Hurricane Mitch: Integrative Management and Rehabilitation of Mangrove Resources to Develop Sustainable Shrimp Mariculture in the Gulf of Fonseca, Honduras

By Victor H. Rivera-Monroy, Robert R. Twilley, Edward Castañeda

In collaboration with Ariel R. Alcantara-Eguren, David Aragones, Delia Martinez, Diego Valderrama

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Integrative Management and Rehabilitation of Mangrove Resources to Develop Sustainable Shrimp Mariculture in the Gulf of Fonseca, Honduras

By

Victor H. Rivera-Monroy[1], Robert R. Twilley[1] , Edward Castañeda[1]

in collaboration with

Ariel R. Alcantara-Eguren[2], David Aragones[2],
Delia Martinez[3], Diego Valderrama[4]

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[1] Department of Biology, Center for Ecology and Environmental Technology, University of Louisiana at Lafayette, Lafayette, Louisiana, USA.

[2] Universidad IberoAmericana, Golfo-Centro, Puebla, México

[3] Laboratorio de Calidad de Agua, ANDAH, La Lujosa, Choluteca, Honduras.

[4] Aquaculture/Fisheries Center, University of Arkansas at Pine Bluff, USA

One of the major difficulties in developing coastal management plans in tropical regions of the world has been the conflict between developing and expanding shrimp mariculture and the conservation and management of mangrove resources. The conflict arises due to the use of mangrove forest areas to establish shrimp farms, a practice that has resulted in significant losses of mangrove forests in different parts of the world during the last 25 years. Due to an increasing awareness of the potential environmental impacts that shrimp mariculture practices have on the environment, the shrimp industry

has began to develop best management practices in cooperation with government agencies, financial institutions, and NGOs. One of these practices include the nonconstruction of shrimp farms in mangrove dominated coastal zones (Stanley, 2000; Boyd and others, in press). The analysis of production trends of shrimp ponds constructed in mangrove areas has shown that these areas are the least desirable to establish shrimp operations due to high concentrations of total sulfur and organic matter that negatively affect shrimp growth and increase maintenance costs. Although this practice will certainly contribute to the conservation of mangrove forest, it is still not clear what will be the impact of wastewater from shrimp mariculture on ambient water quality of adjacent estuarine and coastal waters. The purpose of this study was to evaluate how the rehabilitation of coastal resources damaged by Hurricane Mitch could be integrated into ecological processes of mangrove ecosystems with management practices of shrimp pond mariculture in the Gulf of Fonseca. We first estimated the total area of mangrove resources for the entire Gulf of Fonseca, including coastal areas in Honduras, El Salvador, and Nicaragua, using remote sensing techniques. Mangrove forest and shrimp pond aerial extension and spatial distribution were determined in Honduras, particularly in the southern region of the Gulf of Fonseca, where most of the shrimp industry is located. We also estimated temporal changes in mangrove and shrimp pond cover and analyzed long term data of water quality variables available for the region to understand current levels of fertility in coastal waters. Similarly, we determine what variables control mangrove spatial distribution and structure through field studies during the dry and rainy season (2000-01). Finally, we present estimates of the treatment capacity of mangrove forest to “assimilate” excess inorganic and organic forms of nitrogen and phosphorous and propose a mangrove:pond area ratios to remove nutrient excess in shrimp pond effluents. This strategy is presented as part of a wide range of best management practices available to reduce the risk of a potential nutrient enrichment of the Gulf of Fonseca by shrimp farm effluents.

Total mangrove area estimated (two classes) for the Gulf of Fonseca is 47,757 ha (= 3 m= 42,444; > 3m = 5,313 ha) in 1999. This number includes coastal areas of El Salvador and Nicaragua. Mangrove total surface estimated for the Gulf of Fonseca in Honduras was 42,215 ha, 37,788 ha, and 35,375 ha for 1985, 1992, and 1999, respectively. Thus, there was a reduction of 6,840 ha in a 14-year period. In contrast, total mangrove surface in Punta Guatales (Granjas Marinas San Bernardo, the largest shrimp farm in Honduras) increased from 3,673 (1954) to 4,034 (2000) ha representing a net gain of 361 ha. Approximately 77% of the shrimp pond area in the Gulf of Fonseca in 1999 was located in Honduras in comparison to El Salvador (1.%) and Nicaragua (22%).

