of the Sontecomapan Lagoon System, Veracruz, with Landsat® satellite images

José Luis Hernández-Gaona, Javier Aldeco-Ramírez & Iván Ernesto Roldán-Aragón

Evaluation anthropic fecal contamination through enteropathogenic genera identification in Sontecomapan coastal lagoon, Veracruz

Ruth Soto-Castor* & Alfonso Esquivel-Herrera*

ABSTRACT: Sontecomapan Lagoon is a coastal system located in the Gulf of Mexico which during the late years of the 20th century was equally considered either as a referent of a pristine site or as a man-affected system. Adding to this uncertainty, there are no published references on the microbiological quality of its water. With this purpose, fecal pollution was assessed through the spatial and temporal distributions of total fecal coliforms and enteropathogenic bacteria genera, which were analysed through four surveys (March and September, 2009, January and May, 2010), from ten sampling stations. Total heterotrophic bacteria were simultaneously determined, as well as some physical and chemical variables, and chlorophyll *a,* to assess the prevailing environmental conditions. Fecal coliform densities surpass the permissible levels and the presence of enteropathogenic genera *Salmonella* and *Shigella* poses a further alert. Multivariate analysis (cluster analysis and principal component analysis) highlighted the relation between the abundance and distribution of fecal coliform and enteropathogenic bacteria to the influence of human settlements and livestock husbandry, as well as the lack of correlation between total heterotrophic bacteria and coliform bacteria.

Keywords: Coliform bacteria, Gulf of México heterotrophic bacteria, *Salmonella*, *Shigella*.

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Introduction

Coastal lagoons are transitional zones between continent and sea, and they are considered areas of high productivity (Contreras, 1985). One of the major threats to these coastal waters is that they are subject to the impact from the surrounding communities located at their periphery; this way Lagoons suffer the impact of human discharges mainly from domestic waters and fecal waste from livestock (López-Portillo *et al.*, 2017).

This kind of studies, in the Sontecomapan Lagoon are scarce, even more it has not been emphasized the importance of detecting the enterobacterial genera presence, especially pathogens such as *Salmonella* and *Shigella* (Clayton *et al.*, 2016). The genera mentioned cause diseases such as salmonellosis, typhoid fever, bacillary dysentery and other infections acquired through water or shellfish intake, but other, like *Klebsiella*, also affect the respiratory and urinary tracts, putting human health at risk when in contact with polluted water (Chow *et al.*, 2013). Even though some of these genera may also inhabit in the native fauna of these ecosystems as waterfowl and reptiles, these microorganisms also indicate the presence of ill humans who live close to these ecosystems, (Mather *et al.*, 2016).

The importance of finding these microorganisms in waterbodies is that edible species are extracted from these systems to be commercialized and consumed locally and regionally, putting human health at risk when organisms such as fish and crustaceans may have fecal bacterial pathogens (Clayton *et al.*, 2016).

Coliform bacteria have been used for a long time as indicators of fecal contamination (Fecal Indicator Bacteria, FIB) of water and their detection indicates that human or animal excreta have been discharged to urban effluents, or to water discharges from agriculture and from livestock raising activities. Finally, these discharges arrive to coastal lagoons and the sea (Mallin *et al.*, 2000; Sidhu *et al.*, 2012).

Enterobacteria survival and growth depend on the environmental conditions at the discharge site and on the biotop where they finally arrive; for example, mangroves are rich in organic matter and nutrients that sustain the growth of the native and allochthonous bacterial communities in the lagoon ecosystems, including those of enteric origin (Cho *et al.*, 2010, Stumpf *et al.*, 2010, Vasco *et al.*, 2015).

Because of this, the purpose of this study is: a) to analyze the temporal and spatial distribution of heterotrophic and coliform bacteria, and the presence of enterobacterial genera of anthropic origin at Sontecomapan, b) to determine the relation of physicochemical and environmental variables with the FIB concentrations.

Material and Methods

Study area and sampling points

Sontecomapan is a coastal lagoon located at 18°30’ N to 18°34’ N and 94°47’ to 95°11’ W, covering about 891 ha; freshwater inflow comes from three rivers (López-Portillo *et al.*, 2017) (Fig. 1).

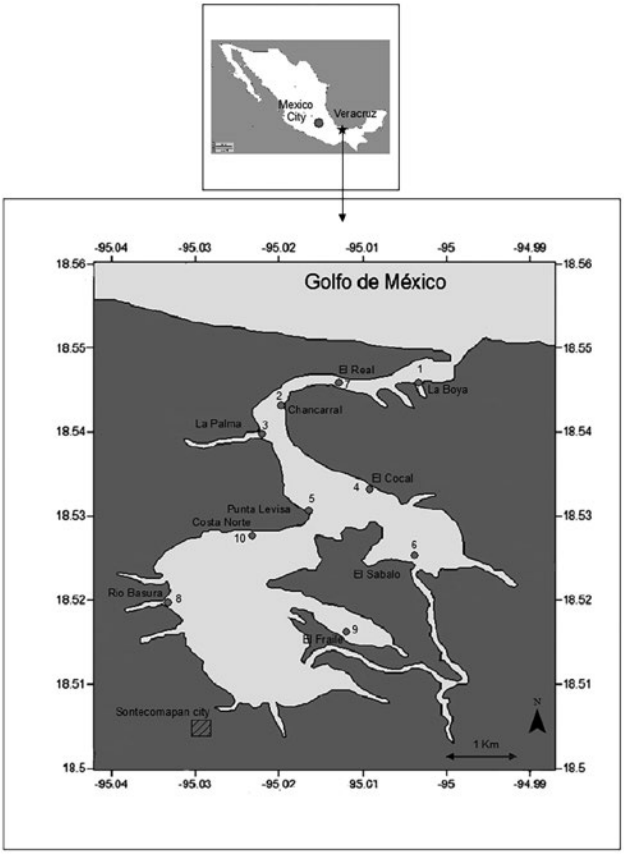


Figure 1. Location of the sampling stations at Sontecomapan lagoon.

This lagoon is connected to the sea through a 5.5 m-depth and 137 m-width inlet delimited by a sandy bar known as Barra de Sontecomapan. The lagoon is shallow with a mean depth of 1.67 m (López-Portillo *et al.*, 2017). The mangroves and wetlands bordering this lagoon are part of the wildlife protection zone known as Reserva de la Biosfera de los Tuxtlas (Calva *et al.*, 2005).

Coastal ecosystems in the Gulf of Mexico, particularly coastal lagoons, are submitted to the influence of three climatic seasons: north wind, dry, and rainy (Contreras, 1985). Annual rainfall ranges between 3000 and 4000 mm. The region is considered as isothermal with maximum temperatures of 26 °C during May and minima of 22 °C in January. Generally, water turbidity is high, with a mean transparency of 0.60 m for the Secchi disk. Water surface temperatures have an average of 24 °C and no vertical thermic difference is noted along the water column. The southern portion of the lagoon is mesohaline (salinity from 5 to 18 PSU), while the central portion is considered as polyhaline (25-30 PSU); the mouth shows euhaline values (30-40 PSU) (Contreras, 1985).

Ten stations were sampled at subsurface depth of 0.3 m and at 0.3 m above the water-sediment interface (Fig. 1). The choice of the sampling stations was made considering representative locations of the diverse ecological and hydrological features as we as inflow from rivers, streams and the influence of domestic, agricultural or aquaculture discharges (Table 1).

Table 1. Sampling points names, station number and coordinates

Sampling Point	Coordinates
La Boya	N 18°33’02.6”
1	W 94°59’26.8”
El Chancarral	N 18°32’53.2”
2	W 95°00’52”
La Palma	N 18°32’21”
3	W 95°01’02.1”
El Cocal	N 18°32’21”
4	W 95°00’24.8”
Punta Levisa	N 18°32’10”
5	W 95°00’42.3”
El Sabalo	N 18°32’09.8”
6	W 95°00’49.2”
El Real	N 18°33’19.3”
7	W 95°00’51.7”
Rìo Basura	N 18°31’41.3”
8	W 95°02’68”
El Fraile	N 18°30’51.7”
9	W 95°00’39.5”
Costa Norte	N 18°32’49”
10	W 95°01’20.7”

Sampling was conducted in dry and north wind seasons (March, 2009 and January, 2010, respectively) and in rainy time (September, 2009 and June, 2010), covering the three climatic seasons for this region (Contreras, 1985).

At each station, one sample of surface and one of bottom water were taken with a Van Dorn bottle. Subsamples were collected into sterile 250 ml flasks, which had been previously cleaned and rinsed with 1% sodium thiosulfate solution to inhibit residual chlorine. The samples for bacteriological analysis were kept in the dark at 4 °C and processed within 6 hours.

Physical and chemical data were simultaneously measured at each sampling station, on the four surveys and the results are described later in this volume (Esquivel & Soto).

For the direct count of total bacteria, subsamples were immediately drawn from the 250 ml flasks into 20 ml sterile amber flasks and preserved with 0.22 μm-filtered formaldehyde to a final concentration of 2%, and then stored in the dark at 4 °C. Water samples were filtered on black 0.22 μm pore size polycarbonate membranes (25 mm diameter) and then stained with the fluorochrome 4',6-diamidine-2-phenylindol (DAPI) for 15 min (Porter & Feig, 1980). Bacteria were counted with epifluorescence microscopy with an Olympus BIMAX50 microscope provided with a 50W mercury lamp and a narrow-band excitation filter (365/366 nm). A minimum of 400 cells was counted in each sample (Kepner & Pratt, 1994).

Total and fecal (thermotolerant) coliform bacteria and *Escherichia coli* were estimated by the Most Probable Number (MPN) method in multiple fermentation tubes (Hussong *et al.*, 1981; APHA AWWA & WPCF, 1992). Cellular morphology was described from Gram-stained slides, to confirm the presence of gram-negative bacilli and comma-shaped bacilli. The total coliform bacteria were cultured for 24 hours at 37.0 °C in lactose broth medium and Brilliant Green Bile Lactose broth and the fecal coliform bacteria were cultured for 48 hours in both broths at 44.0 °C. Biochemical identification of enteropathogenic bacteria was carried out based on Farmer *et al.* (2007). Biochemical confirmation of presumptive isolates of enteropathogenic genera was performed for pure cultures using the bioMérieux API 20E and API 20NE kits (L' Étoile, France) for biochemical identification of fermenting facultative and non-fermenting Gram-negative strains, respectively. As a further test, cytochrome-oxydase and catalasa were applied on the pure cultures for growth for a period of 16 to 20 hours at 37 °C. The results were run through bioMérieux's APILab Plus V.3.0 to further identify each isolate.

Statistical analysis was performed on environmental data after normalization; all data (not averages) were entered into the analysis. Multivariate analysis (cluster analysis and principal component analysis) were applied to data for their clustering and ordination to analyse the association between environmental conditions. Bacterial abundance and distribution data were transformed to z for their cluster analysis (Legendre & Legendre,

1984) based on Euclidean distance and Ward's aggregation algorithm for clustering by sampling site; values transformed to $\log_{10}$ (X+1) were employed for ordination through principal component analysis in order to analyse the links between environmental conditions (Pielou, 1984). For all these analysis, Statistica for Windows 7 (StatSoft, Tulsa, Ok.) was employed.

Results

Environmental conditions

During the surveys here considered, the lagoon was characterized by: (i) warm water with a minimum of 24 °C and a maximum of 33 °C, (ii) Secchi disk depth ranged from 0.18 to 2.10 m, (iii) pH from 6.4 to 7.8, and iv) salinity from 0.1 to 36.0 PSU (Esquivel & Soto, this volume). Nitrate and nitrite showed low values for the four surveys; the highest concentrations were detected in September with 10.8 mg l^{-1} and 0.06 mg l^{-1}, respectively. Ammonia reached its highest concentrations in June, with 3.57 mg l^{-1}. Total phosphorus ranged from undetectable to 2.71 mg l^{-1} in January, while soluble reactive phosphorus had a minimum of 0.80 mg l^{-1} in January and a maximum of 26.84 mg l^{-1} in June. Dissolved oxygen concentrations ranged from 0 mg l^{-1} in March to a maximum of 9.42 mg l^{-1} in June, at the onset and end of the dry season, respectively. Chlorophyll *a* had average values of 0.98 μg L^{-1} in March 2009, 0.38 μg L^{-1} in September 2009 and 0.73 μg L^{-1} in January, 2010.

Total bacteria

During the period considered, total bacteria densities ranged from 2.4 x 10^4 (surface, March and September, 2009) to 1.1 x 10^6 cells mL^{-1} (bottom, March, 2009). The average values ranged from 6.0 x 10^4 (September, 2009, surface) to 5.9 x 10^5 cells mL^{-1} (June, 2010, bottom).

Coliform

Total coliform ranged from 3 MPN 100 mL^{-1} (station 10, Costa Norte in March, 2009) to 1200 MPN 100 mL^{-1} (at various surface sampling points between 2009 and 2010). Mean total coliform were 298 MPN 100 mL^{-1} at surface, and 409 MPN 100 mL^{-1} at bottom water. Total coliform average values ranged from 85 MPN 100 mL^{-1} (January, 2010) to 611 MPN 100 mL^{-1} (September, 2009) for surface, and from 67 MPN 100 mL^{-1} (January, 2010) to 762 MPN 100 mL^{-1} (March, 2009) for bottom water (Table 2, Figs. 2 and 3).

Table 2. Mean and range for the bacterial population (with units) at the surface level (0.3 m depth) and bottom during the four surveys

Surface	March, 2009	September, 2009	January, 2010	June, 2010
Total coliform MPN 100 mL^{-1}	392 (3-1200)	611 (100-1200)	85 (14-250)	103 (4-800)
Fecal coliform MPN 100 mL^{-1}	646 (26-1200)	534 (42-1200)	455 (25-1200)	30 (3-110)
Total bacteria cells mL^{-1}	6.4×10^4 (2.4×10^4-1.6×10^5)	6.0×10^4 (2.4×10^4-1.6×10^5)	2.0×10^5 (3.3×10^4-5.0×10^5)	5.0×10^5 (8.0×10^4-9.0×10^5)
Bottom	March, 2009	September, 2009	January, 2010	June, 2010
Total coliform MPN 100 mL^{-1}	762 (46-1200)	624 (67-1200)	67 (14-175)	18 (4-1000)
Fecal coliform MPN 100 mL^{-1}	590 (33-1200)	336 (21-1200)	352 (30-1200)	207 (7-1000)
Total bacteria cells mL^{-1}	4.1×10^5 (9.5×10^5-1.1×10^6)	9.6×10^4 (2.6×10^4-3.7×10^5)	2.0×10^5 (4.1×10^4-4.1×10^5)	5.9×10^5 (2.6×10^5-9.0×10^5)

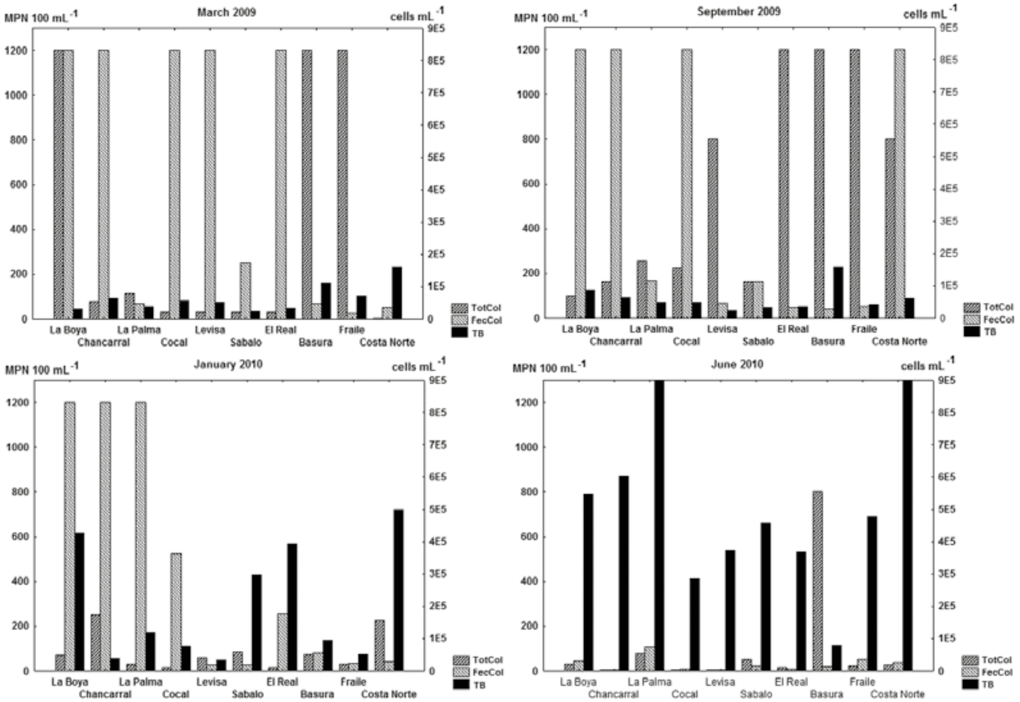
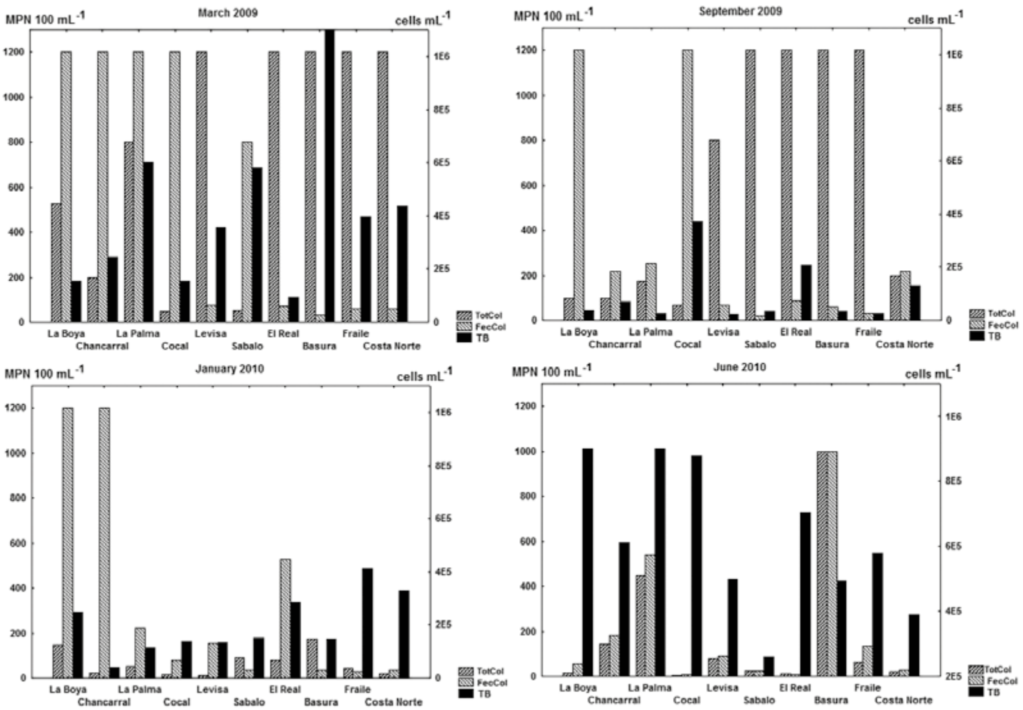


Figure 2. Variations of the total (TotCol) and fecal coliforms (FecCol) concentrations (left scale) and the total bacteria (TB) concentrations (right scale) at the surface water (0.3 m depth) of each station.


March 2009

September 2009

January 2010

June 2010

Figure 3. Variations of the total (TotCol) and fecal coliforms (FecCol) concentrations (left scale) and the total bacteria (TB) concentrations (right scale) at the bottom level of each station.

Fecal coliform ranged from 3 MPN 100 mL^{-1} (station 5, Levisa; June, 2010) to 1200 MPN 100 mL^{-1} (various surface and bottom water samples, except from June, 2010). Means of fecal coliform were 417 MPN 100 mL^{-1} at surface and 371 MPN 100 mL^{-1} at bottom water. Mean of fecal coliforms at surface ranged from 30 MPN 100 mL^{-1} (June, 2010) to 646 MPN 100 mL^{-1} (March, 2009) and from 207 MPN 100 mL^{-1} (June, 2010) to 590 MPN 100 mL^{-1} (March, 2009) (Table 2, Figs. 2 and 3). A lack of correlation was observed between total bacterial concentrations and coliform, as well as between total and fecal coliform.

Enterobacteriaceae Identification

The highest number and diversity of enterobacterial isolates from surface and bottom water samples was obtained in January, 2010 while the lowest number and diversity was found in the sampling for June, 2010 (Table 3). Isolates of Enterobacteria genera such as *Enterobacter*, *Klebsiella*, *Escherichia*, *Serratia*, *Citrobacter*, and *Proteus* (some of which are part of coliforms) were obtained in all surveys; they were found in the Basura stream and were found also near the settlements at Chancarral and El Real. The fecal indicator *Escherichia* was isolated in all the seasons, both in the samples of surface water and bottom water. *Salmonella* was mainly detected in September and June, corresponding to the rainy season, while the enteropathogenic genus *Shigella* appeared in all the surveys. It must be emphasized that the latter genus is associated to ill humans, but *Salmonella* can originate from human or non-human sources. The greater abundance and genera diversity was associated to the sampling points influenced by streams (Basura and La Palma), and shallow areas with contribution from livestock and poultry excreta (El Chancarral and Fraile) and domestic waste (El Real) (Figs. 4 and 5). The highest number of enterobacterial isolates was obtained from the bottom water samples, except for January (Figs. 4 and 5).

Table 3. Number of bacterial isolates by survey, Sontecomapan, Veracruz

March 2009	September 2009	January 2010	June 2010
69	47	92	34

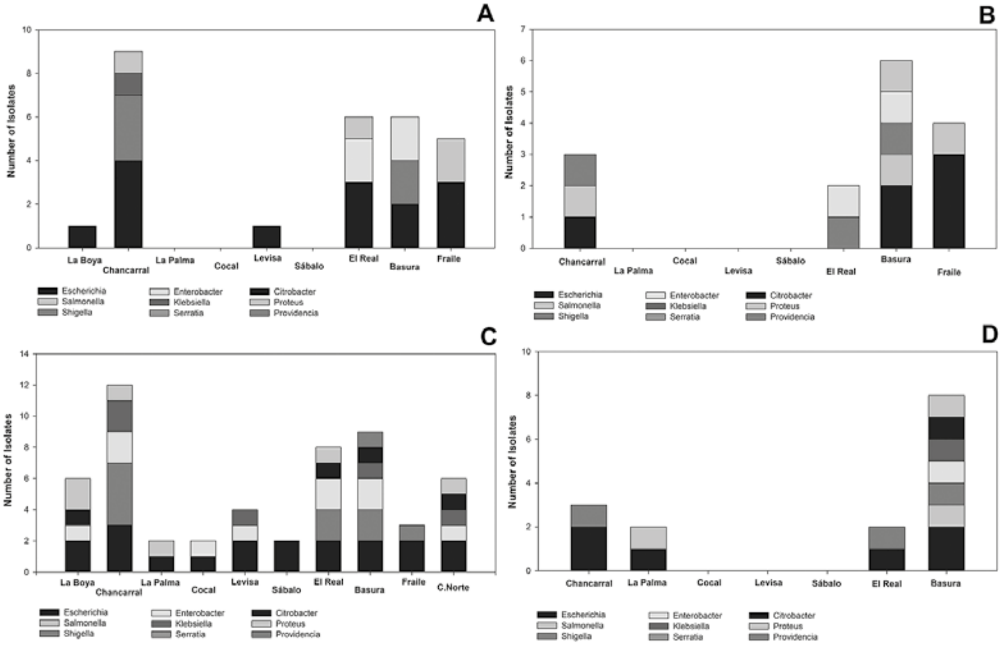


Figure 4. Generic composition of *Enterobacteriaceae* per sampling point and survey at surface: A. March, 2009 B. September, 2010 C. January, 2010 D. June, 2010

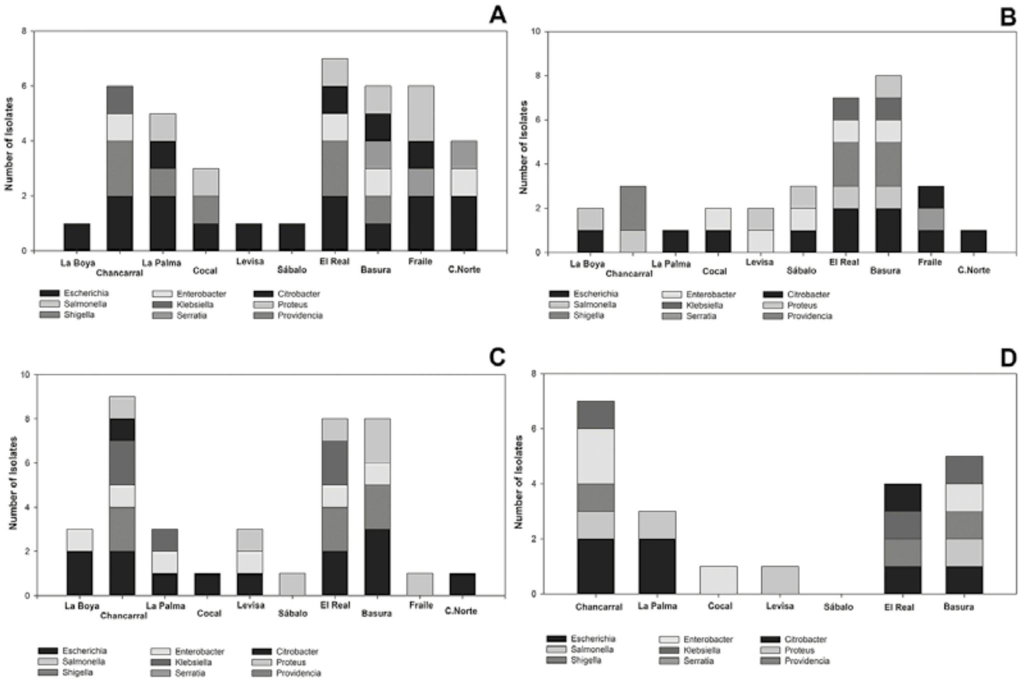


Figure 5. Generic composition of *Enterobacteriaceae* per sampling point and survey at bottom: A. March, 2009 B. September, 2010 C. January, 2010 D. June, 2010

In terms of abundance by survey, the dry and Nortes seasons, showed the highest abundance with 69 and 92 isolates respectively, whereas rainy season surveys, had 47 and 34 bacterial isolates (Table 3). In relation to diversity, surface water samples showed lower genus diversity than bottom water samples. *Salmonella* and *Shigella* genera were also present (Figs. 4 and 5).