The conspicuous structural patterns of mangrove forest in the southern Gulf of Fonseca are strongly related to the frequency of tidal flushing and salinity gradients, which determine the zonation and species composition within the study area. Salinity stress has been alleviated by hydrological modifications as a result of shrimp mariculture development in the Pedregal and San Bernardo estuaries. Areas close to channels discharging effluents support high growth rates of propagules and seedlings allowing the establishment of mangrove species that are less tolerant to high salinities (i.e., *Laguncularia racemosa*).

The soluble reactive phosphorous (SRP) values observed in the Pedregal and San Bernardo estuaries are among the highest reported for coastal ecosystems. Total suspended sediments were consistently higher in the estuarine waters that in pond effluents, particularly during the dry season. Potential treatment capacity values to treat nitrogen and phosphorous by mangrove forests were high and positive. The magnitude of the treatment capacity of dissolved inorganic nitrogen (DIN) was high (2.08 kg ha^{-1} d^{-1}). It is apparent that most of the DIN entering into the mangrove forest could be processed through denitrification alone. Treatment capacity for total nitrogen was also high (1.49 kg ha^{-1} d^{-1}), although this result has limited interpretation due to the processes involved in the assimilation of particulate and organic N forms by mangrove trees. Potential treatment capacity of SRP (0.66 kg ha^{-1} d^{-1}) and TP (0.61 kg ha^{-1} d^{-1}) is high. SRP and TP loading rates are equivalent to daily requirements for plant uptake and much lower that for accumulation in soils. Thus, based on these rates, it is apparent that phosphorous (P) in pond effluents could be readily uptake by mangrove forest. An increase of P concentrations could promote tree growth particularly in areas where scrub mangroves are dominant in the Gulf of Fonseca. Further work is needed to evaluate how the interaction between P concentrations and salinity in mangrove soils regulate the distribution and growth rates of mangrove forest in the Gulf of Fonseca. Based on treatment capacities for N and P, we estimated ratios of mangrove wetland to pond less than one. Due to the potential improvement of water quality of shrimp pond effluents as result of the use of mangrove forest in the region, it is strongly recommended that this treatment be considered as a best management practice in the Gulf of Fonseca. Although shrimp farm operations are strictly a profit oriented industry, it is possible to develop research programs that could allow the participation of different institutions to financially support research addressing specific problems in mangrove ecology, particularly questions related to mangrove ecophysiology and nutrient cycling. Although these research areas are critical to develop mangrove conservation plans in the Gulf of Fonseca region, they are the less understood in mangrove ecology. There are indications that the shrimp industry is interested in the development of active mangrove conservation programs. And this interest can be translated into economic incentives within the industry to promote a rational use of mangrove forest and develop regional plans toward the sustainability of the shrimp industry not only in Honduras but also in tropical regions.

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[1] Department of Biology, Center for Ecology and Environmental Technology,
University of Louisiana at Lafayette, Lafayette, Louisiana, USA.

[2] Universidad IberoAmericana, Golfo-Centro, Puebla, México

[3] Laboratorio de Calidad de Agua, ANDAH, La Lujosa, Choluteca, Honduras.

[4] Aquaculture/Fisheries Center, University of Arkansas at Pine Bluff, USA

Background

One of the major difficulties in developing coastal management plans in tropical regions of the world has been the conflict between developing and expanding shrimp mariculture and the conservation and management of mangrove resources. The conflict arises due to the use of mangrove forest areas to establish shrimp farms, a practice that has resulted in significant losses of mangrove forests in different parts of the world during the last 25 years (Boyd and Clay, 1998; Twilley and others, 1998b). In addition to the loss of mangrove areas, it is also argued that ecological services are lost since mangrove wetlands are considered a sink of inorganic nutrients and sediments and exporters of organic matter (Rivera-Monroy and others, 1995; Childers and others, 1999). These ecological functions are considered critical since they are related to maintaining both water quality in estuarine waters and the productivity of economically important fisheries (Twilley, 1997).

Due to an increasing awareness of the potential environmental impacts that shrimp mariculture practices have on the environment, the shrimp industry has began to develop best management practices in cooperation with government agencies, financial institutions, and NGOs. One of these practices include the nonconstruction of shrimp farms in mangrove dominated coastal zones (Stanley, 2000; Boyd and others, in press). The analysis of production trends of shrimp ponds constructed in mangrove areas has shown that these areas are the least desirable to establish shrimp operations due to high concentrations of total sulfur and organic matter that negatively affect shrimp growth and increase maintenance costs (Boyd, 1991; Boyd, in press). Although this practice will certainly contribute to the conservation of mangrove forest, it is still not clear what will be the impact of wastewater from shrimp mariculture on ambient water quality of adjacent estuarine and coastal waters. Water exchange is critical, since shrimp pond management operations require high volume of brackish water to assure optimal conditions (e.g., high phytoplankton productivity; oxygen levels > 3 g L^{-1}) for shrimp growth, particularly in semi-intensive shrimp pond operations. This type of operation requires water for pond filling, routine exchange (up to 15% of pond volume), and replacement for evaporation and percolation (Hopkins and others, 1993; Teichert-Coddington and others, 2000; Boyd and others, in press). Recent studies in Honduras has showed that routine water exchange may not be necessary (Boyd and others, in press), yet current surveys indicate that this management practice is still widely applied throughout the world (Boyd, 2001).

The main problem to evaluate the impact of shrimp pond effluents on the water quality of receiving coastal waters is its nonpoint source nature. Pond effluents come from an extensive number of farms over large areas within different schedules and levels of production intensity (Stanley, 2000). Actual loading rates of nutrients from shrimp farms to adjacent coastal waters are not well established due to the lack of regular sampling of organic and inorganic nutrients (mainly nitrogen and phosphorous) in pond intake and effluent water. Apparently this lack of data is strongly related to the absence of water quality standards and/or lack of enforcement to regulate effluent loads. Recent studies in Honduras show that effluent loadings are in the range of 35 kg of nitrogen (N) and 12 Kg of phosphorous (P) for 1000 kg of live shrimp produced (Boyd and Teicher-Coddington, 1995). Estimates from shrimp farms in Colombia indicated a loading rate range of 34-74 kg ha^{-1} yr^{-1} for inorganic N (Rivera-Monroy and others, 1999). In Asia, low density of shrimp operations produce loadings of 455 and 328 kg of N and P, respectively, per 1 ton of shrimp produced (Dierberg and Kiattisimkul, 1996). The overall impact of these organic and inorganic loadings on the productivity of the receiving waters is not clear, mainly due to the variety and complexity of the interaction of physical and biogeochemical factors regulating the uptake and recycling of nutrients in the coastal zone. The potential effects of nutrient loadings into estuarine waters are several, among the most important are: a) eutrophication of adjacent estuaries, b) increased sedimentation due to organic matter, and c) reduced dissolved oxygen in receiving waters (Phillips and others, 1993; Hopkins and Sandifer, 1996; Stanley, 2000). In part, the degree to which these effects can become dominant depend on the water circulation, residence time, and geomorphology of the estuary where the shrimp farms are established.