Multivariate Analysis

A cluster analysis from abiotic and biotic variables showed two main trends in relation with the station location and the sampling depth. First, no stratification was detected for any sampling point or season. Second, sampling stations differed from the north and eastern portion to those located in the south and western, as shown by the two major clusters. These further were subdivided into lesser subclusters. One of them comprised the sampling stations La Boya (1), Chancarral (2), and El Real (7), situated along the channel connecting to the sea. The second cluster included sampling points Cocal (4), Levisa (5) and Sábalo (6), at the intermediate part of the lagoon. The sampling points at the inner part of the lagoon, Basura (8), Fraile (9) and Costa Norte (10) fell into another cluster. These subclusters are equivalent to the euhaline, polyhaline and mesohaline zones found by Contreras (1985), respectively. The stream at La Palma clustered apart because it is influenced by freshwater (Fig. 6).

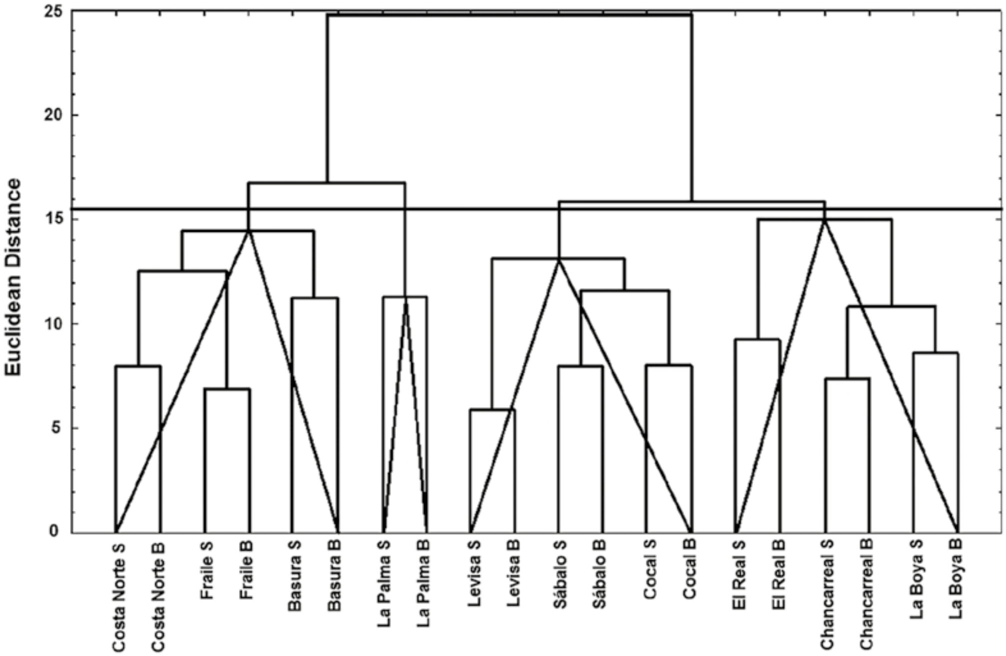


Figure 6. Dendrogram of Euclidean distance through Ward's method showing similarity between sampling stations based on the 11 parameters studied. Samples were noted S and B for surface and bottom, respectively.

PCA was performed on the independent data sets (11 variables, 10 stations, and 4 months). The first two eigenvalues accounted for 52% of the total variability. Survey clustering was based on the PCA, and resulted as follows: June, 2010 (early rainy season) was the survey that most differed from the other, with positive values for the first ordination axis. March 2009 and January, 2010 (dry and north wind season) formed a subgroup with negative values for the first and second ordination axes. September, 2009 (full rainy season) had the most negative values for the first ordination axis, but positive for the second (Fig. 7).

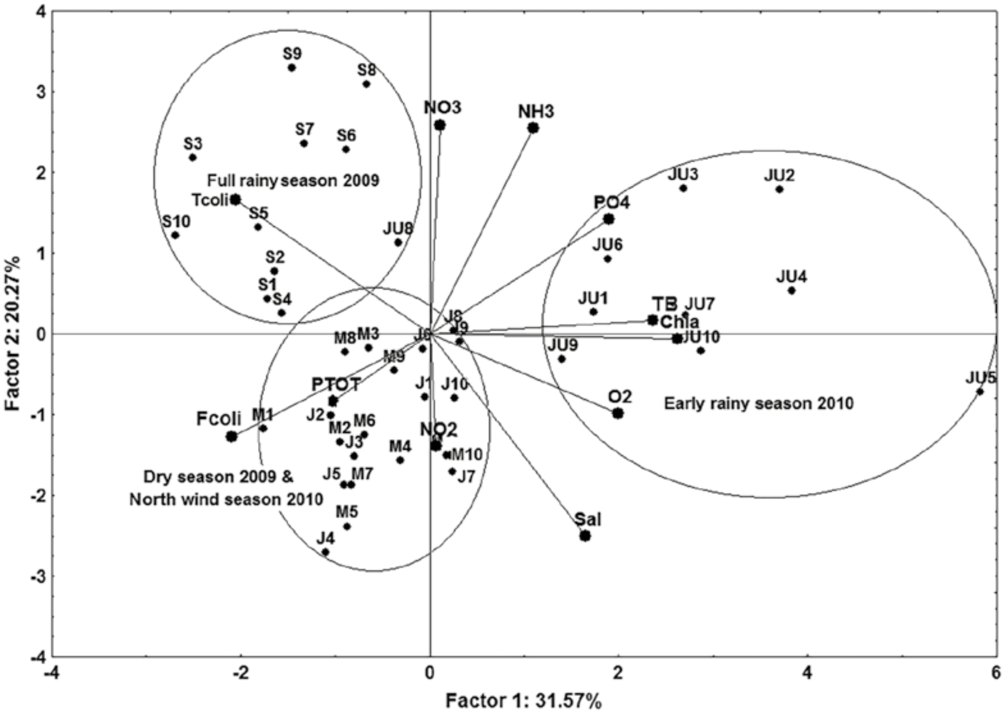


Figure 7. Principal component analysis (PCA) of the two first axes based on 11 studied parameters from all samplings. Eigenvalues for each PCA axe are reported. Ordination by parameters and sampling points. Chla chlorophyll *a*, Fcoli fecal coliform, NH3 Total ammonia, NO2 Nitrite, NO3 nitrate, PO4 soluble reactive phosphorus, PTOT total phosphorus, TB total bacteria, Tcoli total coliform.

PCA, by variables (Fig. 7), shows that the main ordination axis closely correlates at its positive part with chlorophyll *a,* and total heterotrophic bacteria and, to a lesser degree, with dissolved oxygen, salinity and reactive soluble phosphorus, which occurred with higher values during the early rainy season of 2010. At the negative part of this axis occurred the variables linked to sanitary indicator bacteria (total coliforms and fecal coliforms); a correlation analysis was run which determined that heterotrophic bacteria densities were directly correlated to chlorophyll *a* concentration ($p<0.001$).

Salinity negatively correlates with the full rainy season of 2009 (September), which presented the lowest salinities. The second ordination axis has nitrate and total ammonia at its positive end and nitrite at the negative. Total coliform correlate to the same rainy season survey of 2009, when they were more abundant, whereas fecal coliform and total phosphorus correlate to the north wind season of 2010 (January) and the dry season of 2009 (March); these last two surveys are located at the negative part of the second ordination axis (Fig. 7).

Discussion

Even if Sontecomapan is a relatively small coastal lagoon, due to its morphometry it presents a complex behaviour which, in a spatial sense, allows the regionalization that appears in the cluster analysis (Fig. 5). There is another overlaid variation pattern which results from seasonal variation, especially between the dry and rainy seasons. This affects the distribution and survival of native planktonic bacteria, as well allochthonous bacteria, including those of fecal origin.

Total bacterial counts were lower than would be expected for a tropical coastal lagoon, as they varied between 10^4 and 10^6 cells ml^{-1}, values which correspond to oligotrophic tropical waters and lie below the expected range for coastal waters (10^6 to 10^7 cells ml^{-1}) (Schut *et al.*, 1997), and especially for surface water (Weise, 2004). At a coastal lagoon, a complex interaction occurs between environmental factors which determine the fate of microbial communities, including their flushing by seaward currents, as occurs in the so-called *displacement-dominated lagoons* (López-Portillo *et al.*, 2017), which may explain these low bacterial densities (Evanson & Ambrose, 2006; Surbeck *et al.*, 2006; Ulrich *et al.*, 2016).

Even if total bacterial counts made with direct count (Porter y Feig, 1980) were low, except for surface water during June, 2010; fecal (thermotolerant) coliform bacteria exceeded USA-EPA's (1988) recommended maximum permissible values for bathing water of 126 NMP 100 mL^{-1} and of 14 NMP 100 mL^{-1}, with no more than 10% of the samples exceeding 43 NMP 100 mL^{-1} for shellfish extraction or harvesting. They also surpassed the maximum allowable value for fecal coliform of 240 NMP 100 mL^{-1} for services involving direct contact with water according to Mexican legislation, except for surface and bottom water in June, 2010 (NOM-003-SEMARNAT-1997).

Coliform bacteria have become the standard for fecal pollution assessment because they indicate the health risks associated to food handling, seafood and water (Balière *et al.*, 2016). However, water quality diagnostics are enriched when identification of enterobacterial genera, mainly pathogens such as *Salmonella* and *Shigella*, is considered. Enteric bacteria were formerly believed to have a limited ability for surviving outside the digestive tract but, as they have been isolated from environmental samples with relative ease, this implies that they have adapted to outside environments. In the case of *E. coli*, it has been determined that it can survive in the dark for up to six times more than other microorganisms in river water, which may explain its constant presence in the Sontecomapan water (Avery *et al.*, 2004; Sinton *et al.*, 2007).

Our study revealed that fecal coliforms were more abundant in March, in what appears to be the marine-influenced part of the lagoon, comprising La Boya (1), Chancarral

(2), La Palma (3) and Cocal (4) (Fig. 1), but this could be explained by considering the upstream sources at La Palma which also contributed to increase inorganic nitrogen and phosphorus (Esquivel & Soto, this volume). Total bacteria were less abundant in September, when large rainfall and low-tide resulted in the seaward transport of planktonic bacteria, but total and fecal coliform appeared at high densities, because of rainfall-associated runoff. On other occasions, the highest bacterial counts appeared at the stations that were most protected from current disturbance or at the stations under the influence of streams or rivers inflowing from human settlements (Aslan-Yilmaz *et al.*, 2004; Ahmed *et al.*, 2016). Thus, fecal coliforms were abundant in January at the northern part of the lagoon, comprising the communication mouth to the sea and the channel, while total coliforms were scarce at most sampling stations. The river inputs explained the highest fecal coliform densities reported from La Palma (3) and El Fraile (9) (Fig. 1). This same trend has been described by other works, when the natural inputs of rainwater runoff add to the freshwater plumes incoming from rivers and contribute to fecal contamination, thus temporal and spatial variation should also be considered (Geldreich, 1996; Ulrich *et al.*, 2016). Another enteric bacteria source comes from drainage and runoff that collect debris from farming facilities, impacting the water quality with fecal pollution (Duse *et al.*, 2016; Vasco *et al.*, 2015).

The present results confirmed that the major contribution of fecal contamination comes from human settlements and farms in the lagoon's periphery; fecal wastes from poultry, pigs or cattle are discharged mainly at the stations of El Real and Chancarral. This poses a threat due to the improper use of antibiotics, such as automedication or improper livestock husbandry practices which result in bacterial resistance to them. *Salmonella* has been isolated from animals and agricultural lands of the United Kingdom and the United States, finding multidrug resistance plasmids to different antibiotics (Hancock *et al.*, 2000; Helms *et al.*, 2005; Leekitcharoenphon *et al.*, 2016).

It should also be mentioned that other natural sources for fecal contamination into the lagoon are waterfowl excreta that reach the coastal environments and arrive to this place to feed, reproduce and nest. The importance of these birds arrival is that some of the enterobacterial genera of the human digestive tract have also been isolated from the intestine of these wild-type birds that defecate in the mangrove. There are reports since the 1950s that *Salmonella enterica* serovar *typhimurium* has been isolated from different bird types that inhabit and travel seasonally throughout North America, Asia and Australia, leading to the spread of these microorganisms across all continents (Hernández *et al.*, 2012; Giovannini *et al.*, 2013; Asakawa *et al.*, 2014).

While the prevailing environmental conditions of the estuarine environment can produce a die-off of coliform bacteria, as would be expected for a good indicator of fecal pollution, there is a continuous input of them. This situation was observed at sampling points such as El Real, Chancarral, and the lagoon mouth, where septic tanks are commonly used, however, because of their improper design or maintenance, they frequently leak or overflow. Several studies have mentioned that another source of fecal contamination besides pluvial runoff water, are dead birds and small mammals which are dragged into water bodies (Ahmed *et al.*, 2008; Simmons *et al.*, 2008).

The presence of enteropathogenic genera in coastal water bodies, like Sontecomapan, is indicator of their survival in the environment, even if conditions and are not the optimal for their growth. These microorganisms have adapted to changing conditions which have allowed them to survive or even thrive in adverse environments. This has been confirmed by *Salmonella enterica* and *E. coli* survival models performed under laboratory conditions where it was found that at least a few cells can survive under desiccating conditions (Koyama *et al.*, 2017).

Likewise, *Shigella, Klebsiella* and *Enterobacter*, which were isolated at some of our collection sites, have been recovered from aquatic environments, soil and wastewater and are therefore able to survive in conditions that require only minimal water and energy sources in natural environments. Their sanitary importance is that they can cause infections in the digestive and urinary tracts and meninges, so that the pollution of aquatic systems by sick humans or animals excreta poses a health risk to whoever comes in contact with polluted water (Abbott, 2007).

In soil studies, it was found that *E. coli* developed a mechanism known as soil-persistence that allows it to survive in soil, and this means that this bacteria is highly versatile for metabolizing some substrates at environmental conditions similar to those presented in this work. The ability of *E. coli* to live and grow for long periods out from its host's gut poses some doubt concerning its adequacy as a good indicator of water quality. It is believed that the ability of *E. coli* for surviving stress conditions is due to the regulatory gene's response *Rpo S* (Gayán *et al.*, 2016; Somorin *et al.*, 2016).

Resistance to oxidative stress, osmotic stress and desiccation have also been studied in environmental strains of *Salmonella* (Shah *et al.*, 2012; Robbe-Saule *et al.*, 2003) and *Citrobacter* (Dong *et al.*, 2009). In Sontecomapan these genera occurred in the stations Basura, Chancarral, La Boya and Fraile, where desiccative conditions could occur due to their shallow condition.

The composition of the microbial community is influenced by the environmental changes, mainly at the shallow or circulation-restricted areas of lagoons (Crabill *et al.*, 1999; Conley, 2000; Dorner *et al.*, 2007), where water condition may rapidly deteriorate (Newton & Mudge, 2003). Excreted enteric bacteria are confronted by a highly competitive environment which is less nutrient-rich than the digestive tract of their hosts. This has become a reason why the adaptive mechanisms of these microorganisms may have evolved, enabling their survival when they are out from their normal habitat (Savageau, 1983). Thus, they may survive or even opportunistically proliferate at coastal environments when provided with adequate nutrients. In Sontecomapan these latter come from continuous discharges of domestic, livestock, or agricultural wastewater and waterfowl droppings.

In addition, the above also reflects that the species that make up bacterial assemblages of enteric origin, colonize coastal aquatic environments by taking advantage of the nutrient richness and other environmental conditions that favour their reproduction and growth. Finally, the presence of bacterial communities of enteric origin in water and sediments is also sustained by their continuous discharge and the persistence of bacterial communities in to this lagoon system (Muhammad *et al.*, 2012).

Conclusions

Total bacteria densities positively correlated to chlorophyll *a* concentrations while total and fecal coliform distribution and abundance were related to runoff from the surrounding human settlements. Thus, there was no correlation among total bacteria and coliform.

Sontecomapan is fecally polluted, as determined from total and fecal coliforms, which surpass maxima permissible values for services involving direct human contact and shellfish harvesting or extraction.

The highest coliform counts appeared during the rainy season, due to runoff from the upper parts of the catchment basin, with lesser values during the dry season.

Potentially pathogenic enteric bacteria were detected, such as those from the *Salmonella* and *Shigella* genera.

The presence of human settlements and human-related activities like cattle and poultry husbandry have an important contribution to fecal pollution in Sontecomapan, but wildlife waterfowl and mammals also play a role.

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Metal pollution in aquatic ecosystems: a case study on Sontecomapan coastal lagoon (State of Veracruz) considered as a reference of pristine ecosystems

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Abstract: The presence of Cr, Ni, Zn, Cu, Co, Cd and Pb in sediments, submerged vegetation and bivalve mollusks on the Sontecomapan Lagoon, Veracruz during 1991-1992 were evaluated as well as their physicochemical and sedimentological parameters related to the solubility, geoaccumulation and bioavailability. The total and bioavailable concentration of metals in surface sediments was quantified, the highest levels were obtained at locations with large river discharges; Cr, Ni and Zn had the highest total concentrations and Ni presented a high ecological risk which represents more than 90% mortality incidence for benthic species due to the intrinsic toxicity of this metal accumulated in the lagoon sediments. The low-carbonate sediment contains evidence of terrestrial origin, and there was a good correlation between the silts and clays with the organic carbon which values were within those reported for coastal lagoon ecosystems. The content of essential metals such as Cu and Zn were high in *Crassostrea rizophorae* oysters while potentially toxic elements such as Ni and Cr had high concentrations in the seagrass *Ruppia maritima* and mussel *Brachidontes exustus*. The Cd reported levels higher than the Pb in all evaluated organisms, mainly bivalves.

Keywords: Metals, Marine pollution, Sediments, Benthic organisms, Coastal lagoon, Gulf of Mexico.

Introduction

Currently, coastal lagoons in Mexico face serious problems caused by pollution, which produces considerable damage to the organisms that inhabit them (Botello & Páez-Osuna, 1986). These ecosystems are extremely fragile and have been suffering very important transformations along the entire country's coast. These are caused by river damming,

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closing of the communication between the lagoons and the ocean, and municipal and industrial waste water discharge from the surrounding populations, which contain various contaminants, including an important number of metals (Villanueva & Botello, 1992). Mendoza-Carranza *et al.* (2016) reported that in southeastern Mexico metal pollution has been associated with agrochemical use, urban expansion, and particularly petroleum extraction. Most of these pollutants are transported by runoffs to rivers and are finally deposited in water bodies with low hydrological dynamics such as coastal lagoons where accumulation occurs primarily in sediments (Lester, 2014; Melgar *et al.* 2008).

Metals and metalloids are highly reactive elements which are accumulated in minerals and organisms within the aquatic environment. These contaminants are present in the form of trace elements, being in water, and in the sediment or the marine biota, and thus can become toxic if found in high concentrations. The occurrence and bioaccumulation of many pollutants through the food-chain could present a risk to human health (Comby *et al.* 2014). There are other elements that do not belong to the transition group but, through mobilization by human activity, their levels have exceeded the concentrations found naturally, such is the case of lead. The impact that these increases have on terrestrial and aquatic ecosystems is very high, since they can affect organisms at both, somatic and genetic levels (Song *et al.* 2014). There are many metals such as: copper, cobalt, chromium, zinc, fluorine, and vanadium that are a natural part of biologic systems, but when their levels are increased or they change their chemical form they can become toxic.

Metallic elements are found in all living organisms and play several roles. They may be structural elements, stabilizers of biological structures, components of control mechanisms (e.g., in nerves and muscles), and in particular, are activators or components of redox systems. Thus, some metals are essential elements, and their deficiency results in biological functions impairment. Essential metals, when present in excess, may even be toxic; it should be recognized that metals might also have deleterious effects on other animal species and plants; such effects may lead to modification of an entire population or species assembly in an ecosystem and also may be of great significance to human life so theyshould be considered in the total evaluation of environmental pollution by metals and their compounds (Nordberg *et al.* 2007).

The effects of pollutants on human health have led to the evaluation of trace metals presence in the last decades of research, since they have characteristics such as persistence and bioaccumulation capacity and are considered as dangerous substances, they are also a part of complex mixtures in the coastal zone from non-point sources, especially in developing countries such as Mexico (Diop *et al.* 2015; Lei *et al.* 2016). Sontecomapan Lagoon is a region with no intense industrial influence, with fishing, tourism and local agricultural activities, for self-consumption, so it is likely that the potentially toxic metals found in this lagoon ecosystem come from industrial areas such as Coatzacoalcos and Minatitlán located in the southern region of the state of Veracruz, and these anthropic emissions are transported to the study area mainly by winds and rains compared to the specific contributions of temporary runoff around the lagoon.

General characteristics of metals

Lead (Pb) and Cadmium (Cd) are non-essential metal for organisms. However, Pb exists in every tissue and organ in mammals and it presents two different oxidation states, Pb^{+2} and Pb^{+4}, the first being predominant in an aquatic environment; in sea water, it is found as $PbCl_2$ (43%), $PbCO_3$ (42%) and $Pb(OH)_2$ (9%), additionally there are some sulfurous compounds formed with this metal under anaerobic conditions in marine sediment. Pb quantities increase in places where there is mud discharge from treatment plants, port dredging and waterways, or where there is discharge from industrial waste into rivers, estuaries and coastal lagoons (Nordberg *et al.* 2007; Jacob *et al.* 2018). Cd in freshwater environments is closely associated with colloidal matter as $CdCl_2$ and $CdSO_4$; in the ocean, 66% is present as Cd^{+2} ion along with $CdCO_3$ (26%), $Cd(OH)_2$ (5%), $CdCl_2$ (1%), and $CdSO_4$ (1%); in coastal waters and estuaries, a high proportion of this metal is found associated with particles, forming complexes. The main sources of Cd in aquatic environments come from drainage water from agricultural soil and discharge from mining and industry in addition to municipal wastewater discharge and the sludge from treatment plants (UNEP, 1985; Nordberg *et al.* 2007); its presence in marine environments reduces the survival of fish larvae and juveniles, mollusks and crustaceans (Fukunaga & Anderson, 2011).

Chromium (Cr), Copper (Cu), Nickel (Ni), Cobalt (Co) and Zinc (Zn) are considered to be essential metals for living organisms when found at low concentrations and under certain chemical forms. For example, Cr III is insoluble and difficult to transport into the cells whereas Cr VI is very toxic because of its solubility and easy penetration into the cells. In rivers and oceans, Cr is found mainly in hydroxide complexes such as CrO^{-2} ion, becoming part of the manganese nodules in marine waters. In coastal environments, it is intimately associated to wastewater discharge from industry: mining, tannery, galvanoplasty and fertilizer industry (Nordberg *et al.* 2007; Jacob *et al.* 2018). Cu acts in metabolism at enzymatic levels and the effects of its toxicity or deficiency on animals depends on its interactions with other elements, such as molybdenum and zinc. Ni, although it is an essential metal, in high concentrations it can cause cancer in lungs and nose, as well as severe dermatitis; it is intimately related with vanadium because it is also employed as a catalyst in the petroleum refining process; its high concentrations are caused directly by activities related to the oil industry (Nordberg *et al.* 2007; Jacob *et al.* 2018). Co acts as a coenzyme, depending on the chemical form; variations in its concentration can represent a risk and cause of dermatitis (Ponce-Vélez, 1988; Ponce-Vélez & Botello, 1991; Nordberg *et al.* 2007). Zn is found abundantly in the human body; it generally shows low toxicity to aquatic life, although some species are sensitive to low concentrations; however, the emission of zinc into the atmosphere can cause lung damage (Jacob *et al.* 2018).

Metals in water

In water, metals can be found dissolved, in a colloidal state, or in suspension, adhered to materials such as organic matter (Goldberg, 1976). The state of metals speciation in the aquatic media is difficult to establish, since these elements form complexes with inorganic ligands and organic compounds present in water, thus forming a reservoir of metals in solution. The study of metallic complexes formation in an aquatic environment is of great importance if you consider the toxicity that some of these metals have on the organisms present in the environment, and aids to determine their final destination (Mandelli, 1979). Heavy metals can be considered as other aspect of the human fingerprint as well, although all of them are present in the earth crust, their wide presence in waters, soils and atmosphere is mainly due to human activities; their sources in the environment are difficult to identify, as they could be a combination of natural and anthropogenic sources, both diffuse and point (Bayen, 2012).

Metals in sediment

In marine sediments, the concentrations of these elements are more easily measured than those found in water columns, mainly due to the fact that metals tend to sediment or be adsorbed because of their high chemical reactivity. Thus, it is important to carry out a geochemical study to determine the degree of contamination for the water body. There are several advantages in using sediments for this type: they are relatively stable deposits of real and potential contaminants; they can provide a historic record of the contamintants contribution from anthropogenic activities; they allow the procurement of more adequate samples, reproducible and easier to analyze compared to water and organisms, and they facilitate data collection in a wide range of environmental conditions (GEMSI, 1983). For geochemical studies, it is important to carry out analyses to quantify organic carbon concentrations, the presence of calcium carbonates and the granulometric composition of the sediment, since the affinity of certain metals can depend on some of these parameters. Organic material, for example, is a source of energy for the sedimentary system, and biological agents are the main factors for diagenetic changes, which allow for materials transformation in both, their composition and their physicochemical properties; it is here where trace elements are retained. This organic material can be found in aquatic systems in two forms: particulate and colloidal (Libes, 1992).