Another critical factor in controlling the potential negative effect of pond effluents is the distribution and density of mangrove forests that surround tidal creeks and estuaries in tropical regions. Water exchange through tidal inundation between mangroves and estuarine waters is a processes that regulates water quality and affect the export of organic material from the mangrove forest to the coastal zone (Rivera-Monroy and others, 1995; Dittmar and Lara, 2001). Nutrient fluxes measured at the boundary between mangrove and estuarine waters indicate that mangrove forests import inorganic nitrogen and sediment and export dissolved organic matter and particulate nitrogen. This exchange of nitrogen suggests that mangrove forests are net sinks of inorganic nitrogen (Corredor and Morel, 1994; Rivera-Monroy and others, 1995) through the immobilization of N in the sediments (Rivera-Monroy and Twilley, 1996). It is this functional role that has stressed the importance of mangrove forest in the N cycling of tropical and subtropical regions and has prompted suggestions to use these forested wetlands as tertiary treatment of pond effluents. It has been proposed that this strategy could be considered a potential integrative management approach for the conservation of mangroves and the sustainability of the shrimp industry (Robertson and Phillips, 1995; Rivera-Monroy and others, 1999).

Although this approach to practice both conservation of mangrove forests and maintain a profitable shrimp industry is possible, there are not studies that directly evaluate its potential economical and ecological benefits. Among the major problems to implement this approach are the lack of data on the biogeochemistry of mangrove wetlands in different geomorphological settings, limited data sets to calculate effluent loading rates, and adequate aerial estimates (temporal and spatial) of mangrove forests around regions where shrimp farms are established. Given the long history of acute social and political conflicts between mangrove conservationists and shrimp producers, particularly regarding the expansion of the industry in coastal regions (Boyd and Clay, 1998), it is understandable that no efforts have been directed to implement management strategies that link both mangrove ecology and sustainable shrimp mariculture (Rivera-Monroy and others, 2001). We believe that developing management practices based on sound research programs could potentially solve current conflicts between both groups and advance the development of coastal management plans in tropical regions.

The Gulf of Fonseca, Honduras: A case study for potential development of technologies for mangrove conservation and management, and shrimp mariculture sustainability.

The shrimp industry in Honduras has experienced rapid growth during the last 10 years: from ~6,000 ha in 1989 to 14,000 ha in 1999 (Boyd and others, in press). Most of the farms are located in the eastern region of the Gulf of Fonseca, particularly around the region of Puntas Condega and Guatales. In contrast to other Latin American countries (e.g., Ecuador; Twilley and others, 1998a) most of the shrimp ponds have been established in salt flats significantly reducing the loss of mangroves in this region given the rapid growth of the industry (Vergne and others, 1993). In addition to establishing farms in higher elevations, there are several studies evaluating the impact of shrimp pond effluents on the environmental quality of the adjacent estuaries (Ward, 1999; Green and Tookwinas, 2001). Also studies have assessed the overall shrimp production in relation to not only pond environmental conditions but also to climatic variables (Teichert-Coddington and others, 1994). This type of study is not common in Latin America and emphasizes the increasing understanding of the shrimp industry of the importance of evaluating its sustainability under a wider ecological perspective (Olsen, 1995; Twilley and others, 1998a; Paez-Osuna and others, 1999).

The effect of Hurricane Mitch in Honduras also stressed the potential impacts of this type of climatic event at levels that are beyond the usual temporal and spatial scales generally considered in shrimp pond management. Although Hurricane Mitch did not hit the Gulf of Fonseca directly, its indirect effects are reflected in major diversions of freshwater (e.g., Rio Choluteca, Rio Negro) that will have a long-term effect on the hydrology of estuaries (e.g., San Bernardo) where shrimp farms are located. It is not clear if undisturbed mangrove forests contributed to reduce the impact of the storm in the Gulf of Fonseca; results from other regions show that the wetland vegetation is a major factor in controlloing erosion problems and storm surges along coastal areas (Cahoon and Lynch,1997; Day and others, 1997; Twilley, 1998). Damage to the shrimp industry infrastructure in the Gulf of Fonseca was severe but the recovery was relatively rapid (Boydand others, in press; Valderrama and Engle, in review) given the intensity of Hurricane Mitch and the tremendous damage to private property and loss of life throughout the country. More recently, Hurricane Michelle (November 2001) caused high precipitation in Honduras, ending a long dry season that destroyed large commercial crops and brought famine to poor rural areas. This contrasting effect of both hurricanes on coastal areas of Honduras shows that coastal management plans need to consider large-scale ecological and climatic events for developing sustainable rational use of coastal resources.

The purpose of this study was to evaluate how the rehabilitation of coastal resources damaged by Hurricane Mitch could be integrated into ecological processes of mangrove ecosystems with management practices of shrimp pond mariculture in the Gulf of Fonseca. We first estimated the total area of mangrove resources for the entire Gulf of Fonseca, including coastal areas in Honduras, El Salvador, and Nicaragua, using remote sensing techniques. Mangrove forest and shrimp pond aerial extension and spatial distribution were determined in Honduras, particularly in the southern region of the Gulf of Fonseca, where most of the shrimp industry is located. We also estimated temporal changes in mangrove and shrimp pond cover and analyzed long term data of water quality variables available for the region to understand current levels of fertility in coastal waters. Similarly we determine what variables control mangrove spatial distribution and structure through field studies during the dry and rainy season (2000-01).

Finally, we present estimates of the treatment capacity of mangrove forest to “assimilate” excess inorganic and organic forms of nitrogen and phosphorous and propose a mangrove:pond area ratio to remove nutrient excess in shrimp pond effluents. This strategy is presented as part of a wide range of best management practices available to reduce the risk of a potential nutrient enrichment of the Gulf of Fonseca by shrimp farm effluents. Another aspect of this project was to establish goals of mangrove rehabilitation and creation by using ecological models that can project growth of mangroves under different site criteria and evaluate the susceptibility of coastal waters to eutrophication under different scenarios of mangrove rehabiltation and shrimp farming in the intertidal zone. Results for the modeling component will be presented in another document.

Study area

The Gulf of Fonseca is a shared ecosystem that encompasses the periphery of El Salvador, Nicaragua, and Honduras on the Pacific Coast of Central America (Benitez and others, 2000). The southern region of the Gulf of Fonseca, Honduras is located between 12°59' and 13°30' N latitude and 86°43' and 87°48 W' longitude (fig. 1) (Vergne and others, 1993).

The Gulf of Fonseca, a tectonically originated bay, is a flooded coastal indentation formed by land movements associated with faulting and volcanism (Pritchard, 1967; Ward and Montague, 1996). The geomorphological characteristics of the Gulf of Fonseca’s coastal plain can be classified as a drowned river valley type estuary, with extensive mud flats and deltaic-like shoal areas, especially its eastern arms (Pritchard, 1967). It has a free connection with the Pacific of some 30 km in width and 20 m average depth. The entire coastal area of the Gulf of Fonseca encompasses about 1,000 km^2 of estuaries (consisting of mangrove forests, creeks, and tidal flats), islands, and seasonal lagoons (Admiralty, 1951; Vergne and others, 1993).

The climate of this region is characterized by two distinct seasons in the year, the dry season and the rainy season. The winter dry season extends from November through April, during which the region becomes quite arid. The rainy season, typically extending from May through October, is in fact interrupted in July by a brief dry period, known as the *canícula* in Honduras (Vergne and others, 1993). Mean annual precipitation ranges from 500 mm in the northeast to more than 2400 mm in the southwest, whereas the average annual evaporation of about 2800 mm largely exceeds precipitation generating an annual water deficit in this region (Hargreaves, 1980). The highest temperatures are registered in April and the lowest usually in September. Mean annual air temperature is about 30°C (Vergne and others, 1993).

Major rivers influence the estuaries in the Gulf of Fonseca's coastal region. The Choluteca River drains into La Jagua and El Pedregal estuaries, while the San Bernardo estuary comprises the mouth of the Negro River. Tides are semidiurnal with a mean vertical tidal range of 2.3 m (Vergne and others, 1993).