Precise information on metal interactions is derived from local and regional studies, like the one carried out by Ponce-Vélez (1988) which shows that some toxic metals, like lead and cadmium, have a certain tendency to be deposited in sediments with high concentrations of calcium carbonate. Conversely, clays adsorb metallic ions on their surface and carry them into estuaries, where they are deposited thanks to their cohesive properties and prevailing environmental conditions; therefore, the metallic elements are trapped into the sediments and are thus available to organisms (Förstner & Wittman, 1979). Metals in seawater and marine sediments can incorporate into the aquatic food webs through sediment water exchanges and then are biomagnified at higher levels of food webs (Hosono *et al.* 2011). This process may cause potential damage to mammals, vertebrates and human health as well (Chen *et al.* 2010) (Fig. 1).

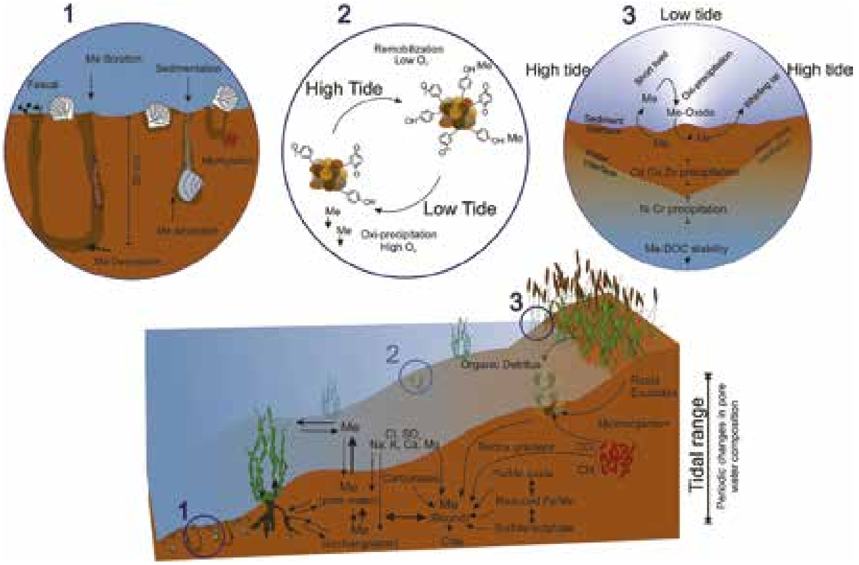

Figure 1. Metal cycle between the different compartments of an estuarine environment (de Souza *et al.* 2016).

Metals in organisms

Some biological processes in the environment can become mechanisms for mobilizing metals, which would otherwise be lost in the sediments. In this regard, bacterial action plays an important role in the mobilization of toxic metals within the aquatic environment. Microorganisms are capable of interacting with metals and change their oxidation state or organic complex thereby affecting the speciation and mobility of the metal elements (Olaniran *et al.* 2013; Jacob *et al.* 2018). However, due to the capability of metals to form complexes with organic matter, they tend to be fixed in the tissue of exposed organisms. This is one of the greatest problems that metals present as contaminants in the aquatic environment, since organisms commonly suffer from a phenomenon of **bioconcentration**, which is the ability of organisms in the same trophic level to absorb a contaminant into their tissues from the environment that surrounds them. Conversely, a process of **bioaccumulation** can occur; this refers, not only to the capacity to concentrate it but also to incorporate it into their tissues through metabolic processes. Therefore, at some point in time, the contaminant concentration in tissues is greater than that of the surrounding environment. Another phenomenon is **biomagnification**, which refers to a contaminant increase within successive trophic levels, that is, a concentration of a certain contaminant in organisms greater than that in the organisms they fed upon, which causes a greater concentration in the higher levels of the trophic web; this event has been proven for metals like mercury (Cabrita *et al.* 2017; Cipro *et al.* 2017), cadmium (Majer *et al.* 2014), and potential biomagnification for Zinc (Cheung & Wang, 2008; Cardwell *et al.* 2013) (Fig. 2).

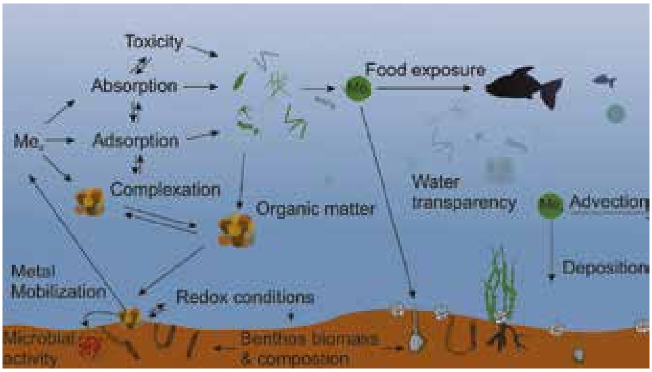

Figure 2. Routes and metal interactions in the coastal zone. Mercury as an example of toxic metal (Mandelli, 1979).

Some organisms are indicators of contaminants, either through fixation of these in their tissues, because of their abundance, or through their unexpected absence. In order for a species to be used as an indicator of particular characteristics of a biotype in environment, it must:

1) Be easily identified, even by those who are not experts.
2) Be abundant in its preferred habitats, throughout an extensive geographical region.
3) Exhibit approximately the same degree of tolerance to a particular parameter.

4) Possess a moderately long life cycle.
5) Be relatively sessile or at least not able to temporarily escape from the environmental changes by rapid migration (Farrington *et al.* 2016).

The levels of metals in mollusks and other invertebrates are often considerably higher than in other constituents of marine environment due to their habitat and feeding habits (Sun *et al.* 2011). Compared to sediments, mollusks exhibit greater spatial sensitivity and therefore, are the most reliable tool for identifying sources of available metal contamination (Hamed & Emara, 2006). Under certain environmental conditions, metals might accumulate up to toxic concentrations and cause ecological damage (Bai *et al.* 2011). Metals such as iron, copper, zinc and manganese, are essential since they play important roles in biological systems (Hogstrand & Haux, 2001), whereas non-essential metals such as Pb, Cd and Hg are usually potent toxins and their bioaccumulation in tissues leads to intoxication, decreased fertility, cellular and tissue damage, cell death and organ malfunction (Oliveira Ribeiro *et al.* 2002).

This set of characteristics is met by certain types of algae, marine phanerogams and some mollusks, such as oysters and mussels. Thus, these have been used as bioindicators to quantify levels of inorganic contaminants that may threaten the ecosystem balance.

Justification

This work was carried out in order to have a reference point for a pristine area, since the situation faced by the coastal systems of the Gulf of Mexico is extremely serious (González-Fierro, 1995). Thus, we searched for a lagoon that was distant from the highly impacted zones and also, that was of ecological and economic importance. Therefore, the Sontecomapan Lagoon, in the State of Veracruz, was chosen because in this area there is no industry that can discharge waste directly into the system and no petrochemical complexes that could originate high levels of contamination.

Region of study

The Sontecomapan Lagoon (18° 0’ and 18° 34’ N; 95° 00’ and 95° 04’ W) belongs to the basin region formed by San Martín Tuxtla and Sierra de Santa Martha volcanos, in the State of Veracruz (Mexico). It is located in the mountain range of Los Tuxtlas, which separates the tertiary basins of Veracruz and the Salina del Istmo. The ground is made up of clastic volcanic rock, where breccia lavas, and basaltic and andesitic tuffs are predominant. The bottom of the lagoon is covered with ash from volcanic activity, fluvial bed load from nearby areas, or Eolic processes from preexisting volcanic ash (Carranza-Edwards, 1975).

It is a small lagoon of irregular shape that measures about 12 km in length and 1.5 km in width. At the northeastern side it has a mouth that permanently connects with the ocean by a narrow canal. At one side there is a sediment deposit, and on the other end of the mouth there is a basalt flow named "Roca Morro", and further down there is a valley that allows the accumulation of organic matter, thus generating a fertile area with agricultural importance. The area presents a climate type Am (f) that corresponds to warm-humid with rains all the year and an average annual temperature greater than 22 °C (García, 1988, 1990). The presence of volcanic ash has been reported at the lagoon bottom; the sediments are mainly sandy clay and belong to the morphometric unit II (Carranza-Edwards, 1975). The fresh water contribution comes from various tributaries at different points of the system, mainly from the southern and southeastern zones, and it is closely related with the climatic conditions and variations in the region. The rivers that feed it are: La Palma, Coscoapan, Viejo Coscoapan, Sábalo, and Haltajapan, and the streams are: Sumidero, Basura, Sontecomapan, Chuniapan, El Fraile, Los Pollos, and La Boya (Fig. 3).

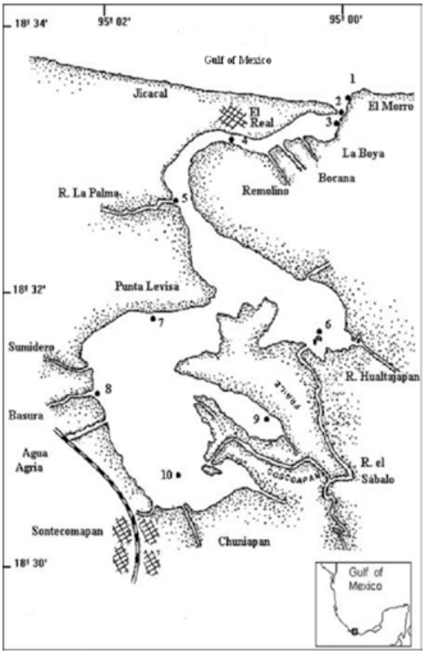

Figure 3. Location of sampling sites in the lagoon Sontecomapan, Veracruz. (CONANP, 2004).

The lagoon system is divided into several zones: the sandbar, which extends from the beach to Roca Morro, and the canal "El Real", which encompasses the zone of the Palma River and extends to a canal that opens up and makes up for the majority of the lagoon, with an average depth of 1.50 m. Total annual precipitation is between 3000 and 4000 mm. February and March are the driest months and September is the wettest with over 6000 mm. The temperature of the region is considered to be isothermal, with a maximum of 26 °C in January. The salinity data show a decreasing gradient from the sandbar to the southern side, at the mouth of the Coscoapan River (Contreras, 1993). The major body of the lagoon is completely surrounded by a belt of mangrove, *Rhizophora mangle* and *Avicennia* sp., strongly impacted; there is submerged vegetation like *Ruppia maritima*, which is present in a patchy distribution. The aquatic plankton is characterized by the presence of filamentous algae, diatoms, protozoa, dinoflagellates, rotifers, crustaceans and fish larvae. Among the commercially important species are white shrimp, oyster, prawn, snapper, sea bass, mullet, red snapper, big-eye thresher, croaker, and white mullet; this book includes recent information on the flora and fauna of the lagoon. Since 2004, the site RAMSAR 1342 has been rich in biodiversity and is an important ecosystem in the migratory route of various bird species (CONANP, 2004).

Materials and Methods

Sampling

Ten stations at the lagoon were sampled for collection of organisms, water and sediments during two annual cycles (1991-1992), taking into consideration the different climatic seasons of the region: "dry season" (March-June), "rainy season" (July-October), and "Nortes season" (November-February) (González-Fierro *et al*., 1994; González-Fierro, 1995); it should be mentioned that in sites 1 and 3 only organisms were obtained. The physicochemical parameters measured in water column were: temperature, salinity, pH, depth, and transparency.

In sediments, organic matter, calcium carbonates and texture percentages were determined. To record temperature we used a Taylor thermometer (-10 to 200 °C); salinity was measured using a refractometer; pH values were registered using a Piccolo field pH meter; depth and transparency were measured with a Secchi disc.

Sediments

In order to obtain sediment samples, we used a Van Veen dredge. In order to avoid sample contamination by metals, only the sediments from the central part of the dredge were taken, without coming into contact with its walls. The sediments were then placed into plastic bags into a cooler to be sent to the laboratory for processing.

Water

Water samples were collected in one-liter plastic bottles in surface water, 5 ml of concentrated sulfuric acid were added and the samples were then kept frozen until they were processed in the laboratory.

Organisms

All organisms were collected manually, placed in plastic bags and stored in a cooler (González-Fierro, 1995).

Laboratory processing of samples

Sediment conditioning

Samples were thawed at room temperature, then dried at 50 °C in an oven for 48 h, except for the portion that was used for granulometry analysis.

Organic Matter (determined as Organic Carbon OC)

Between 0.2 and 0.5 g of dry sediment (using the fraction that was retained in a 0.2 mm mesh) was placed in a 500 ml Erlenmeyer flask, 10 ml of 1N potassium dichromate were added and mixed. 20 ml of concentrated sulfuric acid were added and mixed for one minute, gently swirling the flask in order to ensure a complete mixture of the reagents with the sediment, and taking care that the sample did not remain adhered to the walls so that it remained in contact with the reagents. The mixture was allowed to sit for an additional 30 minutes and then diluted to 200 ml with distilled water, then, 10 ml of 85% phosphoric acid, 0.2 g of sodium fluoride and 15 drops of diphenylamine indicator were added. The mixture was titrated with a solution of ferrous sulfate 0.5 N. The color changed from brownish green to brilliant green. As control, a blank was run in the same manner without sediment (Gaudette *et al.* 1974)

Carbonates

The samples were dried in porcelain capsules at 110 °C for 8 h, then put into porcelain mortars to pulverize and homogenize them, and stored in polyethylene bags for posterior analyses. The determination was carried out using a Bernard calcimeter, which consists of a graduated glass column joined, using a latex hose, to a glass bubble that serves to level the column. At one end, we placed a reaction flask with two entries, one to the Bernard column and the other to a burette containing 50% HCl. The calcimeter measures the displacement in a graduated column after the addition of HCl to the sediment. Since this is a manometric analysis, it is necessary to consider the pressure and temperature in the system when equilibrating the column with the atmospheric pressure. Thus, in the calculations we included a factor determined by the Berthelot equation (real gases) (Hesse, 1971).

Granulometry

For the determination of the percentage of gravel, sands, silt, and clays, we followed the techniques proposed by Folk (1974), known as sieves and pipettes, based on the rate of sedimentation in a ratio inversely proportional to the size of the particle.

Metals in Water

Water samples were thawed at room temperature and a 50 ml aliquot was digested with $HClO_4$ at 100 °C until approximately 10 ml of residue was left. This was diluted to 50 ml, 3 ml of HBF_4 were added, and the sample was read in an atomic absorption spectrophotometer (Villanueva, 1987).

Metals in Sediments

For determination of total concentration, the material used for the analysis was washed for three days in HCl 2N, then for three days in HNO_3 2N, and finally rinsed in distilled water and stored in plastic bags for later use. The sediment was dried at 60 °C, pulverized and sifted through a plastic screen, 5cm in diameter. 0.25 g were placed in a PTFE vessel with 10 ml of inverted aqua regia (HNO_3:HCl, 3:1). The digesters were maintained at 100 °C ± 10 °C for approximately 18 h and the residue washed in bidestilled water. The supernatants were read in a flame atomic absorption/emission spectrophotometer (Shimadzu), using blank controls for every group of eight samples, and standards of known concentrations (Loring & Rantala, 1977; modified by Páez-Osuna & Osuna-López, 1990). For determining the bioavailable fraction, 2 g of sediment were washed for 24 hours in 25 ml of 25% CH_3-COOH, and then rinsed in bidistilled water. Supernatants were collected and read by atomic absorption in the same manner (Agemian & Chau, 1976).

Metals in Organisms

The collected organisms were dried at 60 °C for 48 h and tissue was pulverized in a porcelain mortar. A sample was taken for metal extraction by acid digestion with concentrated HNO_3. Once the acid was added, the samples were dried at 350 °C for two hours and then, the operation was repeated. The ashes obtained were resuspended in 2N HNO_3, centrifuged and if necessary, and filtered. Samples were read in an atomic absorption spectrophotometer (Goldberg *et al.* 1983).

Results and Discussion

The data reported in this paper correspond to the original research developed by González-Fierro in 1995 at the Grupo de Contaminación Marina del Instituto de Ciencias del Mar y Limnología de la Universidad Nacional Autónoma de México, as part of an integral project to carry out the environmental diagnosis of the Sontecomapan Lagoon and to record the first data on various environmental pollutants such as metals; a general description of the metals concentrations in sediments and organisms of this lagoon has already been published in Spanish by the authors of this chapter (González-Fierro *et al.* 1994).

Water

Physical and chemical parameters

Table 1 shows the physical and chemical parameters data recorded *in situ* during annual cycle, in the dry season in March, 1991 greatest depth, transparency and salinity were recorded, the latter variable behaved as expected because of high evaporation leading to maximum values of 24.9 ups; however, high depth may be justified due to unusual rains input prior to sampling, which increased the strap depth of the lagoon up to 6 m.

It notes that, from the 1991 drought, subsequent sampling showed a freshwater to brackish (oligohaline) behavior in the lagoon system studied, as the salinity ranged from 0.25 ups in rains from August, 1992 and 4.0 ups in rainy July, 1991, a scenario that directly influences metal solubility in the water column (Owsianiak *et al.* 2015). The pH, another important factor for the solubility of metals evaluated in this work, stayed about 7, being more basic in dry seasons of 1991 and 1992, with 7.94 and 7.71, respectively; the lowest value was 6.8 recorded in rainy September, 1991. It is known that when the pH is neutral or close to 7, metals are in solution in water and have a longer residence before reaching the sediments (Mendoza-Carranza *et al.* 2016).

Metals

It was not possible to reliably determine the concentrations of metals in water, because they were below the detection limit of the instrument used (<1 μg/l), an atomic absorption spectrophotometer, even in the case of high aqueous solubility metals such as cadmium.

Sediments

Organic Carbon

Table 2 shows the results of Organic Carbon (OC) which are presented in percentage values as an indicator of organic matter presence in the lagoon sediments; there is a noticeable homogeneity in the mean along the six sampling periods. The overall range of OC was 0.11 (August, 1992 rainy season) to 4.16% (September, 1991 rainy season), which shows that the transport of organic matter occurs through the important runoff draining into the lagoon. The duration and intensity of the rains, that can vary from year to year, influence the organic material deposited in sediments and is reflected in these different values from one year to another during the rainy season, so it is important to have data periods longer than an annual cycle to evaluate the oscillations in large temporary records.

Table 1. Physicochemical parameters in water column of Sontecomapan Lagoon, Veracruz. 1991-1992

Season (Date)	Depth (m)	Transparency (m)	Temperature (°C)	Salinity (ups)	pH
Dry Season (March 1991)					
Mean	2.7	1.01	25.37	13.66	7.94
Standard Deviation	1.49	0.37	1.09	5.69	0.21
Range	1.4-6.0	0.5-1.5	24-27	8.1-24.9	7.5-8.1
Rainy Season (July 1991)					
Mean	2.03	0.55	22.62	1.0	6.95
Standard Deviation	1.32	0.33	1.84	1.84	1.33
Range	0.7-4.0	0.1-1.0	20-25	0.0-4.0	6.7-7.3
Rainy Season (September 1991)					
Mean	NR	0.72	28.5	1.91	6.8
Standard Deviation	NR	0.06	1.87	1.02	0.17
Range	NR	0.6-0.7	25-30	0.0-3.0	6.5-7.0
"Nortes" Season (November 1991)					
Mean	1.14	0.63	26.06	0.42	7.44
Standard Deviation	0.22	0.31	1.52	0.78	0.64
Range	0.9-1.45	0.2-1.1	24-28	0-2.0	6.5-8.0
Dry Season (March 1992)					
Mean	1.7	0.81	28.62	NR	7.71
Standard Deviation	0.79	0.29	1.24	NR	0.6
Range	1.0-3.0	0.4-1.4	27-31	NR	6.8-8.6
Rainy Season (August 1992)					
Mean	1.26	0.31	29.17	3.12	6.98
Standard Deviation	0.55	0.15	1.28	8.83	0.44
Range	0.6-2.5	0.1-0.6	27-30	0-0.25	6.4-7.8

NR Not Reported

Table 2. Organic Carbon as an indicator of organic matter content in the sediments of Sontecomapan Lagoon, Veracruz. Values in %. 1991-1992

Site	Dry Season (March 1991)	Rainy Season (July 1991)	Rainy Season (September 1991)	"Nortes" Season (November 1991)	Dry Season (March 1992)	Rainy Season (August 1992)
2	0.15	0.23	NA	0.54	0.17	0.11
4	0.23	0.13	NA	1.04	0.82	0.15
5	1.39	1.44	0.37	0.23	0.90	0.24
6	2.10	2.20	0.88	2.03	2.26	1.11
7	0.28	0.48	0.36	0.43	0.46	0.91
8	1.63	1.77	2.77	2.19	2.70	2.75
9	1.87	2.02	2.87	3.17	2.38	1.97
10	3.77	2.24	4.16	3.34	3.74	1.79
Mean	**1.43**	**1.31**	**1.90**	**1.62**	**1.68**	**1.13**
Standard Deviation	1.23	0.90	1.58	1.24	1.27	0.97
Range	0.15-3.77	0.13-2.24	0.36-4.16	0.23-3.34	0.17-3.74	0.11-2.75

NA Not Analyzed (very small sediment sample)

Granulometry

Figure 4 shows sedimentary texture considering only the four particle categories most reported in the literature on coastal and marine sediments pollution. The sand content was dominant in the sampling sites 2-7 with values between 59.74 (Station 6) and 99.4% (Station 2) which does not represent a separate depositional environment for the retention and storage of metals; in contrast, the sites 8 to 10 had clay dominance, except at site 8 wherein the sediment was silty clay type with a ratio of 44: 43% and the remainder was made up of sand and gravel; in general, a higher metal content would be expected at these three sampling points (Burdige, 2006; Szava-Kovats, 2008; Belabed *et al.* 2013).

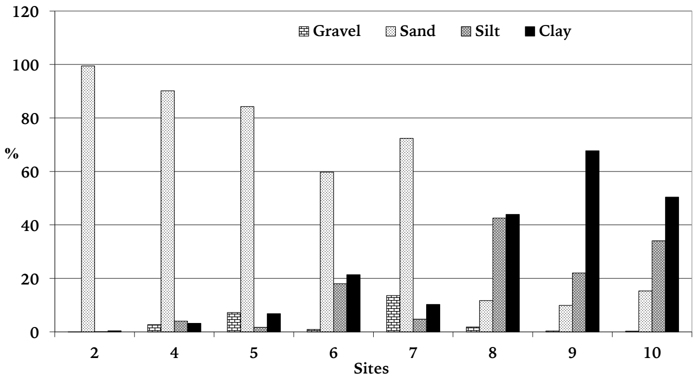

Figure 4. Type of sediment in the Sontecomapan Lagoon, Veracruz. 1991-1992.

Correlations were made with all the registered data and were significant between silts or clays and organic carbon were obtained as shown in Figure 5, which point out towards the existence of a depositional environment enabling for the accumulation of metals at specific sites (6, 8, 9 and 10); it should be noted that the sands and OC show a significant inverse correlation (r=-0.9376, *p*=0.0006) agreeing with what is reported in literature; with gravel, no relationship was found probably due to the small amount found.

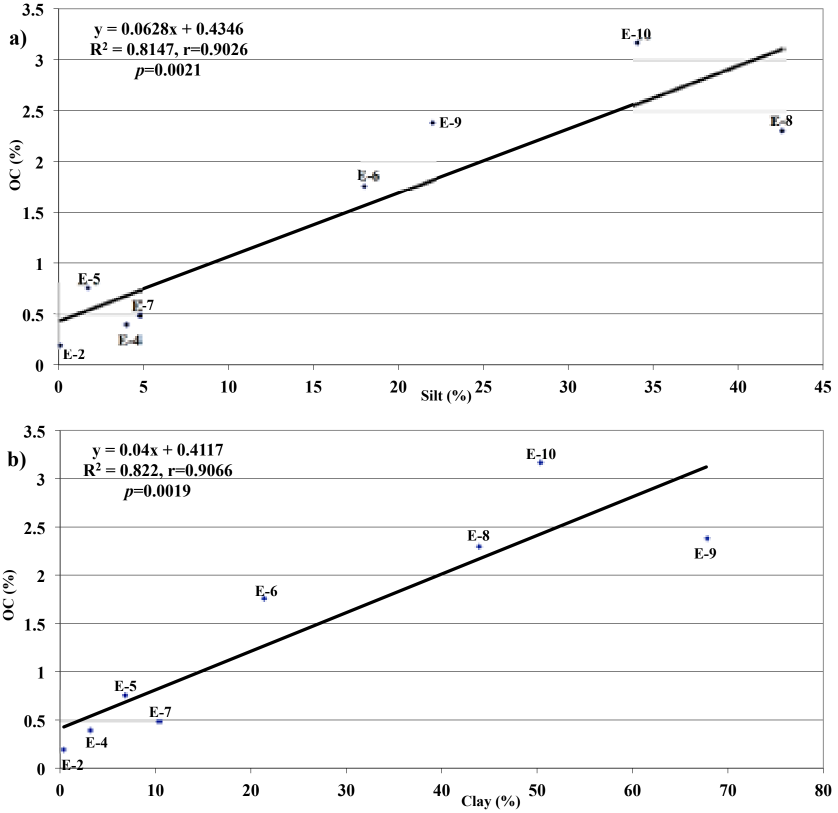

Figure 5. Correlations between silts and OC (a), and clay and OC (b) in sediments of Sontecomapan Lagoon, Veracruz. 1991-1992.

Calcium Carbonate

The values of this sedimentary component were low in the lagoon because at Sontecomapan it can be in the form of colloids which prevented its detection in this study, since in all samples it was < 0.30%. It has been reported that low carbonate content in sediments may indicate a terrestrial origin or a siliciclastic type (Vallejo-Toro *et al.* 2016).