Mangrove forests composed by *Rhizophora mangle* (L.), *Avicennia germinans* (L.) Stearn, *Avicennia bicolor* Standley, *Laguncularia racemosa* (Gaertn), and *Conocarpus erectus (L.)* surround the estuaries and embayments of the Gulf of Fonseca (Oyuela, 1994). The conspicuous geometric feature (dendritions) of mangrove forests, horn-shaped estuaries (San Bernardo and El Pedregal estuaries), and tidal flats are the most relevant physiographic characteristics that are driven by the hydrology in this coastal zone (Admiralty, 1951).

Remote sensing of mangrove forests and shrimp ponds in the Gulf of Fonseca: temporal analyses and spatial distribution (1956-2000)

The estimation of mangrove areas in the Gulf of Fonseca has been historically one of the main research activities to evaluate the degree and extension of human impacts in the region. Since the shrimp industry was established in the region in the 1980s, there has been an increasing interest in assessing the role of the industry in the reduction of the mangrove forest area. Although, in general, shrimp ponds in the Gulf of Fonseca have been constructed in salt flats, mangrove forests around the farms are cut to construct access roads and structures to supply estuarine water into the ponds (Vergne and others 1993).

Due to the environmental and climatic conditions of the Gulf of Fonseca, mangrove forests show different degrees of stress related to high salt content (>60 g/L) in soils. This stress is reflected on sharp differences in tree height and species composition along short distances (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales). Most studies evaluating mangrove surface in the Gulf of Fonseca have considered these forest structural differences to develop a variety of classification classes. More recently, Sanchez (1998) used two classes (“Red Mangrove” and “Other Species”) to estimate conversion rates of mangrove areas to shrimp ponds and areas for commercial salt production for the period 1989-1995. COHDEFOR (1987) used a wider classification using tree height, and in minor degree, species composition (“dwarf”, “stress”, and “mature”). In this study, we used two classes (>3 m and = 3m) to asses mangrove spatial extension and distribution for the years 1985-1999.

Methods

To estimate mangrove aerial cover in the Gulf of Fonseca (Honduras) we used three (1985, 1994, 1999) Landsat Thematic Mapper (TM) images provided by USGS (table 1). Previous to image analysis, a “target” area was selected to facilitate the identification of mangrove cover (fig. 2). TM images were analyzed using the software Idrisi32 (Release 2) (2001). Although images for 1993 and 2000 were available, they were not used in the analysis due to partial geographical cover of the target area and interference by clouds. In addition to estimating mangrove and shrimp pond areas along the coast of Honduras, we focused the analyses in the southern region of the Gulf of Fonseca where there is a high density of shrimp farms. In particular, we analyzed distribution of mangrove forest and shrimp ponds in Punta Guatales where the largest shrimp farm in Honduras is located (Granjas Marinas San Bernardo). Land use for 1954, 1985, 1993, 1994, 1997, 1999, and 2000 was estimated for this farm using satellite images (figs. 3-8) (table 1); land use for 1954 was determined using digitized vegetation maps provided by USGS (fig. 9). We used two categories (>3 m; =3 m) to describe spatial differences of mangrove forest in the Gulf of Fonseca. These categories were selected based on fieldwork performed in Punta Guatales where mangrove structural variables were measured (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales). Other studies have differentiated mangrove classes based on salinity tolerances. For example COHDEFOR (1987) classified mangroves as “dwarf,” “stress,” and “mature,” a classification system also used by Vergne and others (1993). Although our criteria does not allow a direct comparison of mangrove areas by classes estimated by those studies, there is a general agreement in the classification since the “stress” and “dwarf” mangroves include trees <3 m tall. Trees >3 m tall are considered “mature” (see page. 15, in Sanchez,1998). Another reason only two categories were used in this study was the logistic (and time) limitation to visit different locations to do "ground truthing" in the central and northeastern section of the Gulf of Fonseca.

Results and discussion

Total mangrove area estimated for the Gulf of Fonseca is 47,757 ha (= 3 m= 42,444; > 3m = 5,313 ha) in 1999; this number includes coastal areas of El Salvador and Nicaragua, (fig. 10). Mangrove total surface estimated for the Gulf of Fonseca in Honduras was 42,215 ha, 37,788 ha, and 35,375 ha for 1985, 1992, and 1999, respectively (tables 2 and 3). Thus, there was a reduction of 6,840 ha in a 14-year period. In contrast, total mangrove surface in Punta Guatales (Granjas Marinas San Bernardo) increased from 3,673 (1954) to 4,034 (2000) ha, representing a net gain of 361 ha (tables 4 and 5; figs. 3-9).

Mangrove surface estimated in this study contrast to values reported by other authors in different years (table 6). For example, we estimated 42, 215 ha in 1985 and 35,375 ha in 1999. In contrast, Sanchez (1998) calculated 46,890 ha in 1989 and 41,900 in 1995. Sanchez's values from 1995 also contrast with estimates by AFE-COHDEFOR for the same year, showing a difference of 5,300 ha. It is difficult to evaluate the source of variation since the original revised documents do not describe with detail the actual methods used to estimate the areas.

Total area of shrimp ponds estimated for Honduras in 1999 is 15,589 ha, an increase of 14,735 since 1985 (845 ha) (table 2). According to Vergne and others (1993), shrimp ponds increased from 1,064 to 11,515 ha in the ten-year period of 1982-1992. Sanchez (1998) also reported a significant increase of 9,774 ha in pond area from 1989 (2,620 ha) to 1995 (12,394 ha). Shrimp pond surface within coastal regions of El Salvador (fig. 11) and Nicaragua (fig. 12) in the Gulf of Fonseca were 229 ha and 4621 ha in 1999, respectively; although values for Nicaragua are slightly underestimated due to uncompleted cover of the Landsat image for that year (fig. 12). Thus, approximately 77% of the shrimp pond area in the Gulf of Fonseca was located in Honduras in 1999 (fig. 13).

Forest Structure and Soil Properties of mangrove forests in Punta Guatales

Methods

Field Sampling

Field Experimental Design

Four sites were located along the San Bernardo (S1 and S2) and El Pedregal (S3 and S4) estuaries. Four transects were established along the elevation gradient in order to evaluate forest structure and soil properties in all sites. Three zones (fringe, transition, and dwarf mangroves) were clearly distinguished in all sites and sampled between October 2000 and August 2001.

Forest Structure

The spatial distribution and species composition of mangrove forests was assessed once at the beginning of the study (October 2000) using the point-center quarter method (PCQM) (Cintron and Novelli, 1984). All trees ≥ 2.5 cm in diameter were tagged and registered in each site to determine species composition, basal area ($m^2\ ha^{-1}$), tree density (stems/ha) and height (m). Structural indices were calculated according to Cintron and Novelli (1984).

Soil chemistry

Measurements of porewater nutrients, porewater sulfide, porewater salinity (g/L) and temperature (ºC), redox potential (Eh), pH, and water were monitored seasonally between February and August 2001 in all sites. Soil Eh (0, 10, and 45 cm depth) was measured in situ using a multidepth platinum probe (Hargis and Twilley, 1994) and for soil pH using a Digi-Sense pH meter. Porewater samples were collected at 45 cm depth using a plastic siphon and syringe (McKee and others, 1988) and analyzed for temperature and salinity (YSI salinity/conductivity/temperature meter) and sulfide concentrations (Lazar Model IS-146 sulfide electrode). A second porewater sample was filtered using a GF/F filter and store frozen until assayed for inorganic nutrients.

Nutrient concentration in soil samples

Soil samples were collected along all transects at each zone (fringe, transition, and dwarf mangroves) once at the beginning of the study (October 2000). Two cores per zone were collected using a 5-cm diameter core. Soil samples were divided into 10 and 20 cm intervals and stored for further analyses of total carbon (C), nitrogen (N), and phosphorus (P) contents.