Total Metals

The criteria for evaluating sediment quality assessed in a wide study from the mid-90s by a group of Canadian and American researchers (Long *et al.* 1995), have become the benchmarks to assess potential or actual risk that the content of various elements and compounds represent to the benthic community. International government entities as the US Agency of the Atmosphere and Ocean (NOAA), established a reference guide to assess possible biological damage from toxic substances accumulated in the sediments of aquatic ecosystems (Buchman, 2008). Thus, there are two critical values, the ERL (effects range low values) whose value indicates a low probability of causing alterations on biotic benthic species and ERM (effects-range median values) concentration from which the deleterious effect on these organisms occurs; this means that a value with magnitude between ERL and ERM has a chance of benthic damage.

Figures 6, 7 and 8 show the variability between periods and stations for the total concentration of the seven metals on Sontecomapan sediments. Copper (Cu) showed the highest level at site 9 with 131 μgg^{-1} during the rains of August, 1992 while the minimum was 1.1 μgg^{-1} at site 2 during "Nortes" season in November, 1991; most of sedimentary total Cu was < 80 μgg^{-1}. Overall, 43.5% of the sites had Cu concentrations > ERL (34 μgg^{-1}) (Fig. 6) where point 9 already mentioned the most likely alteration to the benthos.

For cobalt (Co) no sediment quality criteria is yet established. However, its records in this study showed concentrations around 35-45 μgg^{-1} with the exception of the rainy season and "Nortes" 1991 where values decreased below the analytical detection limit (Fig. 6).

Chromium (Cr) showed its highest level at site 10 with 150.6 μgg^{-1} during the rainy season (July, 1991), and the minimum of 14.7 μgg^{-1} towards the end of this climatic period in September of the same year. When compared to the criteria for sediment quality, 41.3% of Cr concentrations exceeded the guideline value of ERL of 34 μgg^{-1}, mainly in stations 6 8, 9 and 10 but without exceeding the upper ERM of 370 μgg^{-1} (Fig. 6).

Total nickel (Ni) concentrations are in the range of 3.07 to 269.6 μgg^{-1} at 4 sites during the dry season of 1992 and 2 sites in the rainy season of July, 1991, respectively. The levels found in this study show a worrying situation regarding the biological risk posed by Ni stored, since 13% (6 sites) of the records were in the range ERL<concentration<ERM, while 50% of the data were > ERM (51.6 μgg^{-1}) which means a real biological risk and inherent ecological impact to the benthos; the magnitude of total Ni concentrations decreased by 1992, mainly in the rainy season (August), as even non-detectable data were obtained from 17.4% of sediment sampling sites (Fig. 7). Total concentrations of zinc (Zn) did not exceed the lowest quality criteria for sediment ERL (150 μgg^{-1}) and its range was between 11 (station 6; August, 1992) and 104.4 μgg^{-1} (at site 10 during the rainy season (July, 1991) (Fig. 7).

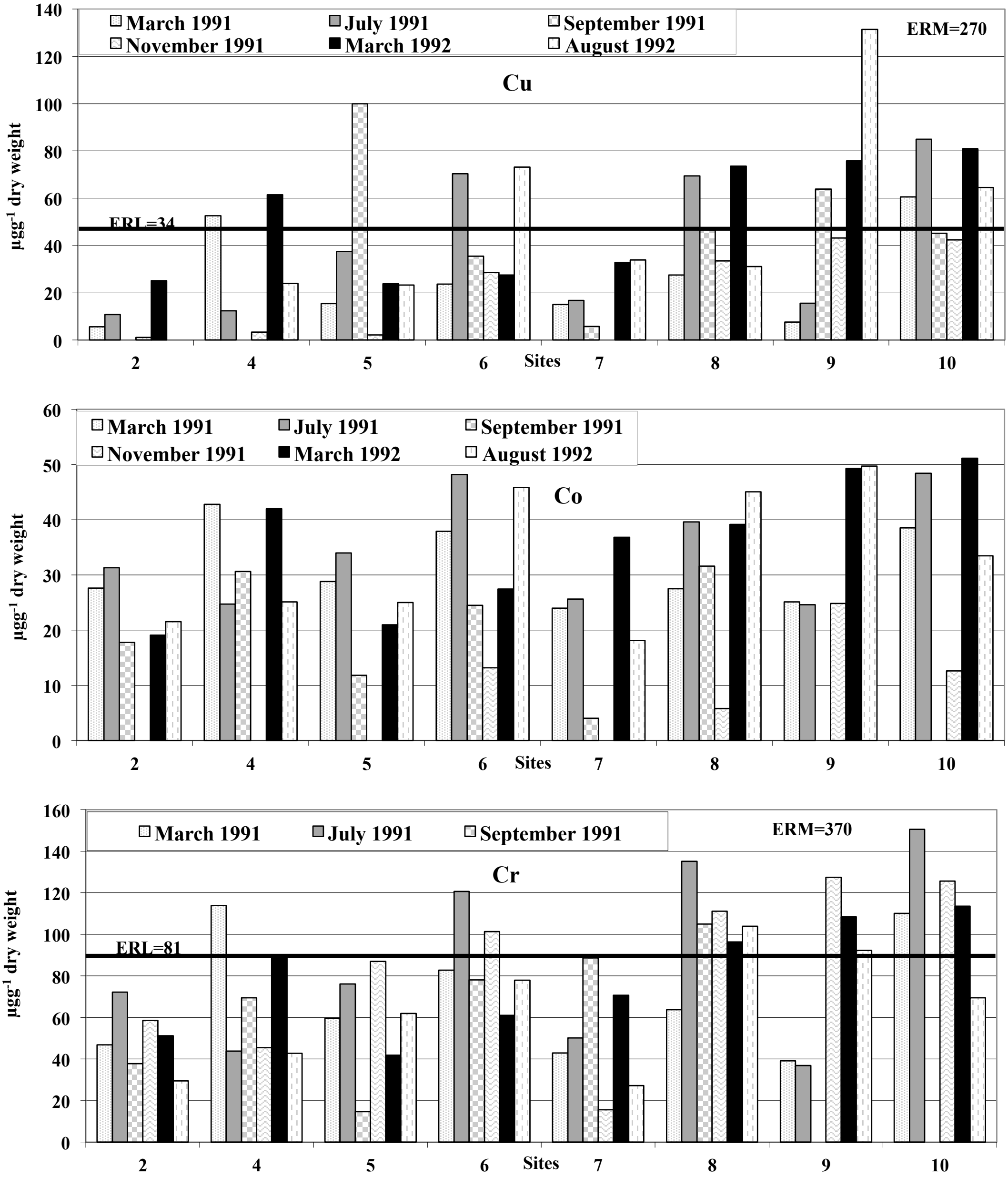

Figure 6. Total concentrations of Cu, Co and Cr in surface sediments of Sontecomapan Lagoon, Veracruz. 1991-1992.

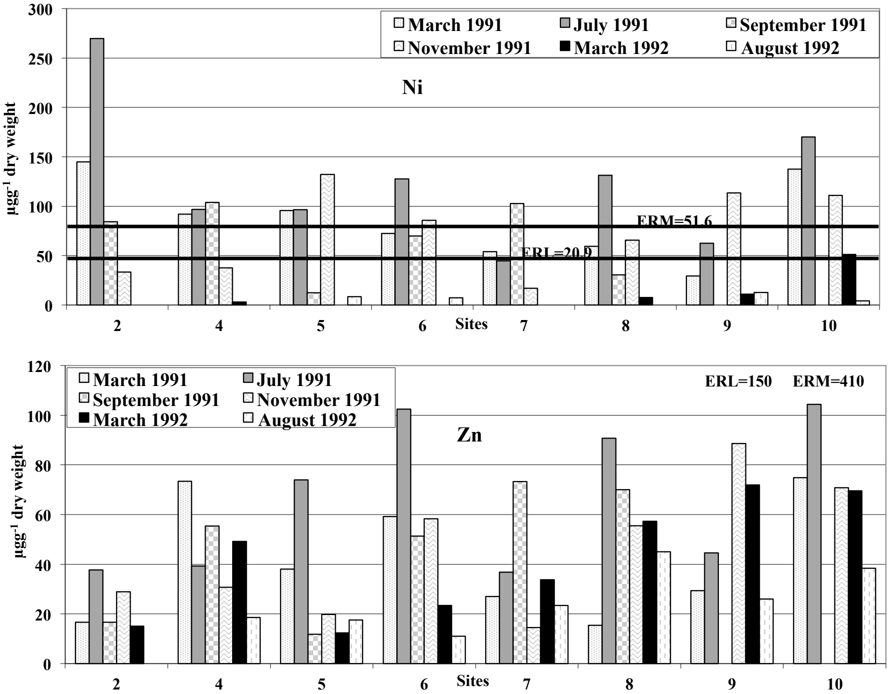

Figure 7. Total concentrations of Ni and Zn in surface sediments of Sontecomapan Lagoon, Veracruz. 1991-1992.

Total cadmium (Cd) concentration showed quite uniform values, with a minimum of 0.3 μgg^{-1} at site 7 in September, 1991 and a maximum of 2.7 μgg^{-1} in sampling point 10 during "Nortes" (November, 1991) (Fig. 8). Cd was undetected in 40% of samples, six of the sites evaluated had levels <ERL (1.2 μgg^{-1}) while 50% was ERL<concentration<ERM with a probability of actual benthic damage in most cases.

The lead (Pb) concentration ranged between 2.77 (site 5; August, 1992) and 48.2 μgg^{-1} (site 6; July, 1991). Most data were <ERL (46.7 μgg^{-1}), only point 6 mentioned above surpassed this guide level, and close to it were the concentrations found in sites 8 and 10 in the rainy season of 1991 and at three sites values were under the detection limit (Fig. 8).

The mean values for the concentrations of the seven metals determined in the sediments of the Sontecomapan Lagoon, Veracruz, are summarized in Figure 9; Cr, Ni and Zn had the highest levels in decreasing order, followed by Cu, Co and Pb, being Cd the one that recorded the lowest average value of the whole study. The three dominant metals are likely to be derived from human activities in areas surrounding the lagoon system such as the use of agrochemicals that have metals in their molecules and manufacturing processes, chronic supply of crude oil that contributes to the presence of nickel as well as emissions by the engines of the fishing and tourist boats in the study area.

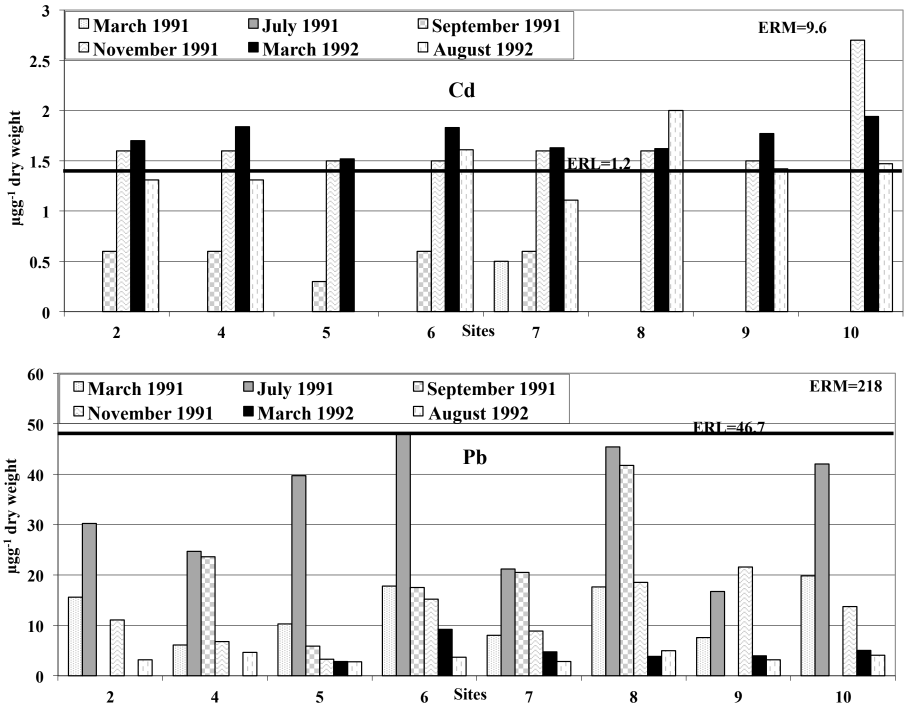

Figure 8. Total concentrations of Cd and Pb in surface sediments of Sontecomapan Lagoon, Veracruz. 1991-1992.

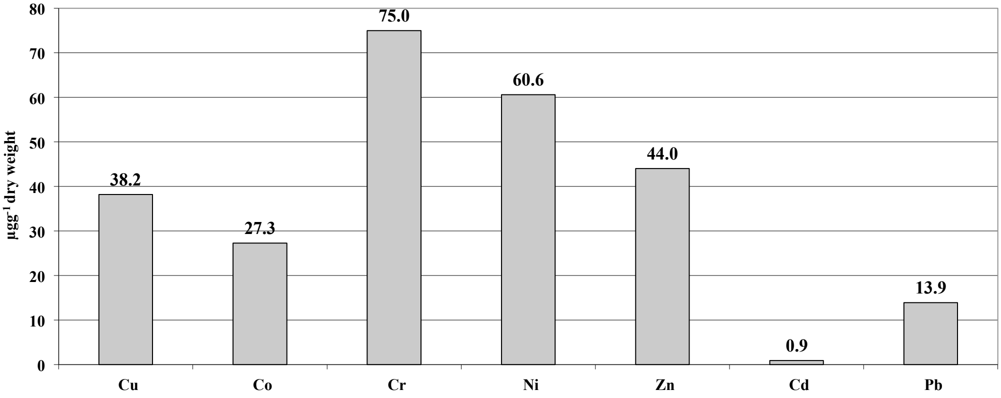

Figure 9. Total concentrations of metals in surface sediments of Sontecomapan Lagoon, Veracruz. Mean values. 1991-1992.

In order to estimate the intrinsic toxicity of sediments in a global way due to the metal assembly contained in this environmental matrix, it is recommended to use the mean Sediment Quality Guideline Quotients (mSQGQ) which is calculated using the individual concentrations quotients of the elements determined in relation to the concentration established in the sediment quality guides already mentioned, particularly ERL of each metal (Long *et al.* 2006); the result is a single, unitless, effect-based index of the relative degree of contamination that can provide a basis for determining the likelihood that a sediment sample would be toxic to sediment-dwelling organisms. Accordingly, mSQGQs provide several advantages for the interpretation of complex mixtures of potentially toxic substances in sediments significance (Wang *et al.*, 2016).

To estimate this indicator of environmental quality for the Sontecomapan sediments, only six metals over the seven recorded have been considered since the Co does not have its quality criterion of ERL sediments (Long *et al.*, 1995). The mSQGQ overall valueof the integrated metal assembly was 1.05 and compared with the reported experimental and statistical data for the incidence of significant toxicity for test organisms of estuarine and marine environments (*Ampelisca abdita* and *Rhepoxynius abronius*), was between 55-60% and classified as ecological risk from medium to high (Wang *et al.*, 2016); this ecotoxicological tool can be used to establish a concentration/response relation for a specific site and to be able to identify the toxicological risk that exists that can be acceptable or not, since as the magnitude of the mSQGQ increases the incidence of toxicity and mortality (Long *et al.* 2006).

However, it is important to note that at the individual level, Ni had the highest total concentration/ERL sediment quality guideline (C_T/ERL) ratio with 2.90 classified as high ecological risk, followed by Cu (C_T/ERL=1.12 medium/high risk), Cr (C_T/ERL=0.93 medium risk), Cd (C_T/ERL=0.75 medium risk) and for Zn and Pb was 0.29 classified as medium/low risk (Wang *et al.*, 2016) this pattern is very important to establish the metals of greatest risk for the ecosystem under study and to be able to define the required mitigation actions required to reduce the environmental impact of these substances.

Statistical analysis of the various sedimentological parameters and total concentrations of metals can provide important information about the origin of these substances and their accumulation patterns in a specific ecosystem. Table 3 shows the linear correlation coefficient between sedimentological parameters and the total concentration of the seven metals determined in this study.

It can be observed that Cu had good relationship with organic carbon (r=0.9380) and Zn (r=0.9554), both essential metals for various metabolic pathways. Co showed a significant correlation coefficient with Cr (r=0.9310) and Zn (r=0.9588). Cr had also a significant relationship with organic carbon (r=0.9131) and Zn (r=0.9415). We must highlight the fact that Ni and Cd showed no significant correlation with any metals or sedimentological parameters. Zn and OC had a significant direct relationship (r=0.9289). Pb correlations were recorded with a lower degree of significance with OC, Co and Cr (coefficients of 0.8055, 0.8360 and 0.8528, respectively).

Table 3. Linear correlation coefficients between the sedimentological parameters and the total concentration of metals in the Sontecomapan Lagoon, Veracruz. 1991-1992. Bold values mean significant correlations ($p<0.05$).

	Gravel	Sand	Silt	Clay	OC	Cu	Co	Cr	Ni	Zn	Cd	Pb
Gravel	1.00	0.35	-0.43	-0.42	-0.48	-0.46	-0.63	-0.68	-0.71	-0.55	-0.52	-0.58
Sand	0.35	1.00	**-0.92**	**-0.97**	**-0.94**	-0.86	-0.71	-0.79	-0.02	-0.83	-0.40	-0.63
Silt	-0.43	**-0.92**	1.00	0.81	**0.90**	0.79	0.78	0.88	0.12	0.82	0.54	0.79
Clay	-0.42	**-0.97**	0.81	1.00	**0.91**	0.86	0.68	0.73	0.07	0.80	0.35	0.54
OC	-0.48	**-0.94**	**0.90**	**0.91**	1.00	**0.94**	0.86	**0.91**	0.26	**0.93**	0.42	0.81
Cu	-0.46	-0.86	0.79	0.86	**0.94**	1.00	0.87	**0.90**	0.18	**0.96**	0.27	0.66
Co	-0.63	-0.71	0.78	0.68	0.86	0.87	1.00	**0.93**	0.27	**0.96**	0.59	0.84
Cr	-0.68	-0.79	0.88	0.73	**0.91**	**0.90**	**0.93**	1.00	0.42	**0.94**	0.50	0.85
Ni	-0.71	-0.02	0.12	0.07	0.26	0.18	0.27	0.42	1.00	0.20	0.08	0.50
Zn	-0.55	-0.83	0.82	0.80	**0.93**	**0.96**	**0.96**	**0.94**	0.20	1.00	0.52	0.75
Cd	-0.52	-0.40	0.54	0.35	0.42	0.27	0.59	0.50	0.08	0.52	1.00	0.54
Pb	-0.58	-0.63	0.79	0.54	0.81	0.66	0.84	0.85	0.50	0.75	0.54	1.00

Sediments are the final recipient of whatever reaches coastal ecosystems and store it for a long time; both beneficial materials like nutrients and risk pollutants such as Cd, Pb, Hg and pesticides, play a significant role in sustainability of food supply for the biological communities that inhabit them as well as to become a secondary source of dangerous substances (Bodin *et al.* 2013). Because of this, the analysis of coastal sediments continues to be a very useful tool to evaluate contamination by metals in these ecosystems since the recording of these elements high concentrations indicates anthropogenic impact rather than natural contributions due to phenomena of geological weathering (Memet, 2011).

Bioavailable Metals

Recent environmental sciences research have aimed to study the chemical speciation of metals in order to better explain distribution, mobility and especially the bioavailability of these elements to ascertain whether the concentrations are determined by these processes and existing factors in the ecosystem (Zemberyová *et al.*, 2007). Metal solution is one of the environmental processes that directly affect metals immediate bioavailability and therefore their absorption and uptake by the biota and their deposition in the sediments after sedimentation or death (Owsianiak *et al.*, 2013, 2015; Chiappetta *et al.*, 2016). The evaluation of the bioavailability of metals in coastal sediments is a very important aspect since it provides information to predict the environmental and toxicological impact of these substances because this chemical fraction is the one that can be solubilized during the process of digestion in organisms and metals can be absorbed by different metabolic pathways and accumulate in various tissues such as liver, kidney, muscle and skeletal, among others, causing different sublethal effects of reproductive or growth type as well as acute effects including death (Tumer and Olsen, 2000).

In this regard, the content of the sedimentary bioavailable metal fraction of the Sontecomapan Lagoon, was evaluated holistically, *i.e.*, without obtaining specific phases as the aforementioned. Figure 10 shows the distribution of bioavailable Cu, Co and Zn concentrations. Cu had the highest concentrations (>20 μgg^{-1}) in sites 4, 5 and 6 of the samples corresponding to rains (August, 1992) and the lowest (<1.0 μgg^{-1}) in the dry and rainy seasons of 1991.

Comparatively, Co was more uniform, with values around 1 to 6 μgg^{-1} with the exception of the sampling sites 9 and 10 in March, 1992 with concentrations >10 μgg^{-1}. For bioavailable Zn, considered an essential element, the pattern was similar to that shown for cobalt, the substantial difference was the very high concentration found at site 2 in March, 1992, which had high levels (11.87 μgg^{-1}), that did not occur for Co (3.29 μgg^{-1}) (Fig. 10).

The detected pattern for Cr and Ni (Fig. 11), may represent a biohazard due to both its chemical form and the magnitude of their concentrations. Overall, Cr had uniform values, with an increasing trend in 1992 and levels <5.0 μgg^{-1}, with the exception of sediments from sites 8, 9 and 10 collected in March, 1992 (dry season). Nickel presented a heterogeneous pattern, as its concentrations decreased to be less than the detection limit in 1992 when data could be recorded only in sites 9 and 10, whereas the maximum was obtained, with 12 μgg^{-1}, at site 9 in August, 1992. The bioavailable concentrations of Cd and Pb (Fig. 12) showed a similar distribution. They were undetectable in 1991 and could only be detected in 1992, but at concentrations higher than those of cadmium. This situation is to be highlighted because, naturally, this metal comes in smaller levels than those of Pb for total concentration (<3.0 μgg^{-1}), a scenario that was reversed for the bioavailable fraction.

Correlation coefficients between bioavailable metals and sedimentological parameters are presented in Table 4. For organic carbon the highest correlation was with Cr ($r=0.9383$) and second with Zn ($r=0.8504$). Co and Ni were also highly correlated ($r=.9440$), followed by the Co-Zn relationship ($r=0.8653$); it should be noted that an unexpected relationship was detected for Pb-Ni since it was of inverse type ($r=-0.8588$), which is not commonly found in these metals behavior. It can be said that desorption mechanisms of Co, Ni and Zn at low pH condition caused by a weak acid environment are similar in comparison with what may occur to Pb evidenced by the inverse correlation obtained with Ni.

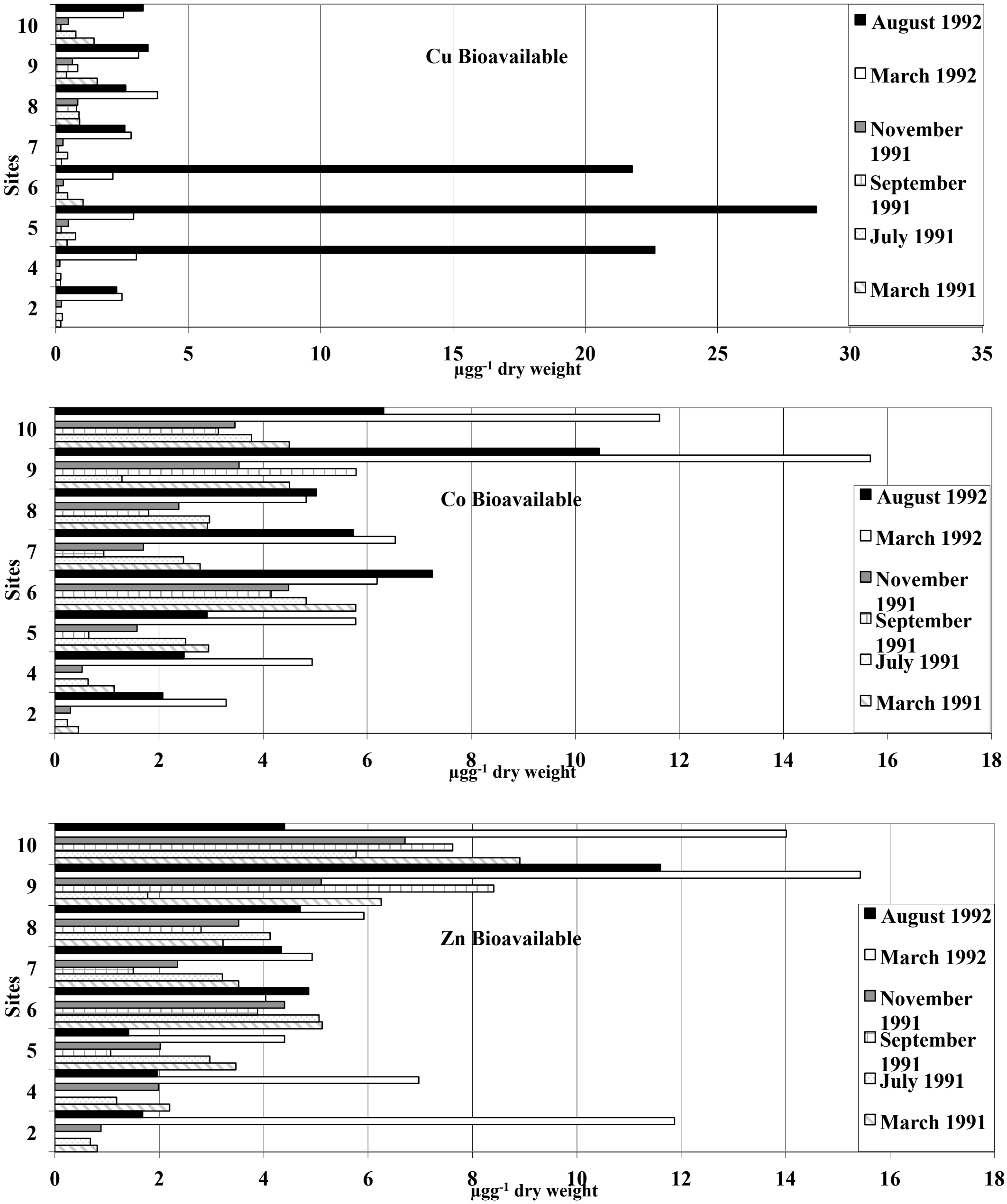

Figure 10. Bioavailable concentrations of Cu, Co and Zn in surface sediments of Sontecomapan Lagoon, Veracruz. 1991-1992.

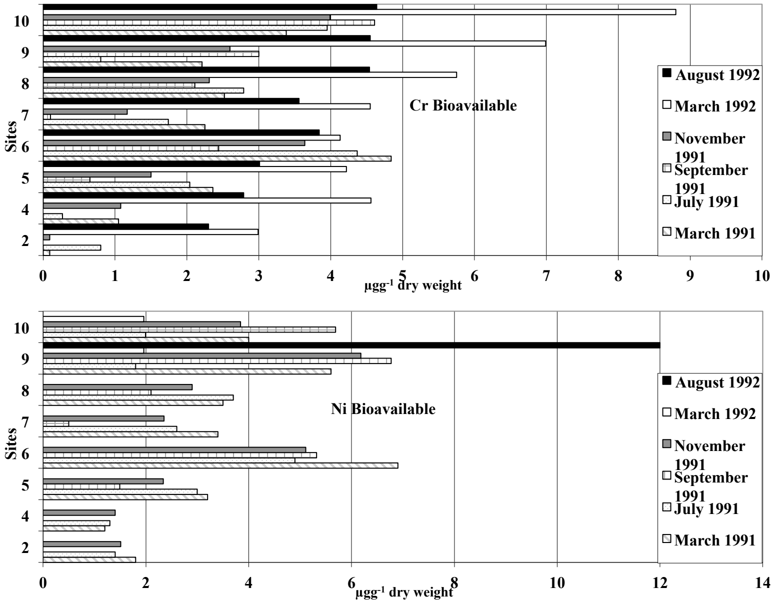

Figure 11. Bioavailable concentrations of Cr and Ni in surface sediments of Sontecomapan Lagoon, Veracruz. 1991-1992.

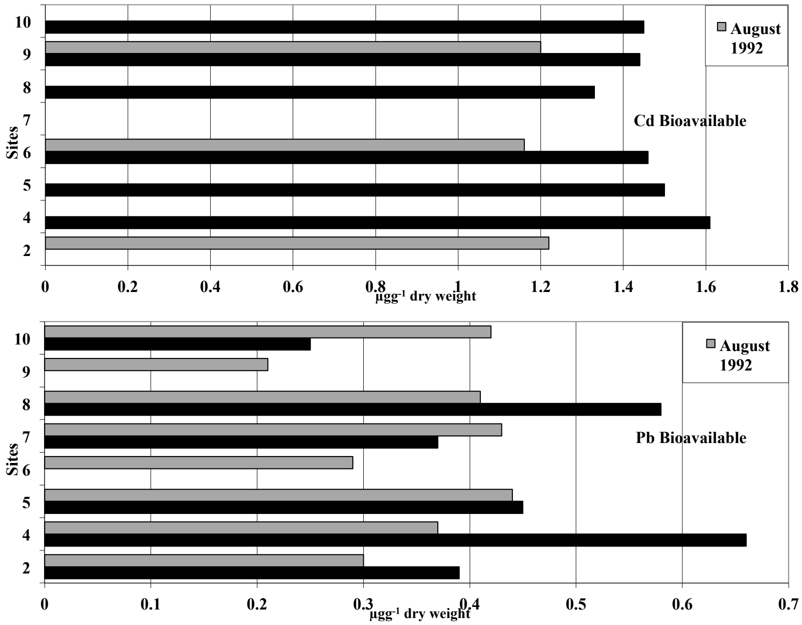

Figure 12. Bioavailable concentrations of Cd and Pb in surface sediments of Sontecomapan Lagoon, Veracruz. Cycle 1991-1992.

The adsorbed metals at sediments in their labile form represent the chemical fraction of greatest risk for biota because they can cause sub-lethal effects at population level in the majority of the species that come in contact with these chemical forms, mainly toxic elements like As Cd, Pb, As, Hg, V and Cr. Examples of these effects are the inhibition of the enzymatic activity of acetylcholinesterase in clams and mussels, oxidative DNA damage on bivalve gills, and vertebral deformities in various species of fish (Zaaboub *et al.* 2015).

Coastal ecosystems are large receivers of harmful substances, including metals, which come from several sources of industry and that acquire environmental relevance due to their accumulation in the sediments of these natural areas where they can be enriched and become a secondary source of contaminants. Most of the metals accumulated in the sediments form complexes with the fine particles, with the existing organic matter, with the Fe and Mn oxides and may be bioavailable for benthic biota or for detritivorous organisms, thus representing a toxicological risk to cause biological damage from sub-lethal to deadly in the food web (Nethaji *et al.*, 2017).

Table 4. Linear correlation coefficients between the sedimentological parameters and bioavailable metal concentrations in the Sontecomapan Lagoon, Veracruz. 1991-1992. Bold values mean significant correlations ($p<0.05$)

	Gravel	Sand	Silt	Clay	OC	Cu	Co	Cr	Ni	Zn	Cd	Pb
Gravel	1.00	0.35	-0.43	-0.42	-0.48	0.04	-0.26	-0.35	-0.37	-0.48	-0.73	0.37
Sand	0.35	1.00	**-0.92**	**-0.97**	**-0.94**	0.44	-0.77	-0.81	-0.70	-0.81	-0.32	0.34
Silt	-0.43	**-0.92**	1.00	0.81	**0.90**	-0.37	0.56	0.80	0.46	0.62	0.25	-0.12
Clay	-0.42	**-0.97**	0.81	1.00	**0.91**	-0.43	0.84	0.75	0.82	**0.90**	0.45	-0.49
OC	-0.48	**-0.94**	**0.90**	**0.91**	1.00	-0.30	0.80	**0.94**	0.69	0.85	0.44	-0.40
Cu	0.04	0.44	-0.37	-0.43	-0.30	1.00	-0.19	-0.14	-0.18	-0.44	0.35	0.20
Co	-0.26	-0.77	0.56	0.84	0.80	-0.19	1.00	0.81	**0.94**	0.87	0.57	-0.78
Cr	-0.35	-0.81	0.80	0.75	**0.94**	-0.14	0.81	1.00	0.63	0.76	0.41	-0.41
Ni	-0.37	-0.70	0.46	0.82	0.69	-0.18	**0.94**	0.63	1.00	0.82	0.71	-0.86
Zn	-0.48	-0.81	0.62	**0.90**	0.85	-0.44	0.87	0.76	0.82	1.00	0.46	-0.64
Cd	-0.73	-0.32	0.25	0.45	0.44	0.35	0.57	0.41	0.71	0.46	1.00	-0.66
Pb	-0.37	0.34	-0.12	-0.49	-0.40	0.20	-0.78	-0.41	-0.86	-0.64	-0.66	1.00

Metals in Organisms

Four species of organisms present in the Sontecomapan Lagoon, Veracruz, were assessed during this study. The *Cladophoropsis membranacea* algae was obtained from sites 1, 4, 5 and 6 in March, 1991 and site 9 in August, 1991. Seagrass *Ruppia maritima* was collected at the sampling stations 6 and 7 during dry and Nortes, 1991. The oyster *Crassostrea rhizophorae* was collected from site 3 of the two 1992 collections and the mussel *Brachidontes exustus* was sampled at sites 7 and 9 in March, 1991 and at sites 6 and 7 in rain and Nortes, 1991 and August, 1992. The bioconcentration factors (BCF) were calculated for the submerged vegetation and the bivalves due to their close relation with the sediments from the following relation:

BCF = Corganism / Csediment. It has been established that the BCF value represents the sessile species efficiency to absorb a certain element of the sediments and to accumulate it in their tissues, reason why high values of BCF mean a greater capacity of bioaccumulation (EPA, 2007).

In Figure 13 the results of the seven metals on the two species of submerged vegetation are analyzed; it can be seen that the Zn found in *C. membranacea algae* had the highest level with 66.7 μgg^{-1} for organisms collected at site 5 during March, 1991, which contrasts with the 18 μgg^{-1} value recorded for the seagrass at a nearby site (station 6) in the same sampling time. The content of potentially toxic metals, Cd and Pb were comparable for both species.

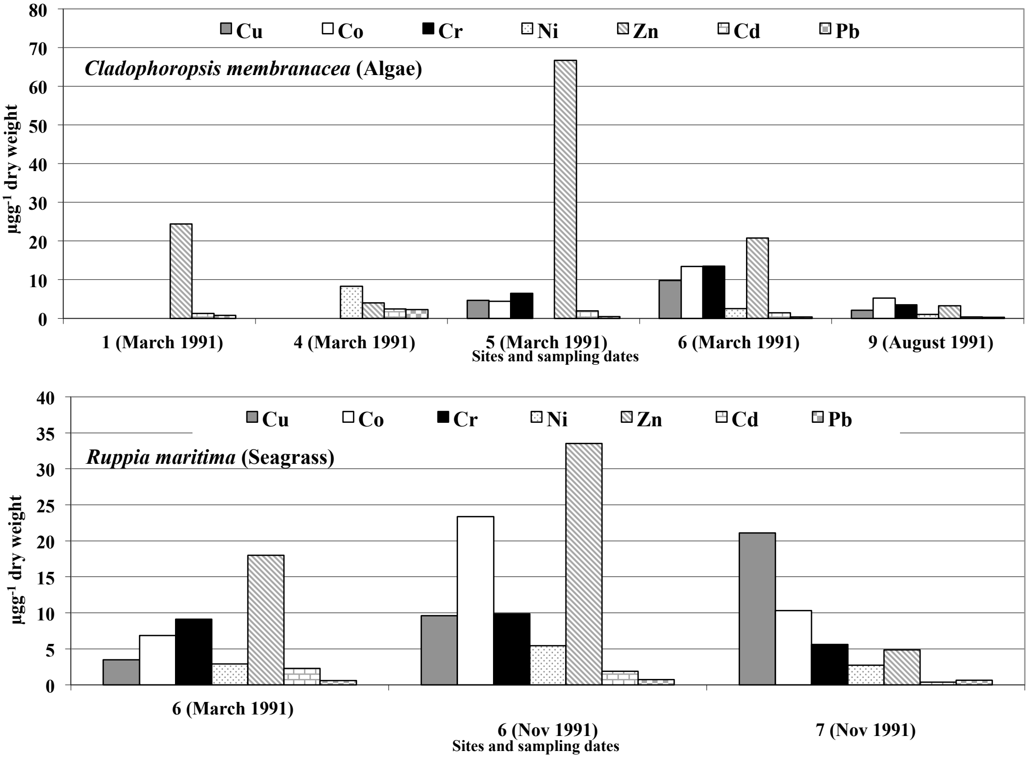

Figure 13. Content of total metals in submerged vegetation of Sontecomapan Lagoon, Veracruz. 1991-1992.

In recent years the need to have more organisms that function as biomonitors has been recognized, in order to know if the concentrations of certain substances such as metals mean an toxicological risk for ecosystems. Aquatic vegetation provides information on current conditions of the metals bioavailability and also provides historical data on the past environmental characteristics at a lower cost than water and sediment analyses which require study of complete sedimentary cores. When determining the concentration of metals in the vegetation, bioavailability of toxicological relevance and actual signs of environmental disturbance can be known (Bonano & Di Martino, 2016).

Seagrass provide comprehensive data on the ecosystem physicochemical and biological parameters where marine phanerogams inhabit and are recognized as suitable organisms for biomonitoring of trace elements (Lewis & Devereux, 2009). Similarly, seagrass has been used to trace metal contamination and, as this process of coastal disturbance has increased in developing countries, it is necessary to continue with bioindicators of this type of submerged vegetation as these plant species are able to provide information on alteration by metals of specific sites by point sources as well as integrate periods where metal discharges have occurred at local and global scale. Govers *et al.* (2014) conducted a meta analysis to compile this benchmark of global trace metal concentrations in seagrass leaves; the values of metals that they found for the majority of the seagrass species that were included in their compilation, were, for Cd 0.1-10 μgg^{-1}, for Cu and Pb 5-80 μgg^{-1}, for Cr 0.5-50 μgg^{-1}, for Ni 1-100 μgg^{-1} and for Zn 5-500 μgg^{-1}; comparatively, the levels of these metals recorded in this study for *R. maritima* were low, close to the lower limit reported by these authors. Brito *et al.* (2016) reported concentrations of several metals on the *Halodule wrightii* seagrass in Brazil; the average values of Cd, Cr and Cu for *R. maritima* from Sontecomapan lagoon were higher while those of Ni, Pb and Zn were lower than those of the Brazilian species.

Another important aspect of including aquatic vegetation in metal contamination assessments is the fact that these species, as primary producers, contribute to nutrients and energy and can transfer their stored metal content by initiating the biomagnification process (Govers *et al.*, 2014). In photosynthetic organisms, toxic metals such as Cd, Cr, Pb, V, As, Hg and others, cause alteration on the photosynthetic mechanism as they decrease the pigments responsible for photosynthesis and damage the process reactions occurring in the presence of light and in obscurity, the membrane structure of chloroplasts and above all, alter the electron transport, lipid metabolism and energy decrease in plants (Navarro-Aviño *et al.*, 2007; Kumar *et al.*, 2014).

Figure 14 shows the results of metal concentrations in the two bivalve species. The highest contents of Zn and Cu were recorded in the oyster *C. rhizophorae* with maxim of 265 and 29.3 μgg^{-1} respectively at site 3 in March, 1992, while for the mussel *B. exustus* recorded the highest contents were 46.62 and 15.1 μgg^{-1} respectively. Co, Cr and Ni showed a reverse pattern, as *B. exustus* had the highest (31, 40.2 and 6.28 µgg^{-1}, respectively); Cd and Pb concentrations were similar. Differences in concentrations of Cu and Ni among similar bivalves species have already been reported in other studies, indicating that they could be significant and surprising; the best-known examples are the differences of metal concentrations in mussels and oysters collected simultaneously at the same locations by the National Status and Trends Programs and the concentrations of Ag, Cd, Cu, Ni, Zn, which tend to bind with the S-containing ligands (e.g., proteins), are typically much higher in American oysters (*Crassostrea virginica*) than in mussels (*Mytilus sp.*), whereas the concentrations of Se, Pb, Hg, Cr, and As in mussels are higher than those found in oysters collected from the same sites (Wang, 2009; De Souza *et al.* 2016).

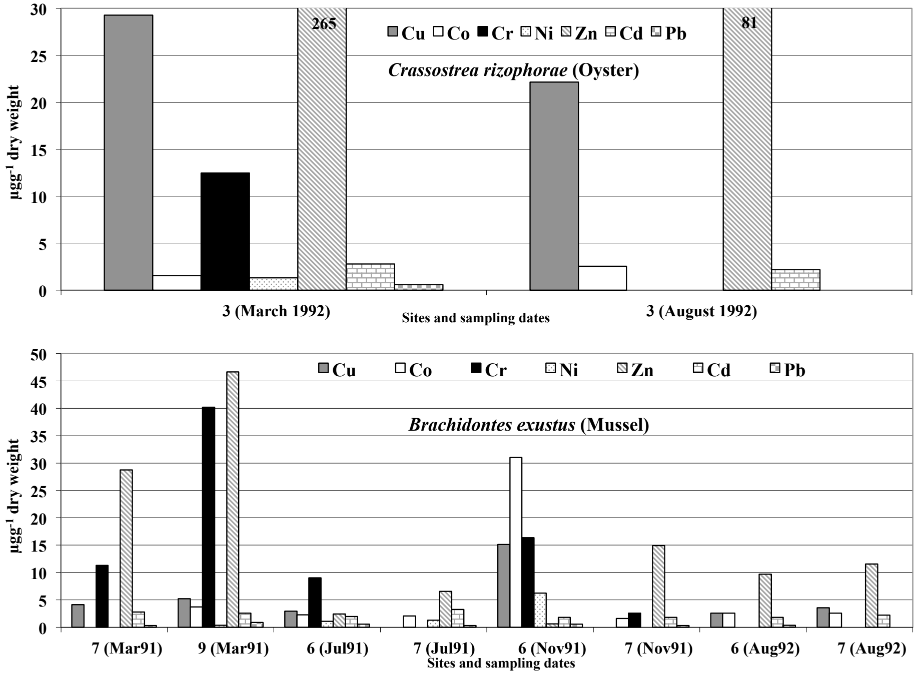

Figure 14. Content of total metals in bivalve mollusks
at Sontecomapan Lagoon, Veracruz. Average values. 1991-1992.

High Cu content in bivalve species (Mussels 12-698 μg g^{-1}; Scallops 3.3-9 μg g^{-1}) is normal since Cu is an essential element and is present in hemocyanin, the blood pigment of these invertebrates, vital for respiration and oxygen transport (Lino *et al.*, 2016). Thus the high concentrations found in this study (4.21 μg g^{-1} for *B. exustus* and 25.72 μg g^{-1} for *C. rizophorae*) can be considered as normal due to the essentiality of this element. The case of Zn, another essential metal for bivalves, is similar to that of Cu, since the content of this element is high in mollusks and plays an important role in the composition of approximately 90 different enzymes of mussels, which justifies significant Zn levels in mollusks from pristine areas (Lino *et al.*, 2016); and in this work, there also were differences between the content of Zn recorded in the oysters and the mussels of Sontecomapan Lagoon, being higher in *C. rizophorae* (172.9 μg g^{-1}) than in *B. exustus* (15.1 μg g^{-1}).

For several decades, bivalve mollusks and other invertebrates have been included in long-term environmental monitoring programs as they accumulate different hazardous substances to a greater extent than other constituents of coastal ecosystems, mainly due to their filtering behavior. Since they are sessile or sedentary organisms, bivalves can provide greater spatial integration than sediment and provide information to identify sources of bioavailable metals. In addition, the metals that these organisms accumulate can reach toxic concentrations and cause biological and ecological damage (Bai *et al.*, 2011). Toxic metals such as Cd, Pb, As accumulated in the tissues of these species can cause from poisoning to decreased fertility, damage and cell death, alteration of various organs and death (El Nemr *et al.*, 2012).

With the full concentrations of metals in organisms, Figure 15 was constructed for global comparison purposes, using the average values for each of the vegetation and bivalves species tested. This allows comparative evaluation of accumulated metal contents since these biota are sessile, permanently exposed to environmental conditions where they live, including exogenous materials of anthropogenic origin, although some metals may be essential, changing their chemical state and attaining potentially harmful concentrations. Zn present in *C. rhizophorae oyster* tissue ranked with 173 ± 129.7 µgg^{-1} while others metals were <25 µgg^{-1} Co and Ni had their highest average concentrations in widgeon grass with 13.51 ± 8.7 and 3.7 ± 1.5 µgg^{-1}, respectively, while the average Cu peaked in the *C. rhizophorae* oyster with 25.7 ± 5.05 µgg^{-1} and Cr recorded its highest average value in the *B. exustus* mussel with 13.7 ± 9.93 µgg^{-1}, followed closely by widgeon grass with 8.21 ± 2.27 µgg^{-1} (Fig. 15).

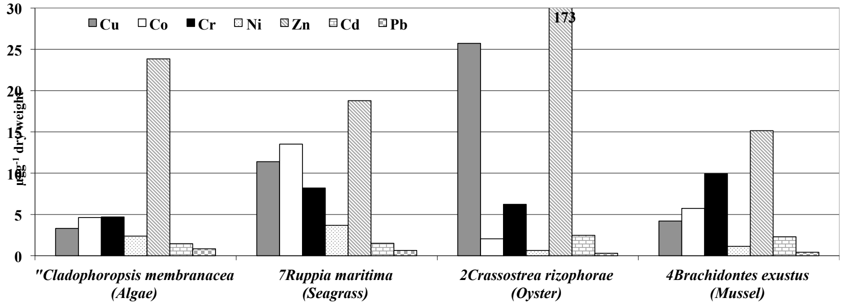

Figure 15. Content of total metals in organisms Sontecomapan lagoon, Veracruz. Average values. 1991-1992.

Table 5 compares the bioconcentration factors (BCF) obtained for the four species of organisms. The *C. membranacea* algae showed lower BCF values than seagrass for Co, Cr, Ni and Cd whereas for Cu, Zn and Pb BCF was higher in chlorophyte than in marine phanerogama; we must highlight the Co BCF value close to 1 in R. maritime, which means that there is practically the same proportion of metal in the sedimentary matrix than in this plant tissue whereas for nickel the low BCF value revealed a very low accumulation. In the case of bivalve mollusks, *C. rizophorae* had high BCF values for Cu (1.17) and Zn (17.57), the latter being of a very high magnitude recorded in the dry season of 1992, probably because this metal is highly used for the formation of its shell. *B. exustus* mussel concentrated higher amounts of toxic metal Cd (2.21) than the oyster which represents a risk to the ecosystem and consumers of this species, since cadmium is currently considered carcinogenic to humans (IARC, 2011).

Table 5. Bioconcentration factors (BCF) of metals estimated for the organisms analyzed in the Sontecomapan Lagoon, Veracruz. 1991-1992

Organisms	Cu	Co	Cr	Ni	Zn	Cd	Pb
Cladophoropsis membranacea (Algae)	0.28	0.24	0.12	0.05	0.56	N.C.	0.11
Ruppia maritima (Seagrass)	0.24	0.97	0.19	0.09	0.40	0.75	0.05
Crassostrea rizophorae (Oyster)	1.17	0.10	0.24	N.C.	17.57	1.65	N.C.
Brachidontes exustus (Mussel)	0.28	0.47	0.34	0.03	0.66	2.21	0.05

N.C. Not Calculated (concentrations not detected)

When organisms absorb heavy metals from the surrounding environment, metabolic mechanisms are initiated to excrete and detoxify them in the case of toxic metals or if the essential elements are in high concentrations in order to avoid harmful effects. Indeed, metals that exceed the amount for the metabolic requirements as well as the accumulation capacity of the species become dangerous, the risk is low or does not exist when the biota stores metals in non-toxic chemical forms. Therefore, the bioaccumulation of metals is a complex process, since there are multiple factors involved such as life stage, biochemical pathways, sediment physicochemistry, type of exposure and biological mechanisms to detoxify and regulate them. However, the tissue content of metals in species is a good measure of these elements bioavailability towards organisms and has long been proposed as a good predictor of metals toxicity in the environment (Fukunaga & Anderson, 2011; Wang *et al.*, 2016).

The bioavailability and potential for bioaccumulation and biomagnification of metals in ecosystem compartments are high. For many species such as bivalve mollusks, ingestion is the main route for acquiring these substances from their environment and at the same time, transferring the metals load through the food chain, making clear the importance of understanding the contaminants dynamics in aquatic environments (Mendoza-Carranza *et al.*, 2016).

Conclusions

The highest total concentrations of metals found in the surface sediments of the Sontecomapan Lagoon were for Cr, Ni and Zn, the first two considered as toxic metals and the latter being an essential element for various biochemical pathways. A large number of total concentrations recorded were higher than the sedimentary quality criteria with a low probability of causing benthic damage, except for Ni since most of the results were higher than the established high probability concentration that cause biological alteration to benthos. The Pb had levels higher than those recorded for Cd in their total concentration, both at the threshold limit to disturb the benthic species. The bioavailable fraction evaluated in the lagoon sediments had an expected pattern since the essential metals were recorded throughout the study cycle while the potentially toxic ones such as Cd and Pb were only detected in the final collections at low concentrations. The content of the metals studied in the various species analyzed showed that marine phanerogams bioaccumulated more than algae and oysters and had a higher content of the essential metals Cu and Zn, while mussels stored at higher levels potentially toxic elements such as Cr and Ni, while the concentrations of Cd and Pb were low in both vegetation and bivalves. The obtained bioconcentration factors showed that sediments are an important source of metals for the oysters, mainly Zn, Cu and Cd and to a lesser degree for mussels and marine phanerogama, for which the monitoring of metals must be maintained through bioindicators analysis and recent sediments in this Mexican lagoon ecosystem.

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Changing trends in mangrove the forest cover of the Sontecomapan Lagoon System, Veracruz, with Landsat® satellite images

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Abstract: The mangrove forest area in lagoons tends to be disturbed by different factors and this leads to ecosystem deterioration. This work analyzes and quantifies the changing trends in mangrove forest cover of the Sontecomapan Lagoon, Mexico, through Landsat® satellite imagery of 1979, 1986 and 2000. In Landsat® images with different RGB compositions, a polygon delineation of the different surfaces was depicted. The segments were transformed into polygons and several classes were identified: Water from the Lagoon System, Mangle, Urban Zone, Agricultural Zone, Other Vegetation Type, Other Type of Coverage or Land Use and Sea Water of the Gulf of Mexico. Maps of persistence, and tables of gains and losses between classes were generated. The results indicated a reduction of Mangrove and Other Vegetation Type, with gain in the Agricultural Zone. An annual loss of mangrove forest area of 422 ha was observed between the extreme dates 1979-2000 (at a rate of approximately 1.64% per year). Due to its commercial value, economic loss was approx. $ 4,194,000 USD ha^{-1} a year^{-1}. The mangrove forest cover's dynamics, tendency and explanation, provide signs about elements that require attention in its management.

Keyword: Land use, Tropical coastal lagoon, Remote sensing, Polygon delineation, Environmental services.

Introduction

The mangrove forest is an ecosystem that works as the interface between the marine and terrestrial worlds. It is especially adapted to saline soils and watery conditions and is distributed geographically in the tropical and subtropical strip, bordering the coast mainly in river mouths, lagoons, estuaries and terrains with flat and muddy relief. In estuaries and islands, this forest is periodically and partially flooded by relatively calm waters, with a small span between high and low tide (Kathiresan & Bingham, 2001). Mangrove species can adapt to different salinity conditions, from brackish to hypersaline water, that's why

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they are called halophyte plants, since they can withstand those changes (Arrueta, 2008). In the world's tropical regions the mangrove forest high productivity is widely recognized. Mangroves provide habitat for fish, mollusks, crustaceans, birds, insects, reptiles and mammals, whose exploitation is carried out by thousands of coastal residents for food and commerce. The mangrove ecosystem also provides environmental services: enhancing water quality in estuaries, protecting coastal areas from flooding, storms and erosion; organic matter production and export to estuarine areas, and retention and accumulation of sediments and heavy metals present in estuary water columns (Bodero & Robadue, 1995). However, because of their location in the coastal zone, mangroves are vulnerable to anthropogenic disturbances and have high rates of deforestation.

In Mexico the mangroves have an extension of 770,057 ha, and extends along the entire Mexican coast. Mexico's mangrove area estimated by INEGI in 1976 was 1,041,267 ha, with 69% located in the Atlantic coast and 31% in the Pacific. The country area delineated as mangrove in the year of 2000 was about 880,000 ha, with 62% located in the Atlantic and 38% in the Pacific. It has been estimated that the annual loss rate of mangrove area on both coasts between 1976 and 2000 was 2.5% and it is expected that, if the same rate continues, in a 25-year period, about 50% of the mangrove in Mexico will be lost (INE, 2005; Calderón *et al.,* 2009).

According to the CONABIO's National Mangrove Inventory (2009b), the published data for assessing the area occupied by mangroves in Mexico have discrepancies in methods, mapping sources and analysis scales, because it does not allow comparisons between different map or date products. These differences do not allow us to know for sure the total surface occupied by mangrove within the national territory. There is also no clear understanding of the factors that have influenced the mangrove changes over the last three decades. There is a wide discrepancy between the several mangrove extension estimates in Mexico since the 1970's. The values range from 440,000 ha to about 1.5 million. The differences are basically due to diversity on methods and analysis scales used in each study. That mangrove forest area estimate (770,057 ha, at a mapping scale of 1: 50,000) only took into account mangrove areas larger than 1 ha. Due to the physical characteristics of Mexican territory, remote sensing techniques were needed in projects focused on natural resources study. The land surface is constantly modified, due to deforestation, human and industrial settlements, roads and more; these modifications can be assessed with the application of remote sensing technology (Soria-Ruiz & Fernández-Ordóñez, 2002; Krause *et al.,* 2004).

Mexico is among the countries with the highest rate of mangrove deforestation in the Americas; however, there is no reliable data on national mangrove cover, making it impossible to accurately estimate mangrove loss rate over time. It is necessary to obtain accurate information to evaluate the changes in the local and regional scales to detect magnitude, trends and agents of the mangrove transformation (Hirales-Cota *et al.,* 2010).

Evaluations on the mangrove cover at the Mandinga lagoon system, (Veracruz, Mexico) resulted in 438.43 ha in 1985 and 351.62 ha in 2005 (1.2% annual loss in a 20 year-period). By 1985, urban use showed 264.68 ha area and by 2005 this urban use was 586.37 ha, which reflects the strong pressure from this growth against the mangrove. Environmental services that were not provided in 20 years by the mangrove was valuated at $ 70.9 million USD; benefits that neighboring communities no longer perceived (Cabrera, 2009). At the Pacific Coast, in the State of Nayarit, Mex., mangrove cover change trends in the Teacapán-Agua Brava lagoon system were evaluated through a multitemporal analysis with Multispectral Scanner (MSS), Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) images from Landsat® satellites, from year 1973 to 2000. A mangrove loss higher than 23,000 ha was observed, mainly due to an agricultural land transition of 40%. The mangrove cover maintained at the same site was estimated at 57,723 ha during the 27 year-period covered by this study, equivalent to 65% of the mangrove area estimated in 1973 (Berlanga-Robles & Ruiz-Luna, 2007).

The lack of an accurate and thorough recognition of the ecological, economic and social importance of mangroves, endorses the loss services provided by these ecosystems (Hirales-Cota *et al.,* 2010). Discordant figures exist on the current deforestation extent and rates, which makes it necessary to assess the current status and mangrove forests change trends. Recording synoptic and recursive coverage at local and regional scales through remote satellite sensing, is a reliable source of information (Berlanga-Robles & Ruiz-Luna, 2007).

The Sontecomapan lagoon system, has an 891 ha area, with a mangrove cover of about 523 ha, and has three phytoplankton species of the six recorded in Mexico: the red mangrove *Rhizophora mangle (Rhizophoraceae),* the black mangrove *Avicennia germinans (Verbenaceae)* and the white mangrove *Laguncularia racemosa (Combretaceae)* (CONABIO, 2009b). This lagoon system is regarded as biologically important because of the connectivity between mangroves, seagrass and coral reefs, thus allowing flow between endemic species. This ecosystem works as habitat: nesting, spawning, feeding, shelter and as nursery area for alevin fish and crustacean larvae; it also provides refuge for flora and fauna and has a high aesthetic, recreational and research value. Due to the ecological functions provided by this lagoon system, it has been included as part of Los Tuxtlas Biosphere Reserve (Gómez-Marín, 2003; CONABIO, 2009a). The human inhabitants of the surroundings use the lagoon for fishing with commercial and subsistence purposes, and as a recreational area (Olguín-Aguilera, 1993) however, these services are threatened by strong anthropogenic pressure, particularly by deforestation.

Therefore, this work's objective is to analyze the mangrove forest cover change trends at the Sontecomapan Lagoon System, as determined from Landsat@ satellite images of the years 1979, 1986 and 2000.

Materials and Methods

Study area

The Sontecomapan Lagoon is located between 18°29.70' and 18°35.39' North latitude, and 94°58.49' and 95°3.85' West longitude (Fig. 1). The lagoon system is divided into several zones: the bar that extends from the beach to Roca Morro, El Real channel that covers La Palma river area and ends in a channel that opens and forms the lagoon, with an average depth of 1.50 m (Olguín-Aguilera, 1993). The lagoon is divided into three zones, due to two deltas formed by the Coscoapan river and the approximated water mirror's area is 891 ha (Portilla-Ochoa, 2005).

The lagoon is about 12 km length and 1.5 km wide, it lies in the basin formed by San Martin volcano and Sierra Santa Marta mountain range, and feeds from several rivers and streams: Palma, Sumidero, Basura, Sontecomapan, Chuniapan, Coscoapan, Old River Coscoapan, El Fraile, Sábalo, Yahualtajapan, Los Pollos and La Boya. These rivers preserve some riverside forests, wet forests, reeds and some high perennial forests. Due to two deltas formed by the Coscoapan river, the lagoon is partially divided into three zones.

From Barra de Sontecomapan and El Real to the Northwest, Jicacal beach extends to the North for about 8 km, inland there is a flat, lowland and floodable area where some mangrove and floodable forest formerly stood, but currently most of it has been converted to pasture field and cropland (Gómez-Marín, 2003). The tidal cycle is diurnal with small tide amplitude (about 70 cm). The lagoon's water level along the year varies from 2 to 3 m due to rain season and floods, also the water quality is affected because of the river drainage; as a result of the basin deforestation the tributaries drag materials, cause erosion, mudslides and water siltation.

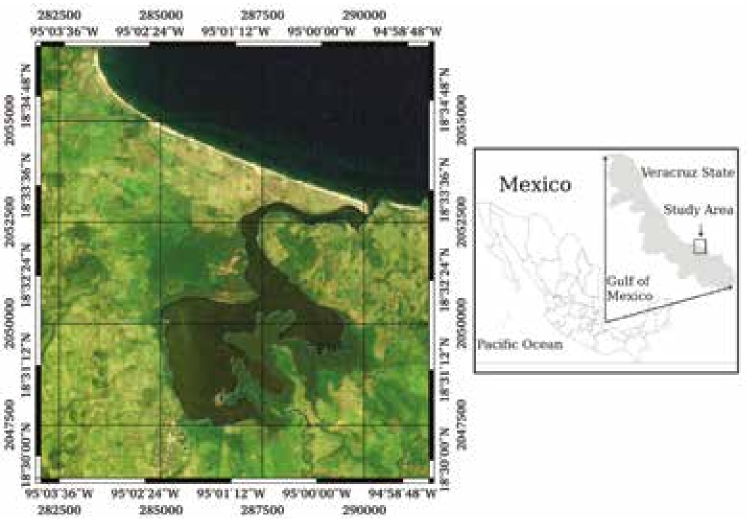

Figure 1. Sontecomapan Lagoon System, Veracruz, Mexico. The lagoon is located in the Southeast coast of the Gulf of Mexico, North America.

According to Köppen's classification the climate of Sontecomapan Lagoon is in Group A: warm, with annual mean temperature over 22 °C with temperature on the coldest month above 18 °C and an average annual temperature of 24 to 26 °C. The extreme upper temperature is 34 to 36 °C, and is usually reached in April and May (the driest months). Precipitation is influenced by humid winds from the Gulf of Mexico, with annual rainfall of 3,000 to 4,000 mm and storms occurring on 20 to 40 days a year (García, 1981).

Currently, the mangrove covers about 523 ha and there is a system of coastal dunes. The fishery in the area is is about 671 mt/year^{-1}, and is equivalent to 0.5% of Veracruz total fish production (Gómez-Marín, 2003). At an economic level, this mangrove is a source of food and income for many families, and is a Natural Protected Area run by the Universidad Veracruzana. The mangrove of Sontecomapan is integrated by three mangrove species, which are *Rhizophora mangle* (red mangrove), *Avicennia germinans (Verbenaceae)* (black mangrove) and *Laguncularia racemosa (Combretaceae)* (white mangrove); the structure, distribution and abundance of these tree species give this mangrove a great biological importance (Carmona-Díaz *et al.*, 2004). Endemic vegetation is composed of high perennial forest, oak forest and deciduous forest. Plant species introduced by man are induced grasses, fruit trees, grains and forages for local consumption (Galván *et al.*, 1999). The Sontecomapan mangrove also functions as habitat for a large number of other plant forms such as mosses, lichens, ferns, non-vascular and vascular epiphytes, lianas and parasitic plants, which makes it very diverse since it is uncommon to find other plants associated with mangrove ecosystems. The presence of vascular epiphytes is a rare feature in mangroves, not only in Mexico but in the rest of the world. The alkaloids and tannins that mangrove trees possess, generally do not allow the establishment of this type of plants, however, in the Sontecomapan mangrove they are widely represented by the families *Pteridaceae, Araceae, Orchidaceae, Cactaceae, Bromeliaceae, Piperaceae, Gesneriaceae* and *Moraceae* (Carmona-Díaz *et al.*, 2004).

Image analysis

Remote sensing is an approach to ecological study and diagnosis that is firmly framed in the scientific method (Lira, 2003). There are several methods for the digital analysis of satellite images, supported by changes in vegetation cover or changes detection in coastal environments, that help to understand the ecosystem dynamics, and to further explain the mangrove forest change trends. Remote sensing images processed using Geographic Information Systems (GIS) technology are a working tool in landscape ecology since they are used for ecological study and diagnosis of an area affected by anthropic intervention (Lira, 2003; Vázquez, 2010). In the GIS environment different thematic maps are worked and edited by interpreting the land use ,thus quantifying the spatial structure according to the degree of landscape modifications (Moizo-Marrubio, 2004; Rosete & Bocco, 2003; Cámara *et al.*, 1996). The high contrast that exists in the energy reflected by vegetation in the red and infrared window of the electromagnetic spectrum is the basis for different quantitative indices used for biomass estimation, environmental changes and identification of soil degradation, among others. The vegetation indices in remote sensing allow to maximize the sensitivity of the plants' biophysical parameters; normalize or model the effects of the sun's angle of incidence, satellite's inclination and atmospheric effect, normalize the soil's variations effects at the bottom of the tree tops, the topography variations; and to relate the information with some measurable biophysical parameters (Vázquez-Andrade & Álvarez-Béjar, 2011).

The different spectral bands recorded are analyzed individually to obtain information about natural resources, but the analysis is enriched with the use of band combinations (Rosete & Bocco, 2003). The acquisition period is crucial for multitemporal analyses, for either detecting changes between two reference dates or to study the plant's cover seasonal cycle (SAGPA, 1998). Changing detection in coastal environments provides indicators of ecosystem dynamics and information on key elements that require special attention in their analysis and management. Different methods for change detection have been developed from the digital analysis of satellite images, all of them based on the premise that changes in the aquatic and terrestrial coverage must generate changes in the images' radiance values that are larger than changes generated by other factors such as differences in atmospheric conditions, sun's axis and soil's moisture conditions (Singh, 1989; Yuan *et al.*, 1998; Berlanga-Robles & Ruiz-Luna, 2007).

The satellite high-quality images needed for the analysis were found on the Landsat.org website, where dates with the characteristics necessary for the study, cloudless and with low moisture in the atmosphere, were found: 1979/02/28, 1986/03/18 and 2000/01/04. The three selected dates correspond to states of the system: 1979, a quasi-pristine state; 1986, the beginning of the enthusiastic urban development and 2000 the beginning of environmental awareness from the government.

The remote sensors used in this study were those of the Landsat® satellite family described below (Table 1): Landsat MSS for 1979 image, Landsat TM for 1986 and Landsat ETM+ for 2000. The path/row (p/r) were 25/47, 23/47, and 23/47 for the pictures of 1979, 1986 and 2000, respectively. These images are part of NASA's contribution to the global research community, and are provided through the Landsat.org website and Tropical Rain Forest Information Center, a member of NASA's Federation of Earth Information Science Partners (Landsat, 2010).

Table 1. Characteristics of the Landsat® satellite images used in this work. Image from year 1979 is a MSS, image from 1986 is a TM y and the 2000 image is an ETM+ (from LANSAT, 2010)

Band	Sensor/Platform					
	MSS / LANDSAT 1-5		TM / LANDSAT 4-5		ETM+ / LANDSAT 7	
	Spectral resolution	Spectral definition	Spectral resolution	Spectral definition	Spectral resolution	Spectral definition
1	80 m	Green: 0.5-0.6 µm	30 m	Blue-Green: 0.4-0.52 µm	30 m	Blue-Green: 0.4-0.52 µm
2	80 m	Red: 0.6-0.7 µm	30 m	Green: 0.52-0.6 µm	30 m	Green: 0.52-0.6 µm
3	80 m	Near Infrared: 0.7-0.8 µm	30 m	Red: 0.63-0.69 µm	30 m	Red: 0.63-0.69 µm
4	80 m	Near Infrared: 0.8-1.1 µm	30 m	Near Infrared: 0.76-0.90 µm	30 m	Near Infrared: 0.76-0.90 µm
5			30 m	Short wave Infrared: 1.55-1.75 µm	30 m	Short wave Infrared: 1.55-1.75 µm
6a			120 m	Long wave Infrared: 10.4-12.5 µm	120 m	Long wave Infrared: 10.4-12.5 µm
6b					60 m	Long wave Infrared: 10.4-12.5 µm
7			30 m	Short wave Infrared: 2.08-2.35 µm	30 m	Short wave Infrared: 2.08-2.35 µm
8					15 m	Panchromatic: 0.52-0.90 µm

Based on information from the study area of the Sontecomapan Lagoon and its surroundings, including the Los Tuxtlas region, a working legend was developed (Table 2), which allowed the cover class assignments to perform an interpretation of what was detected from the Landsat satellite. Hybrid maps were created from the combination of satellite imagery and land cover class maps from representative samples of known objects present in the image according to field tests. Classification was performed once the images' spectral classes were defined, that is, each element of the scene or pixel was classified, assigning maps' elements or more related photos (Bocco *et al.*, 1991).

Table 2. Work legend used in class type classification. First column: Arabic number is the class number for the named class, each class has an assigned color, second column is the type and/or the land use

Class	Vegetation type and/or land use
1. Water/Blue	Lagoon System Water
2. Mangrove/Green	Red Mangrove Black Mangrove White Mangrove
3. Urban Zone/Gray	Villages
4. Agricultural Zone/Brown	Agriculture Cattle ranch Aquiculture
5. Other Type of Vegetation/Orange	Riverside woods Moist evergreen rainforest. Low land flood forest
6. Other Type of Coverage or Land Use/Yellow	Routes of terrestrial communication Flood zones Sand/beach/eroded soil
7. Water/Royal Blue	Gulf of Mexico sea
8. Less than 4 ha.	Less than 4 ha polygons

A classification system was used for the different classes that were assigned to the process of each of the satellite images from the different dates. The land use of the study area was reviewed and a hierarchical system was generated. This working legend (Table 2) allowed us to solve three problems: (1) the coverage classification at various levels of spatial resolution, (2) a hierarchical organization covering the study area that includes defined criteria for each of the seven aggregation levels, and (3), an objective representation of vegetation dynamics, to include classes in continuous transformation as to the change direction of each one. That's why this working legend was adopted as the baseline for land-use/land-cover change estimates. According to the definition given by Singh (1989), detection change is: "the process of identifying differences in the state of an object or phenomenon by observing it at different times".

Given the small size of the study area, it was decided to carry out a visual interpretation of the satellite images. The steps performed were as follows:

- Images from three dates, 1979 MSS, 1986 TM and 2000 ETM+, were downloaded from Landsat.org and read with the IDRISI® ANDES v15 software.
- The images were georeferenced, while a subscene of the originals was obtained, which allowed homogeneity among the images to be processed.
- With each date's subscene, different vegetation indexes or transformations (Normalized Difference Vegetation Index [NDVI], PCA, Tasseled Cap) were generated, which were used later to obtain RGB compositions. The generated combinations were compared to pick the best choice to evaluate land use classes and vegetation of the area.
- The ILWIS® software v. 3.4 was used to develop land use maps use and vegetation. RGB compositions were imported to ILWIS considering UTM coordinates (zone 15Q) and WGS84 spheroid. For the visual interpretation phase, a digital polygon delineation (segmentation) of the observed surfaces on the different RGB images was performed for each year. Study area segmentation for each year was done at a 1:30,000 scale, and the final images were built at a 1:100,000. The minimum mapping unit was stated at 1 ha (10 000 m^2).
- From these images, each polygon's assignment to its corresponding class proceeded (previously established in the work legend). Class assignments were corroborated with INEGI maps and Google Earth images, and a raster image was generated for each date.
- The previous generated images of segments and polygons were in vectors, and each one of them had its own class legend. These images were exported to raster format and read again in IDRISI for its final processing. As this process eliminated smaller areas (polygons) of less than 4 ha, this step generated one more class, which was called "Less than 4 ha".
- The classes and colors used were: Lagoon System Water (Blue), Mangrove (Green), Urban Zone (Gray), Agricultural Zone (Brown), Other Type of Vegetation (Orange), Other Type of Coverage or Land Use (Yellow) and Gulf of Mexico Sea Water (Royal Blue) (Table 2).
- With the IDRISI application "Land Change Modeler for Ecological Sustainability" a project with the images from each date was created, but only a pair can be compared

at time (Eastman, 2006). Therefore, three groups were generated: first, from 1979 to 2000 (extreme dates), second, from year 1979 to year 1986 and, third, from year 1986 to year 2000. The last two groups were called "intermediate dates". This function calculated the changes in ground class coverage (changes in the work legend) for each period. In this process, only seven classes were considered: Water from the Lagoon System, Mangrove, Urban Zone, Agricultural Zone, Other Type of Vegetation, Other Type of Coverage or Land Use, Sea Water from the Gulf of Mexico.

- From the extreme and intermediate dates the change analysis was obtained. Class gains and losses, net change per class, contributions to the net change expressed by each category and persistence were evaluated. From this information, gains and losses tables from one category to another, and persistence were generated.

Results

The results obtained from the analysis of Landsat satellite images from three dates are shown below. First, the change direction of each class is shown, and then the coverage of each class with respect to the extreme dates, from year 1979 to 2000, and on intermediate dates, from year 1979 to 1986 and from year 1986 to 2000.

Table 3 shows the loss, gain and net change in coverage (in hectares) of each class for the intermediate dates range from 1979 to 1986. The class with the greatest area loss was Other Type of Vegetation, with 1,473 ha, and a gain of 579 ha giving a net change of -893 ha (the minus sign means loss). In this interval the mangrove cover lost 325 ha, gained 144 ha, and the net change was -181 ha; the Agricultural Zone lost 311 ha, gained 1,345 ha, and the net change was 1,034 ha; the Urban Zone lost 11 ha, gained 71 and the net change was 60 ha.

Table 3. Loss, gain and net change in coverage, in hectares (ha), of each class for the intermediate dates range from 1979 to 1986

Classes	Lost (ha)	Gain (ha)	Net Change (ha)
1 Lagoon System Water	-31	59	28
2 Mangrove	-325	144	-181
3 Urban Zone	-11	71	60
4 Agricultural Zone	-311	1,345	1,034
5 Other Type of Vegetation	-1,473	579	-894
6 Other Type of Coverage or Land Use	-103	54	-49
7 Gulf of Mexico Sea Water	0	2	2

Table 4 shows the contributions in hectares of each class to the net change in the intermediate dates of 1979-1986. These were as follows: Class 2 Mangrove yielded 24 ha to class 1 Water of the Lagoon System. The class 4 Agricultural Zone removed 49 ha from class 2 Mangrove. Class 5 Other Type of Vegetation removed 113 ha from the class 2 Mangrove. Class 5 Other Type of Vegetation yielded 966 ha to 4 Agricultural Zone.

Table 4. Class contributions of coverage, in hectares (ha), to the net change expressed by class for the intermediate dates of 1979-1986

Classes	Contribution (ha)						
	1	2	3	4	5	6	7
1 Lagoon System Water	0						
2 Mangrove	24	0					
3 Urban Zone	0	-1	0				
4 Agricultural Zone	-1	-49	-4	0			
5 Other Type of Vegetation	2	-113	59	966	0		
6 Other Type of Coverage or Land Use	3	5	4	14	21	0	
7 Gulf of Mexico Sea Water	0	0	0	0	0	2	0

Table 5 shows the loss, gain and net change in coverage of each class for the intermediate dates range from 1986 to 2000. The class with the greatest area loss was 5 Other Type of Vegetation, with 1,644 ha, and gained 620 ha, resulting in a net change of -1,024 ha; the class 2 Mangrove cover lost 263 ha, gained 80 ha, and the net change was -183 ha; class 4 Agricultural Zone lost 471 ha, gained 1,588 ha, and the net change was 1,117 ha; Urban Zone lost 8 ha, gained 50 and the net change was 42 ha.

Table 5. Loss, gain and net change in coverage, in hectares (ha), of each class for the intermediate dates range from 1986 to 2000

Classes	Lost (ha)	Gain (ha)	Net Change (ha)
1 Lagoon System Water	-27	26	-1
2 Mangrove	-263	80	-183
3 Urban Zone	-8	50	42
4 Agricultural Zone	-471	1,588	1,117
5 Other Type of Vegetation	-1,644	620	-1,024

Table 5. (Continuation)

Classes	Lost (ha)	Gain (ha)	Net Change (ha)
6 Other Type of Coverage or Land Use	-91	129	38
7 Gulf of Mexico Sea Water	0	11	11

Table 6 shows each class contributions to the net change, in the time interval of the intermediate dates of 1986-2000. These were as follows: Class 2 Mangrove yielded 8 ha to Water of the Lagoon System. Agricultural Zone took 18 ha from class 2. Other Type of Vegetation removed 150 ha from Mangrove. Other Type of Vegetation yielded 1,123 ha to Agricultural Zone, etc.

Table 6. Contributions, in hectares (ha), to the net change in coverage expressed by class for the intermediate dates of 1986-2000

Classes	Contribution (ha)						
	1	2	3	4	5	6	7
1 Lagoon System Water	0						
2 Mangrove	8	0					
3 Urban Zone	0	-2	0				
4 Agricultural Zone	0	-18	9	0			
5 Other Type of Vegetation	-2	-150	31	1,123	0		
6 Other Type of Coverage or Land Use	-6	-5	0	-15	-22	0	
7 Gulf of Mexico Sea Water	1	0	0	0	0	10	0

Table 7 shows the loss, gain and net change in coverage for each class in the range of extreme dates. The class with the greatest loss of area was Other Type of Vegetation with 2,528 ha, and gained 611 ha, this is a net change of -1,917 ha. Mangrove cover lost 422 ha, gained 57 ha, and the net change was -365 ha. Agricultural Zone lost 304 ha, gained 2,455 ha, and the net change was 2,151 ha. Urban Zone lost 7 ha, gained 109 and the net change was 103 ha.

Table 8 shows the contributions of each class to the net change during the extreme dates interval. These were as follows: Mangrove yielded 32 ha to Water of the Lagoon System. Agricultural Zone removed 54 ha from Mangrove. Other Type of Vegetation removed 277 ha from Mangrove. Other Type of Vegetation yielded 2,102 ha to Agricultural Zone, etc.

Table 7. Loss, gain and net change in coverage, in hectares (ha), of each class for the extreme dates from 1979 to 2000

Classes	Lost (ha)	Gain (ha)	Net Change (ha)
1 Lagoon System Water	-34	60	27
2 Mangrove	-422	57	-365
3 Urban Zone	-7	109	103
4 Agricultural Zone	-304	2,455	2,151
5 Other Type of Vegetation	-2,528	611	-1,917
6 Other Type of Coverage or Land Use	-123	112	-12
7 Gulf of Mexico Sea Water	0	13	13

Table 8. Coverage contributions, in hectares (ha), to the net change in coverage expressed by class for the extreme dates from 1979 to 2000

Classes	Contribution (ha)						
	1	2	3	4	5	6	7
1 Lagoon System Water	0						
2 Mangrove	32	0					
3 Urban Zone	0	-4	0				
4 Agricultural Zone	0	-54	6	0			
5 Other Type of Vegetation	0	-277	88	2,102	0		
6 Other Type of Coverage or Land Use	-5	2	4	2	-4	0	
7 Gulf of Mexico Sea Water	-1	0	0	0	1	13	0

Table 9 shows the coverage in hectares of each class and on each date. It also shows the difference between intermediate and extreme dates. Class 2 Mangrove presented a coverage area of 1,245 ha in 1979, an area of 1,063 ha in 1986 and an area of 880 ha in year 2000. The loss in its coverage area from 1979 to 1986 was 181 ha; from years 1986 to 2000 a reduction of 183 ha was observed, and between 1979 and 2000 the Mangrove cover decreased by 364 ha. The class 4 Agricultural Zone had a loss of coverage of 1,590 ha in the intermediate dates from 1979 to 1986, gained 1,116 ha in the intermediate date from 1986 to 2000, and between extreme dates the area had a loss of 473 ha. The class

5 Other Type of Vegetation, in the range of extreme dates (1979-2000), had a gain of its coverage of 707 ha.

Table 9. Total coverage, in hectares (ha), of each class in each year. The differences between the evaluated years are also shown

Classes	Total Coverage (ha)			Differences (ha)		
	1979	1986	2000	1979-1986	1986-2000	1979-2000
1 Lagoon System Water	895	922	921	27	-1	26
2 Mangrove	1,245	1,063	880	-181	-183	-364
3 Urban Zone	51	111	154	60	42	102
4 Agricultural Zone	3,806	2,215	3,332	1,590	1,116	2151
5 Other Type of Vegetation	1,181	2,912	1,889	1,730	-1,023	707
6 Other Type of Coverage or Land Use	299	249	287	-49	37	-11
7 Gulf of Mexico Sea Water	2079	2,081	2,092	2	11	13
TOTAL (ha)	9,556	9,553	9,555			

With the results of gains, losses and net changes in the study area, it was found that there was a persistence for each of the classes regarding the different dates, which were reduced over time from 1979 to 1986 and 2000 According to one date to another, the study area has undergone a transformation of its classes (Fig. 2). The mangrove had a persistence of 73.86% (919 ha) for the period 1979-1986, for the period 1986-2000 to 75.26% (800 ha), and finally in the period 1979-2000 it had a persistence of 66.13% (823 he has). The same is found with a decrease Other Type of Vegetation that presented 61.31% (2,333 ha), 43.56% (1,268.92 ha) and 33.57% (1,277 ha) for the periods 1979-1986, 1986-2000 and 1979-2000, respectively. With the Agricultural Zone class, a persistence of 73.65% (870 ha), 78.74% (1,744 ha) and 74.27% (877 ha) was obtained for the periods 1979-1986, 1986-2000 and 1979-2000, respectively.

Discussion

The land cover/use change analysis is usually particular or include several ground types. In the first case, as long as there is a suitable reflectance of the ground type to be analyzed over time, and the spectral signature is known, it is possible to make a temporal analysis of how much this land cover type changes. Long & Skewes (1996) used only two classes, one for mangrove and the other for water/land. In another context, if several factors or pressures are present, it will be necessary to consider several types of land coverages to see their evolution over time. The selection of classes is not always easy; minimum area requirements, good reflectance and a well-defined spectral signature are required. Berlanga-Robles and Ruiz-Luna (2007) used four kinds of natural wetlands, an artificial wetland and a land cover class; with ground truth (training areas) generated spectral signatures for each class.

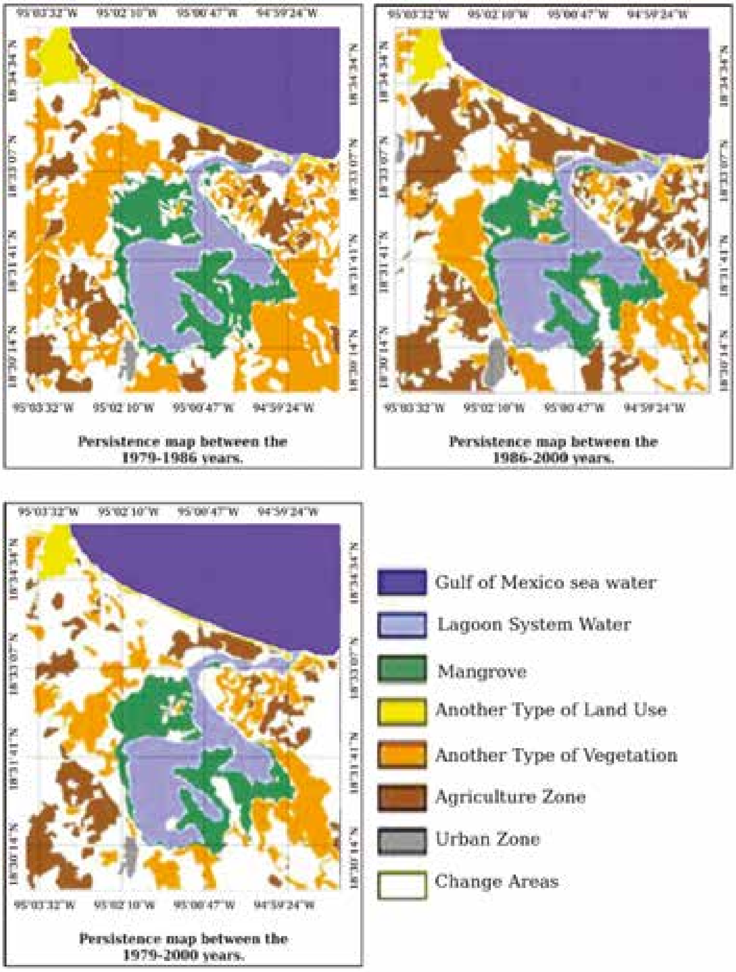

Figure 2. Class coverage change around Sontecomapan, Veracruz, through the three recorded images dates (1979, 1986, 2000).

In the present work it was considered that the pressure or implication of several land cover classes were necessary to understand the process of land use change around the

Sontecomapan lagoon, for this the following classes were included in the work legend: Sea water of the Gulf of Mexico, Water of the Lagoon System, Mangrove, Other Type of Coverage or Land Use, Other Type of Vegetation, Agricultural Zone and Urban Zone. With these classes it was possible to differentiate the change occurred in the study area between different years: 1979, 1986 and 2000. The software tools allowed to observe and quantify the transformation of the mangrove's forest coverage in each class, as well as the coverage transformation of the different classes.

The changes' analysis observed on the vegetation coverage, including deforestation, degradation and revegetation, among others, caused by the anthropic activities, is a fundamental element in the landscape characterization and is expressed as the persistence (Palacio-Prieto *et al.,* 2004). Since the change rate in vegetation has been considered as one of the anthropic influence indicators, then the rates of deforestation of mangrove can be assessed, and thus the disturbance by different factors (pressures) that lead to the deterioration of the ecosystem. This supports the use of vegetation change rates as the basis for land use ordination and management policies (Blanco-Libreros & Estrada-Urrea, 2015). The results indicated that a reduction of Mangrove and Other Vegetation Type persisted, with gain in the Agricultural Zone (based on Tables 2, 4, 6 and 9 it is shown that between 1979 and 2000 agricultural zone increased whereas other type of vegetation decreased). A mangrove forest area total loss of 422 ha was observed between the extreme dates 1979-2000 (1.64% per year approximately); this implies that the mangrove forest coverage dynamics, its tendency and explanation, provide a sign about elements that require attention in its management; if its present trend (1.64% per year loss) is not reverted, a loss of additional 247.8 ha in mangrove coverage in the 2020 year can be predicted.

In a period of 21 years (1979-2000) the mangrove of Sontecomapan had lost an area of 364 ha which correspond to a 17.3 ha/year loss rate; the total mangrove coverage for this last year was of 880 ha. Digital classifications were made to obtain maps showing the mangrove's spatial distribution in the zone, and the proposed classes used seem to be representative for the analysis. Observing the spatial patterns of mangrove change, it is possible to report the different causes and processes that have generated its degradation, fragmentation and loss. Thus, for the periods 1979-1986-2000, the classes that increased were the Agricultural Zone (including cattle growth) and the Urban Zone, indicative of the anthropic influence, while the trend for coverage area was negative, that is loss for the classes Other Type of Coverage or Use and Mangrove forest. These results indicated a disturbance in the study area whose causes obey, as expected, to the great economic activity of this lagoon that has been transforming the landscape in the last 21 years. Despite the satellite images limitations used in this work, coherent results were obtained and were confirmed in the field trip. Krause *et al.* (2004), in a similar change of land use study in Brazil, a loss rate of 17 ha/year^{-1} of mangrove was computed; they found that agricultural and cattle growth classes were the main cause of this loss. They also pointed that results are useful elements for making territorial decisions in their area, in addition to show the geographical distribution of mangroves and other types of vegetation, and establish the basis for a permanent monitoring of the mangrove.

The deforestation rate, as one of the most important indicators of the territory natural conditions, is very high. Pristine-condition areas, like forest soil and water, are reduced due to the anthropic influences which generate a transformation in the ecosystem, causing a high concern since the ecological services that the mangrove yields are diminishing. Due to its use value of \$8,535 USD ha^{-1}/year^{-1} (Pascal, 2014), ecological services (or environmental services) like protection of the shrimp larvae or fish alevins that sustain fisheries, and coast protection from storms, the total economic loss for this 21-year period due to the 17ha/year loss is about \$3,053,225 USD.

Acknowledges

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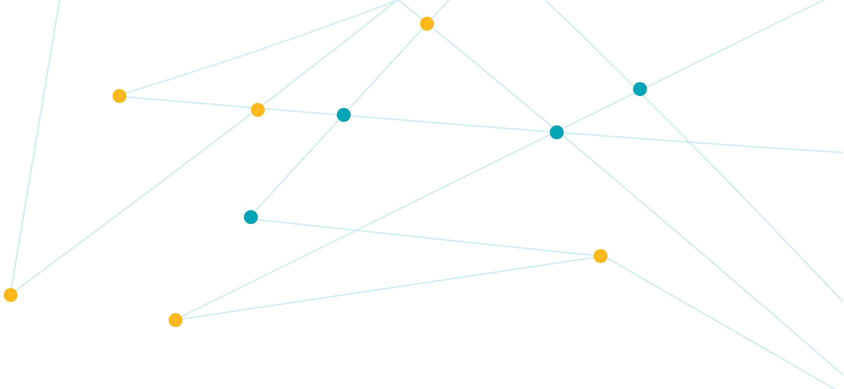

GENERAL DISCUSSION

Main ecological traits of the Sontecomapan Lagoon

The editors*

Keywords Biodiversity, ecological functioning, conceptual ecosystem model, tropical coastal lagoon

1. Hydrochemical characteristics and trophic status

Time and space variability of hydrologic and trophic characteristics of the lagoon were described in detail by Esquivel & Soto-Castor, and by Aguirre-León *et al.* and Figueroa-Torres *et al.* (this issue). They were also briefly documented in some other chapters. All studies agreed that Sontecomapan Lagoon can be divided into three main "habitat zones", mainly driven by their spatial and temporal exposition to tidal exchanges with the sea, freshwater inputs, their morphology and their isolation degree, these are:

- an euhaline zone, in the northern part between La Palma River and the communication with the Sea, characterized by the highest marine influence.
- a mesohaline zone located in the southern inner part of the lagoon with highest freshwater influence.

- an intermediate polyhaline zone with combined effects of marine and freshwater influence.

The contours of these three zones vary seasonally with an extension of the mesohaline zone in the rainy season. Also, Figueroa-Torres *et al.* (this issue) point out that the area near Río Sábalo, which corresponds to the more confined zone, clearly distinguished from all other zones in all seasons.

Esquivel & Soto-Castor (this issue) show that the water column is generally well mixed and poorly stratified in most part of the lagoon, except in the area of La Palma due to the combined influence of tide and river flow provoking episodic salt wedge situations.

The trophic status of the lagoon varies spatially and temporarily. Based on total phosphorous indexes, the lagoon may be considered as mesotrophic to hypertrophic whereas it ranges from oligotrophic to eutrophic according to chlorophyll *a* values (Esquivel & Soto-Castor; this issue). Inter-annual variations of trophic status are pointed by Esquivel & Soto-Castor (this issue). By comparing historical chlorophyll *a* values they show that values for 2009-2010 were the lowest for the period 1990-2010, whereas N-nutrients were within the ranges and soluble phosphorous (SRP) was at higher concentrations than values reported for the same period. They hypothesize that the diminution of chlorophyll *a*, was probably related to exceptional rainfall during this period, especially during their September survey, which probably caused a flushing out of suspended materials to the sea.

2. Living communities - taxonomic richness

Castellanos-Páez *et al.* (this issue) reviewed about 59 papers between scientific articles and book chapters published between 1976 and 2015 and updated the checklists of the different biota of Sontecomapan lagoon. They could establish several species checklists.

- The list of flora (phytoplankton and plants) includes 431 species, where 403 are phytoplankters, 4 macroalgae, 1 seagrass, 3 mangrove, 7 species of coastal vegetation, 1 fern, 2 legumes, 1 liana, 2 bromeliads, and 7 orchids. The list of phytoplankton was also updated by Figueroa Torres *et al.* (this issue) who reported 357 species from a review of the literature and samples collected by them in 2015/2016.
- The list of invertebrate species has a total of 201 species of which are: 5 platyhelminthes, 6 trematodes, 2 nematods, 69 rotifers, 8 acantocephalans, 69 molluscs and 39 crustaceans (1 pentastomid, 3 cladocerans, 14 copepods, 1 rhizocephalan, 1 tanaidacean and 19 decapods), 3 insects (odonata).

- The list of vertebrates represents a total of 185 species of which are: 159 birds, 14 fishes hosting parasites, 1 amphibian, 8 reptiles and 3 mammals. In another paper of this issue, Castillo-Rivera and Lara-Domínguez report 115 fish species in Sontecomapan, of which 4 species are not included in the list of the above14 species hosting parasites: *Diapterus olisthostomus, Lujtanus cyanopterus, Trachinotus carolinus* and *Scomberomorus maculatus*. Thus, the actual list should include 119 fish species and the total number of vertebrate species would be 294.

Taking account of these different contributions, the current number of living species recorded in the Sontecomapan ecosystem is 926.

3. Ecological functioning

From different contributions of this book, we could set a preliminary conceptual model of the trophic flows in the Sontecomapan Lagoon (Figure 1).

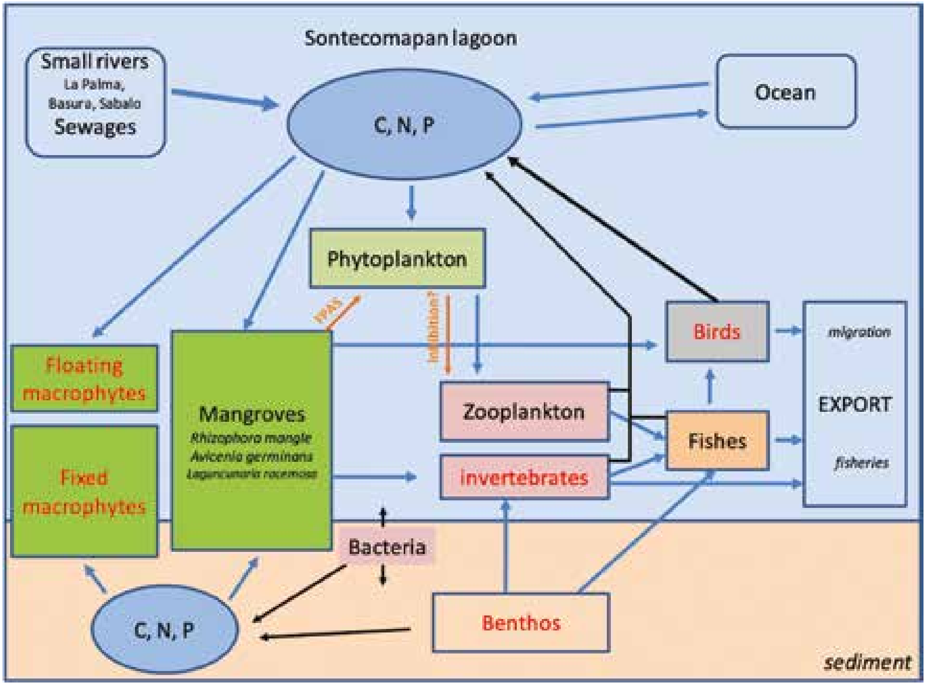

Figure 1. Conceptual model of trophic flows in the Sontecomapan Lagoon. In red characters are the biotic compartments not treated in this monograph. Blue arrows illustrate the trophic flows. Black arrows indicate recycling process (excretion). Orange arrows illustrate inhibitory relationships.

The Figure 1 illustrates the main pathways between the different trophic compartments identified for this model.

The main driving forces of the systems are rainfall (and associated runoff) and tidal exchange which condition the C, N and P equilibrium in the lagoon. The balance between these forces is important, along with wind stress that determine the complete mixing up of water column at most of the lagoon and are responsible for the changing horizontal zonation at different seasons. A further driving force comes from water discharge or run-off from human settlements or other anthropic activities. In non-disturbed conditions, P enrichment should occur from seabird droppings at their perching sites in mangroves and coastal fish P-excretion. Recycling by metazoans which may be several fold higher than that of bacteria (Kremer & Nixon 2012) is an important process to be considered, but data are lacking to estimate the importance of these processes in Sontecomapan lagoon. Mangroves also appear to have a very important role (see below) and Hernández-Gaona *et al.* (this issue) also point out to mangrove area loss as a factor determining the possible Sontecomapan's future states. As in many shallow water systems, the primary production of the lagoon is shared between phytoplankton, floating and fixed macrophytes, mangrove and phytobentos (not shown). In such ecosystem, the balance between these different types of primary producers for the use of nutrients is important for controlling eutrophication. Generally, the importance of macrophytes which compete with phytoplankton for nutrients is a guarantee against the deterioration of water quality and shift between alternative stable states, i.e. between clearwater and turbid water (Scheffer *et al.*, 2001). Also plant beds host macrofiltrators such as mussels and microfiltrators, which may help reducing turbidity of the water within the plant beds.

However, for Sontecomapan, information on macrophytes and phytobentos is lacking. Indirect information by González-Fierro and Ponce-Vélez (this issue), suggests that the seagrass (*Ruppia maritima*) should be important. Invertebrates should also play an important role as the mussels (*Brachidontes exustus*), the oyster (*Crassotrea virginica*), the white shrimp (*Litopenaeus vennamei*), the lobster (*Machrobrachium rosenbergii*) which have commercial interest. Another important issue is to estimate matter export through fishing and birds. Exportation of matter by migrating bird has been shown to be a very important process in lakes or coastal lagoon. For example in the Ichkeul lagoon (Tunisia) where the macrophyte biomass is consumed by the herbivorous wintering waterfowl between October and March, the rather atypical, oligotrophy seems to be strongly linked to the exportation of the nutrients by the migrating birds at the end of the winter (Casagranda *et al.*, 2006).

Despite the lack of information on several trophic compartments, our results highlight some important characteristics of the ecological functioning of the lagoon, as detailed on the following points

The strong variability of nutrient availability drives the plankton dynamics

Nutrients concentrations in the lagoon are driven by autochtonous sources as the mangrove leaf, the water exchanges with the ocean and through the freshwater inputs of the small rivers (mainly La Palma River). Seasonal variation is important with phosphorus peak occurring in early rainy season (June) and nitrogen peak centered on September (late rainy season) (Esquivel & Soto-Castor, this issue). Negative relationship with oxidative forms of nitrogen (NO_3, NO_2) suggests important oxidization of NH_4 into NO_x. Another important source of dissolved mineral nitrogen is the decomposition of the mangrove litters (Aké-Castillo and Vázquez, this issue). At the water - sediment interface, the nutrient availability is also highly dependent on the bacterial activity which drive incorporation and loss processes of N species (i.e. to nitrogen fixation and denitrification) in the bottom water and in the superficial sediments (Ferrara-Guerrero *et al.*, this issue). Due to the shallowness of the lagoon, these processes are highly dependent of climatic conditions. In "Nortes" season N_2 fixation is highest, due to optimal oxic conditions and high organic matter concentrations in the sediments resuspended by wind creating microhabitats with aerobic and anaerobic spaces favoring the bacterial processes. During the rainy season, low salinity and high NO_3 concentration favor the denitrification process.

Then high spatial and temporal variability also drive the dynamics of phytoplankton communities (Figueroa Torres *et al.*, this issue). During the rainy season freshwater inflow decreases the salinity and fertilizes the lagoon in nutrients stimulating new primary production and promoting a phytoplankton community dominated by the chlorophytes. In the dry season, high concentrations in ammonium and high bacterial biomass suggest remineralization processes favoring regenerated production leading to the development of a more diversified phytoplankton community characterized by chrysophytes, cryptophytes, diatoms and dinoflagellates. During the North-wind season high phosphate concentrations may explain the development of another phytoplankton community characterized by euglenophytes and prasinophytes. These seasonal changes in phytoplankton affect the composition and size-structure of zooplankton communities showing the structuring character of trophic conditions on these communities (Benítez-Díaz Mirón *et al.*, 2014; Benítez-Díaz Mirón *et al.*, this issue).

The mangrove plays an important ecological role but is threatened by Human activity

The banks of the lagoon are covered with red (*Rhizophora mangle*) and black (*Avicenia germinans*) mangrove which may play an important role in the ecological functioning serving as refuge for many species (bird, fish, molluscs, crustaceans, etc.). In this monograph, the work by Aké Castillo and Vázquez (this issue) showed that the mangrove also play an important role in the cycle of biogenic elements. These authors refer that NH_4 and NO_3 are quickly released during the early stages of litter decomposition in relation with a rapid mineralization, contrarily to phosphorous form (PO_4). This suggests a stimulating and selective effect of mangrove litter on primary production. However, litter decomposition has also an antagonist negative impact on phytoplankton through release of folin phenol active substances (FPAS) which can inhibit the growth of some species, resulting in a selective impact on phytoplankton community. In particular, some species (*Chaetoceros muelleri* var. *subsalsum*, *Cyclotella cryptica*, and *C. meneghiniana)* were shown to be sensitive to high concentrations of FPAS. *Skeletonema subsalsum* was able to tolerate moderate concentrations of FPAS. These responses support the hypothesis that tolerance to organic compounds in natural systems influence the dynamics of phytoplankton communities.

It was seen that the area of mangrove forest in the lagoon tends to be disturbed by anthropogenic factors, leading to a deterioration of the ecosystem. Hernández-Gaona *et al.* (this issue) using Landsat satellite imagery showed a decreasing trend of mangrove area linked to increase of agriculture zone. The annual loss of mangrove forest area (422 ha; 1.64% per year between 1979 and 200) has also strong economic impact with a loss of approx. $ 4,194,000 USD ha^{-1} year^{-1}.

Low food transfer efficiency at the basis of the food-web

The very low mean zooplankton/phytoplankton biomass ratio, compared to literature data, reveals an overall low food-transfer at the basis of the food chain (Benítez-Díaz Mirón *et al.*, this issue). This low transfer would be partially due to a high predation impact by zooplanktophagous on larger zooplankton (calanoid copepods) as suggested by the zooplankton size-structure dominated by small organisms. This high predation, that lowers the herbivorous biomass, seems particularly important during the north wind season perhaps in relation with the breeding cycles of important fish species as the catfish *Cathorops aguadulce* very abundant in the lagoon (Aguirre-León *et al.*, this issue). The low food transfer between phyto- and zooplankton may also result from inedibility and/or toxicity of a large part of phytoplankton with large proportion of dinoflagellates and diatoms, of which several species may be toxic to zooplankton (Figueroa-Torres *et al.*, this issue).

Benítez-Díaz Mirón et al. (this issue) also show that bottom-up forces affect phytoplankton (NH_4 limitation) and zooplankton (phytoplankton limitation), whereas the top-down control by herbivorous zooplankton on phytoplankton is very low (<10% of the phytoplankton stock day^{-1}). As a consequence, such low exploitation of the phytoplankton production by zooplankton, should foster the sedimentation of un-grazed dead cells and the accumulation of organic matter on the bottom that could lead to the silting up of the lagoon and anoxia events. This tendency, which is also favored by the use of fertilizers and pesticides and associated eutrophication, would cause a degradation of the ecosystem structure and functioning. However, the high rate of water exchange with the ocean and the resulting low water renewal time in the lagoon should preserve the lagoon from these degrading trends, with the exception of the confined areas that must be surveyed and monitored in priority. In addition, tidal inputs of large-sized marine coastal zooplankton (*e.g. Paracalanus aculeatus*, *Temora turbinata* and *Centropages velificatus*) may compensate the "erosion" of large herbivorous zooplankton in the region close to the communication channel thus lowering the negative effects of predation by fish.

Environmental drivers of living communities: important role of salinity

As discussed above, space and time variation of environmental factors in the Sontecomapan Lagoon are driven by seasonal patterns and by spatial and temporal exposition to tidal exchanges with the sea and to freshwater inputs. All studies dealing with functional aspects of the living communities considered in this monograph (bacteria, phytoplankton, zooplankton, mangrove and fish) point out the essential role of salinity in driving their dynamics.

As show by Ferrara-Guerrero *et al.* (this issue) salinity and nutrient availability (NO_3) condition the balance between N_2 fixation and denitrification processes driven by bacterial activity with, low salinity and high NO_3 favoring the denitrification.

The dynamics of phytoplankton communities is also highly dependent on nutrient availability and salinity (Figueroa-Torres *et al.*, this issue). High NO_3 concentration and low salinity favor the development of chlorophytes during the rainy season whereas high ammonium concentrations and high salinity promote a more diversified phytoplankton community during the dry season. However, other physical and chemical factors such as pH, temperature and dissolved oxygen may generate particular microenvironmental conditions driving the distribution and abundance of phytoplankton species in relation with their tolerance capacities.

Nutrients and salinity may also play an indirect role in structuring the zooplankton community, through their impact on their food resource (phytoplankton), as the study by Benítez-Díaz Mirón *et al.* (this issue) shows the high dependence of zooplankton on food concentration. These authors could not evidence a direct link between salinity and zooplankton biomass composition and size structure, perhaps in relation with the euryhaline character of the dominant species (*Acartia clausi*). However, in a previous study (Benítez-Díaz Mirón *et al.*, 2014), they showed that transparency, salinity, temperature pH, as well as food availability were the main determinants of zooplankton abundance, composition, and diversity, explaining the seasonal differences of their communities.

The fish community was also shown to be highly dependent on salinity, temperature, dissolved oxygen, pH, depth and transparency which define their habitat and explains their distribution and migration patterns during their life cycles (Aguirre-León *et al*., this issue). Three fish communities were defined mainly based on the life cycle and tolerance to salinity of the species: Marine euryhaline (ME, 27 species), Marine stenohaline species (MES, 3 species) and estuarine-lagoon permanent inhabitant (E, 1 species.)

4. Contamination and potential sanitary problems

As highlighted in different studies presented in this monograph, the Sontecomapan Lagoon is faced to several ecological and sanitary threats linked to the contamination of water and/or sediment in relation with human activity.

Chemical contamination

Metal concentrations (Cr, Ni, Zn, Cu, Co, Cd and Pb) in sediments, submerged vegetation and bivalve molluscs were monitored during an annual cycle in 1991-1992 by González-Fierro and Ponce-Vélez (this issue), in the frame of a general evaluation of the contamination levels of the coastal systems of the Gulf of Mexico (González-Fierro, 1995). At that time, the Sontecomapan Lagoon was chosen as a reference (pristine) site due to low human activity around the site. Nevertheless, high levels of total and bioavailable concentration of metals in surface sediments were recorded at sites of large river discharges. In particular, Cr, Ni and Zn concentrations were particularly high and Ni presented a real ecological risk for benthic species. High concentrations of Cu and Zn were also found in oysters (*Crassostrea rizophorae*) as well as of Ni and Cr in the seagrass (*Ruppia maritima*) and in mussels (*Brachidontes exustus*). Bioconcentration factors showed that the sediments are an important source for these contaminations. In such shallow and windy environment, the role of sediment resuspension linked to wind or to boat traffic must be of major importance in the dynamics of these contaminants and their bioavailability and transfer to the food-web. The role of planktonic organisms in transferring the contaminants from the water or sediment particles to the food web must be elucidated. These results highlight the necessity to maintain a monitoring of metals in the lagoon.

Faecal contamination

Water quality of the lagoon may be temporarily and/or spatially affected by faecal pollution as shown by Soto-Castor and Esquivel-Herrera (this issue). Indeed, faecal coliform densities surpass the permissible levels and the presence of the enteropathogenic genera *Salmonella* and *Shigella* poses a further alert.

This pollution is mainly due to the presence of human settlements and related activities as agriculture, cattle and poultry husbandry, but wildlife waterfowl and mammals also play a role. The time and space variation of abundance and distribution of faecal coliform and enteropathogenic bacteria are correlated to runoff from the surrounding human settlements. However they have no link with total heterotrophic bacteria which in turn are positively correlated to phytoplankton biomass (chlorophyll *a*). Spatially, the north and eastern zones differed from the south and western zones. Seasonally, total coliforms were the most abundant during the full rainy season in relation with maximal runoff influence over the lagoon. Faecal coliforms were prominent during the dry and north wind seasons particularly in the northern part in relation with the upstream sources at La Palma which also contributed to increase inorganic nitrogen and phosphorus (Esquivel & Soto, this volume). At the opposite total heterotrophic bacteria had their highest concentration in the early rainy season, benefiting from phytoplankton bloom and better oxygenation.

The presence of the enterobacteria genera *Enterobacter*, *Klebsiella*, *Escherichia*, *Serratia*, *Citrobacter*, and *Proteus* in all surveys is linked to their adaptability to survive in conditions that are not the optimal for their growth. They were particularly abundant near the Basura stream and settlement areas (Chancarral and El Real). *Escherichia* and *Shigella* were important in all the seasons whereas *Salmonella* was mainly detected in the rainy season.

In definitive the presence and punctual abundance of coliforms and pathogenic enterobacteriaceae linked to human-related activities may be really problematic particularly in the more stagnant water areas of the lagoon. However the flushing of the lagoon by tidal currents represents a defence against this problem, as underlined by Soto-Castor and Esquivel-Herrera (this issue) who note that total bacteria concentration is lower than would be expected for a tropical coastal lagoon in relation with this flushing.

Harmful algae

Another important issue concerning the water quality of the lagoon and sanitary aspects is the presence of potential harmful phytoplankton species. Figueroa-Torres *et al.* (this issue) identified 46 species that are potentially harmful or toxic among the 357 phytoplankton species they inventoried by combining historical data and current data (own survey in June 2015 and February 2016). These 46 potentially harmful species are mainly represented by Dinoflagellates (20 species) and Diatoms (19 species). Among them, 12 species are toxic and five may affect the humans: the diatoms *Pseudo-nitzschia pungens, Pseudo-nitzschia pungens var atlántica* and *Pseudo-nitzschia seriata,* which produce domoic acid that causes amnesic poisoning by consumption of contaminated shellfish, the dino flagellate *Dinophysis caudata* that causes diarrhea and the Cyanoprokariota *Lyngbya majuscula* that produces dermatitis and respiratory diseases. Five other species (the dinoflagellates *Gonyaulax spinifera*, *Phalacroma rotundatum*, *Prorocentrum gracile*, *Prorocentrum micans* and *Tripos furca*) may cause disease and mortality among fishes and invertebrates. Most of the toxic species are present in the dry season, so special attention should be paid to their presence at this season.

Despite toxic blooms were never detected, it appears necessary to implement a monitoring of toxic phytoplankton in the lagoon. In a context of increasing eutrophication of anthropic origin in the last years blooms of toxic phytoplankton may affect the economic production, human health and marine biota

5. Some perspectives

The implementation of an operational hydrodynamic circulation model will help to better describe the water circulation in relation to tidal and climatic forcings and to better understand the variations of environmental conditions and living communities of the lagoon. It will also allow anticipating exceptional climate episodes (flood, drought, crisis of Anoxia…) or pollution events (eg trade ship stranding at the vicinity of the lagoon) in order to limit their impacts on the ecosystem.

The conceptual model we present here (Figure 1) may serve as a basis for the implementation of a numerical ecosystem model that could be coupled to the hydrodynamic model in order to better understand the ecological functioning of the lagoon and to simulate its evolution in relation with different natural and/or anthropogenic forcings.

Further investigation based on field observation and experimentation is however necessary to develop and test these models of ecosystem.

They will also help, in complement to the data presented in this issue, to identify biological indicators and tools for ecological management and governance of fisheries through simulation of temporal evolution scenarios needed to anticipate problems and to propose concrete solutions. This issue also point out the necessity to follow or implement monitoring actions focusing sensible issues for the ecosystem (e.g. mangrove area, metal and fecal contamination of water and sediment, toxic phytoplankton, etc.).

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Glossary

Abiotic. Not related to a living organism.

Abundance. The total number of a taxon or taxa individuals in an area, volume, population or community.

Acanthocephala. A small phylum of parasitic invertebrates that comprises the thorny-headed worms.

Algal bloom. A massive increase in the number of phytoplankton cells, which can be either beneficial or harmful.

Amphibian. A cold-blooded vertebrate animal of a class that comprises frogs, toads, newts, salamanders, and caecilians. They are distinguished for having an aquatic gill-breathing larval stage followed (typically) by a terrestrial lung-breathing adult stage.

Analysis of variance. A procedure based on the decomposition of the total variance of an experiment's offset into its partial variances, for example those due to experimental treatments and those due to error; variance ratios are computed as the quotient of the variance due to each treatment times the variance due to error, and the test's significance by contrasting this result against F distribution.

Anoxia. A complete lack of oxygen.

Benthic-organisms. Species living on the sea bottom, either attached to a substrate, free moving over it, or embedded or burrowing in it.

Bioavailability. The existence of a chemical element in a state that allows its assimilation by living organisms, so that its positive or negative effects can be exerted.

Bioavailable fraction. Refers to the metal fractions that organisms can absorb in the form that they are present in a given ecosystem.

Biodiversity hotspot. A threatened region with large number of species of at least 1500.

Bioindicators. Attributes of biological systems that are used to discern the properties of an environment. Species or species associations were firstly used as indicators but later other organization levels such as biological communities were employed, which is particularly useful for contamination studies. Indicator species are those living organisms or their remains that allow discerning the occurrence of a phenomenon or condition either at the present or in the past which is related to environmental conditions.

Biomass. The sum of all living organisms in a given area or at a given trophic level, usually expressed in terms of living or dry mass.

Biotic. Refers to the living constituents of an environment.

Biotoxines. Poisonous chemical compounds produced by living organisms.

Bird. Homeoterm egg-laying vertebrate animal distinguished by the possession of feathers, forelimbs modified into wings, a beak, and typically by being able to fly. Formerly it was grouped into class Aves, but currently into Theropoda, and considered as the only extant dinosaurs.

Bottom-up control. The regulation system of trophic levels in which the abundance of individuals at the higher levels is determined by the factors at the lower levels.

Bromeliad. A plant from tropical and subtropical America, typically having short stems with rosettes of stiff, spiny leaves. Some kinds are epiphytic and many are cultivated as pot plants.

Canonical Correspondence Analysis (CCA). A multivariate method to elucidate the relationships between biological assemblages of species and their environment. The method is designed to extract synthetic environmental gradients from ecological data-sets. Gradients are the basis for succinctly describing and visualizing the differential habitat preferences (niches) of taxa through an ordination diagram.

Carbonates. Chemical compounds characterized by the $(CO_3)^{-2}$ complex. The main mineral forms are calcite and dolomite, which are the main constituents of sedimentary or metamorphic rocks.

Cladoceran. A minute branchiopod crustacean of the Cladocera order, which includes the water fleas.

Classification. In taxonomy, the arrangement of organisms into a nomenclatural hierarchy, according to their degree of evolutionary relatedness.

Coastal lagoon. A coastal water body, found on all continents, usually with its main axis oriented parallel to the coastline, and separated from the ocean by a barrier, connected to the sea by one or more inlets which remain open at least intermittently, so that a mixture of seawater and sea biota with coastal or freshwater occurs. They are typically shallow.

Coliforms. Gram-negative, non-sporulated bacilli able to ferment lactose with gas production, considered as indicators of fecal pollution.
Total. Ferment lactose with gas production at 37°C.
Fecal. Fermenting lactose with gas production when incubated at a temperature of 44.5°C.

Colony Forming Units (CFU). The number of bacterial cells present that can grow in a culture medium under the incubation conditions employed. Each colony on a Petri dish in principle is formed from a single growing bacterial cell; thereby counting the colonies allows estimating the number of CFUs in a sample.

Copepod. A small or microscopic aquatic crustacean of the large class Copepoda.

Community. An integrated group of species inhabiting a given area; the organisms within a community influence one another's distribution, abundance, and evolution.

Crustacean. An arthropod of the mainly aquatic subphylum Crustacea, which comprises macroscopic forms such as crabs, lobsters, shrimp, krill or barnacles, but also microscopic forms such as copepods and water fleas.

Data matrix. A table where each row represents a variable or system descriptor and each column a state (a point in time or space) of the system.

Decapod. A crustacean of the order Decapoda, such as a shrimp, crab, or lobster.

Dendrogram. The graphical representation of a cluster analysis where hierarchical branching is observed. The usual convention is that two objects or entities are most similar when the link among them occurs nearer to the plot's base.

Denitrification. Is a microbe-facilitated process of nitrate reduction performed by a large group of heterotrophic facultative anaerobic bacteria, that may ultimately produce molecular nitrogen (N_2) through a series of intermediate gaseous nitrogen oxide products. This respiratory process reduces oxidized forms of nitrogen in response to the oxidation of an electron donor such as organic matter.

Detritus. Organic matter lost from any trophic level by causes different from predation or grazing, in diverse stages of decay, along with associated biota, and usually fine-grained inorganic components such as silt or clay.

Diatoms. Division Bacillariophyta Hustedt. Unicellular microalgae, including both solitaire and colonial forms. Their cells are enclosed within a silica-impregnated cell wall forming two valves and several cingular or connective bands, known as frustula.

Dinoflagellates. Division Pyrrophyta. Mainly planktonic microorganisms moving through water by means of two unequal undilopodia and presenting orange-red pigments. Several among them present photosynthetizing chromatophores in the protoplast and cellulose in their cell wall, and are thus similar to plants. There are also heterotrophic forms that resemble animals because of their locomotive abilities. Most of them present a special type of nucleus with fibrillar chromosomes that remain loosely condensed and visible along all the mitotic cycle.

Direct count. It is an estimation of bacterial numbers in a sample based on cell counts from slides stained with the fluorochrome DAPI and observed under an epifluorescence microscope.

Distribution. This term refers to the patterns governing the spatial position of the individuals from a population across an area or territory.

Diversity. It relates to the absolute number of species in an assemblage, community or sample (species richness), but is also dependent on the way that organism numbers are distributed along the different species (eveness).

Dominance. The extent to which a given species predominates in a community because of its size, abundance or coverage, and affects the fitness of associated species.

Eigenanalysis. A linear algebra technique for the extraction of latent roots and vectors (eigenvalues and eigenvectors) from a square matrix; these describe matrix properties in a decreasing order as related to system variance. It is the basis for multivariate ordination methods such as principal component analysis, factor analysis and canonical correspondence analysis, among others.

Enteropathogenic. Microorganisms with the ability to produce gastrointestinal disease through growth-related causes and their intestinal cells invasion or destruction.

Enterotoxic. Microorganisms that produce toxins that can harm the gastrointestinal tract or induce systemic effects.

Epiphyte. A plant that grows on another plant, especially one that is not parasitic, such as the numerous ferns, bromeliads, air plants, and orchids growing on tree trunks in tropical rainforests.

Euclidean distance. A dissimilarity measure (distance) between two elements from a data matrix; it is the extension of Pythagoras' Theorem to a n–dimensional space.

Eutrophication. The increase of biological production of an aquatic system that originates from a higher availability of nutrient salts. Even though this might appear to be a beneficial process, generally it relates with the proliferation of few primary producer species implying a decrease in species diversity. Other associated effects include oxygen depletion during the night, liberation of toxins by proliferating species, an increase in the sedimentation rate and the acquisition of unpleasant flavors or odors by water; it is thus considered as an undesirable condition.

Fern. A flowerless tracheophyte which has feathery or leafy fronds and reproduces through spores released from the undersides of the fronds.

Fish. Finned poikilotherm vertebrate animal of the classes Agnatha, Chondrichthyes or Osteichthyes, provided with gills during its whole life story, and living wholly in water.

FPAS (Folin Phenol Active Substances). Compounds that react to Folin phenol, mainly tanins and lignins.

Function. Role of organisms within ecosystems.

Gram stain. It is based on the ability to allow the passage of Gram stain (crystal violet) into the bacterial cell. When it does, the cell is stained violet (Gram-positive); if not, the cell is stained red, the color of safranin, the contrast stain (Gram-negative). This character depends on the structure of the bacterial cell wall.

Gross primary productivity. Total carbon fixation rate without subtracting respiration.

Habitat. The locality, site and particular type of local environment occupied by an organism.

Holoplankton. Organisms which remain as plankton throughout their entire life cycle, e.g. algae.

Insect. A small arthropod animal of class Insecta, that has six legs as an adult, and generally one or two pairs of wings.

Interstitial water. The water held in pore spaces in rock, soil or sediment.

Introduced species. Species that are not native to an area but brought in through human activities.

Jaccard index. Also known as the Jaccard similarity coefficient, is a statistic used for comparing similarity and diversity of sample sets. The Jaccard coefficient measures similarity between finite sample sets.

Land use. Also known as land-cover change is a general term for the human modification of Earth's terrestrial surface.

Leaf extract. Substances diluted in water obtained from decomposition of mangrove leaves.

Leaf litter. Leaves settling on the lagoon's bottom from mangrove, as a result of their natural falling.

Legume. A leguminous plant (member of the pea family), especially those grown as a crop.

Liana. An either herbaceous or woody climbing plant that hangs from trees, but is permanently rooted to soil, especially abundant and diverse in tropical rainforests, but which are also found in colder and drier environments.

Macroalgae. Commonly termed "seaweeds", are multicellular, macroscopic benthic algae belonging to green (phylum Chlorophyta), red (phylum Rhodophyta), and brown (phylum Phaeophyta) taxonomic groups. They are found in estuaries throughout the world, attached to substrates or drifting in the water column.

Mangroves. Dicotyledonous woody trees and shrubs that grow above mean sea level to form intertidal forests along subtropical and tropical coasts (Alongi, 2016).

Mangrove forest. Coastal wetland vegetation dominated by tree species tolerant to some degree of salinity.

Marine pollution. The human-related introduction of matter or energy, both directly or indirectly, into the environment that generate noxious effects implying a hazard for living beings or human health, handicapping productive marine activities.

Marsupial. A mammal of Infraclass Marsupialia whose members are born incompletely developed and are typically carried and suckled in a pouch at the mother's belly. Marsupials are found chiefly in Australia, New Guinea and America.

Metalloid. Elements that have characteristics between metals and non-metals.

Metals. Elements able to conduct heat and electricity, which have a characteristically shiny appearance. In their native state they are solid, except for mercury (Hg) that regularly is liquid at room temperature.

Misidentification. Incorrect assignment of a specimen to a taxon.

Mollusk. An invertebrate of phylum Mollusca, which includes snails, slugs, mussels, and octopuses. They have a soft unsegmented body and live in aquatic or damp habitats; most kinds have an external calcareous shell.

Most Probable Number (MPN). Estimated number of active cells in a sample or culture, based on the probability of certain combination of positive and negative results (for example bacterial growth) in different sample dilutions. This is obtained from the Poisson statistical distribution, and generally is shown in MPN tables for different combinations of positive results at various dilutions.

Mutagenic. A chemical or physical agent that changes the genetic material of an organism, usually DNA, thus increasing the frequency of mutations above the natural background level.

Natural services. Tangible or intangible benefits that ecosystems provide to humans.

NDVI. Normalized Difference Vegetation Index. The most common method to measure and map the density of green vegetation over the Earth using satellite remote sensors.

Nematode. A worm of the large phylum Nematoda, such as a roundworm or threadworm.

Nitric oxide. A free Nitrogen-Oxygen chemical radical produced through the bacterial denitrification process.

Nitrogen fixation. Is a process by which atmospheric nitrogen (N_2) is converted into ammonia (NH_3) in the presence of nitrogenase. The fixation process frees nitrogen atoms from their triply bonded diatomic form, $N\equiv N$, to be used in other ways.

Nitrogenases. The enzymes responsible for the nitrogen (N_2) reduction to ammonia (NH_3). Nitrogenases are the only family of enzymes known to catalyze this reaction, which is a key step in the process of nitrogen fixation. They are only found naturally in certain microorganisms such as the symbiotic *Rhizobium* and *Frankia*, or the free-living *Azospirillum* and *Azotobacter*.

Nitrous oxide. Nitrous oxide (N_2O) accounts for about 5.9% of all U.S. greenhouse gas emissions from human activities. Nitrous oxide is naturally present in the atmosphere as part of the Earth's nitrogen cycle, and has a variety of natural sources. However, human activities such as agriculture, fossil fuel combustion, wastewater management, and industrial processes are increasing the amount of N_2O in the atmosphere. Nitrous oxide molecules stay in the atmosphere for an average of 114 years before being removed to a sink or destroyed through chemical reactions. The impact of 1 pound of N_2O on warming the atmosphere is almost 300 times that of 1 pound of carbon dioxide.

Nonparametric methods for the species richness estimation. Methods and estimators concerned with the problem of predicting the number of new species that will be discovered in an additional survey given that an initial survey has already been conducted. These are termed nonparametric because they are not based on the parameter of a species abundance model.

Nortes. Climatic season characterized by strong winds coming from the North to the Gulf of Mexico, commonly occurring from October to March.

Odonate. A predatory insect of the order Odonata; a dragonfly or damselfly.

Orchid. A plant with complex flowers that are often showy or bizarrely shaped, having a large specialized lip (labellum) and frequently a spur. Orchids occur worldwide, especially as epiphytes in tropical forests, and are valuable hothouse plants.

Ordering. Deploying (to plot) a cloud of data points in a two or three-dimensional coordinate system, to make visible the relationships between data points in a multidimensional space. It is also the order of a points set with respect to one or more axes.

Organic matter. In soils or sediments, it is the product of the chemical decomposition of organisms and classified in humic (generally refractory) and not humic (generally labile) compounds.

Orthorectification. Process of removing the effects of image perspective (tilt) and relief (terrain) effects for the purpose of creating a planimetrically correct image. The resultant orthorectified image has a constant scale and so the features are represented in their ‘true’ positions. This allows for the accurate direct measurement of distances, angles, and areas (i.e. mensuration).

Otter. A semiaquatic fish-eating mammal of the weasel family, with an elongated body, dense fur, and webbed feet.

Pentastomid. Any of numerous parasitic wormlike animals of the crustacean class Pentastomida (often regarded as a distinct phylum or subphylum), which infest the lungs and nasal passages of vertebrates.

Phylogeny. The evolutionary development of an organism or group of organisms in time, as distinguished from ontogeny, which is the development of an individual from fertilization of the egg until adulthood.

Phytoplankton. A community of microscopic organisms of relatively simple shapes, mostly living by photosynthesizing and suspended in the water column.

Platyhelminthes. A phylum of invertebrates that comprises flatworms.

Predation. The killing and eating of other animals.

Primary production. Mainly based on the biochemical process of photosynthesis, in which solar energy is biochemically fixed. Primary producers in the aquatic environment are macrophytes, phytoplankton and phytobenthos.

Primary productivity. It is the rate at which energy is stored in the organic matter of autotrophs through photosynthesis.

Principal component analysis. It is based on the rigid rotation of coordinate axes (without deforming) so that the variability and structure of the data point cloud can be represented as a bi- or tridimensional plot.

Productivity. The rate at which something is produced. It is also the quotient of the organic matter produced at a trophic level times the biomass from which it was produced.

Red tide. A natural phenomenon caused by the numerical increase of one or several microalgae in water, which because of its concentration gives a characteristic coloration to the water, however some may be colorless.

Remote sensing. Is a way of collecting and analyzing data to get information about an object without the instrument being in direct contact with the object.

Reptile. Poikiloterm vertebrate animal of a class that includes snakes, lizards, crocodiles, turtles, and tortoises. They are distinguished by having a dry scaly skin, for being devoid of external gills during their whole life history, and typically laying soft-shelled eggs on land.

Rhizocephalan. A parasitic crustacean of the order Rhizocephala.

Rotifer. A microscopic multicellular aquatic animal of the phylum Rotifera.

Rotifera. A phylum of microscopic suspensivorous metazoans (multicellular organisms). About 2030 species are known and are classified into three main groups: Seisonida of only 3 known marine species, Monogononta with 1570 species and Bdelloidea with 461. They are organisms of a size between 50 to 2.000 µm. They are composed approximately by 1000 cells and filter small food particles from the water column by a ciliated crown located in the front portion of the body. The crown also can be used for locomotion, however some species are sessile.

Seagrass. A grass-like monocotyledonous plant that lives in or close to the sea.

Secchi disk. A metal or plastic disc 0.3 m in diameter, white or with white and black alternating quadrants, used to estimate the depth of effective penetration of sunlight into the water, based on the point where the observer loses sight of it.

Sediment. Particulate matter that has been transported by wind, water or ice and subsequently deposited, or that has been precipitated and settled from water; sedimentation.

Soluble reactive phosphorus. Filterable portion of inorganic phosphorus stable in solution and that is easily assimilated by the primary producers.

Species accumulation curves. A ‘species accumulation curve’, or ‘collector’s curve’, is a plot of the cumulative number of species discovered S(n), within a defined area, as a function of some measure n of the effort expended to find them.

Species richness. Species richness refers to the number of species in a given area; the more species, the greater the species richness.

Spectral Signature. Is the wavelength composition of radiation reflected as a function of wavelength-specific absorption and reflection by some surface.

Synonymy. According to the 'International Code of Zoological Nomenclature', a chronological list of all scientific names that have been applied, correctly or incorrectly, to a given taxon.

Systematics. The study of the historical evolutionary and genetic relationship between organisms and their phenotypic similarities and differences.

Tanaidacea. A small order of malacostracan crustaceans (division Peracarida) often included in the Isopoda and intermediate in character between that order and the Cumacea.

Taxa. Plural of taxon.

Taxon. A grouping in a scientific classification of organisms, e.g. a family, genus or species.

Top-down control. The regulation system of trophic levels by which the abundance of herbivores in the lower trophic levels is determined by factors from the higher levels.

Trematod. A parasitic flatworm of the class Trematoda.

Trophic cascade. The effect of size changes of a given population on lower non-consecutive components of the food web.

Trophic level. A level in a food web.

Treatments. Any factor for which its effects are to be measured in an experimental design.

UPGMA. Abbreviation of Unweighted Pair-Group Method with Arithmetic mean. It is a simple agglomerative (bottom-up) hierarchical clustering method.

Valid name. According to the 'International Code of Zoological Nomenclature', the correct scientific name for a taxon.

Viable count. The estimation of bacterial numbers in a sample by counting the Colony Forming Units (CFU) on an agar plate inoculated with a known dilution of the sample.

Zoogeographical province. A bio geographical unit that is distinguished by a fauna which is more or less distinctive at the species, genus, family, or order level. Areas with at least 10% endemic species are considered provinces.

Zooplankton. Microscopic animals that live and drift in water.

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Tropical coastal lagoon environments provide a number of ecosystem services, but are threatened by the pressure imposed by human activities and climatic change; these systems are particularly vulnerable because of a high demographic growth. Therefore, the understanding of their ecological behavior and the characterization of lagoon health indicators have attained importance. Under this perspective Mexican (UAM-X) and French (UMRs MIO and MARBEC) researchers have collaborated from 2011 to 2014 as part of one action of the international exchange program ECOS/ANUIES, and chose the Sontecomapan lagoon (at the Mexican state of Veracruz) as a case study.

This book provides information of the ecological behavior, water quality indicators, and details of microorganisms and plankton, which due to their short life cycles and their high reactivity to environmental conditions are good

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