Topography

Soil elevation was determined with a CST/BERGER Automatic Level along 100 - 300 m transects in each study site. Surveys were performed during the dry season (February 2000).

Laboratory analyses

Ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), and phosphate (PO_4^{-3}) concentrations of porewater samples were determined by colorimetric methods (Parson and others, 1984; Strickland and Parsons, 1972). Subsamples of soil cores were oven-dried at 60 ºC to a constant weight and ground with a Wiley Mill to pass through a 250 µm mesh. Carbon (C) and nitrogen (N) contents were determined on two replicas of each sample depth with an Elemental Analyzer NA 2500 using standard protocols. Total P (P) of soil samples were extracted with 1 N HCL after combustion in a furnace for 3 hr at 550 ºC (Aspila and others, 1976).

Statistical analyses

To assess the patterns of mangrove forest structure and soil properties, fixed factor models were used. The data were analyzed using a multi-factor ANOVA approach (Proc GLM; SAS Institute Inc., 1999). Site, zone, season, and species were considered as main factors. The assumptions of normality and homoscedasticity for ANOVA were tested prior to analyses (Proc Univariate and Proc GLM; SAS Institute Inc., 1999). Data were transformed in order to meet the assumptions for ANOVA when required. Seasonal differences in porewater salinity were evaluated using a t-test (Proc ttest; SAS Institute Inc., 1999) due to unbalance observations between group pairs. Nonparametric correlations (Proc Corr; SAS Institute Inc., 1999) were conducted among structural attributes and porewater salinity.

Results

Forest Structure

Mean basal areas were significantly different among sites ($p = 0.0034$). Site S4 presented the highest basal area (36.6 m^2 ha^{-1}), while sites S1, S2, and S3 showed similar values (fig. 14). There were no significant differences ($p = 0.8849$) in mean basal areas among species. *A. germinans* and *R. mangle* had similar basal areas in our study sites (13.6 m^2 ha^{-1} and 12.6 m^2 ha^{-1}, respectively). *Avicennia germinans* was found in all sites, and this species represented 56-100% of total tree density among the sites (table 7). The relative importance (I_v) of *A. germinans* was high in S2, S3, and S4 and differed from S1, where *R. mangle* was the most important species (table 7). In contrast, the I_v index by zones indicated that *R. mangle* was the dominant species in the fringe zone, except in S2. In this station, *A. germinans* spatial distribution was monospecific. The transition zone was mainly dominated by *A. germinans*, with few individuals of *R. mangle*. The scrub mangrove zone was dominated only by *A. germinans* (table 8).

Mean tree heights were significantly different between sites and zones ($p < 0.0001$) and ranged from 8.4 ± 0.36 m in S1 and S4 to 5.4 ± 0.40 m in S2 (fig. 15a). Moreover, the fringe zone presented the highest tree height (10.2 ± 0.43 m) among all sites, followed by the transition zone (5.91 ± 0.21 m). The scrub mangroves zone has the lowest mean tree height (1.87 ± 0.11 m) (fig. 15b). The total density of all trees with dbh (diameter at breast height) ≥ 2.5 cm was lower in S1 (1,507 trees ha^{-1}) and S4 (1,795 trees ha^{-1}) compared to S2 (2,395 trees ha^{-1}) and S3 (2,581 trees ha^{-1}) (table 7).

Soil chemistry

Mean pore water salinities were not significantly different between sites ($P = 0.8210$). Mean salinity values in all sites ranged from 55 g/L ± 7.5 (S1) to 61 g/L ± 6.2 (S2) (fig. 16a). On the other hand, mean salinity values between zones were highly significant ($p < 0.0001$). The fringe zone presented the lowest salinity (48.2 g/L ± 3.1), while the highest (96.6 g/L ± 4.7) salinities were registered in scrub mangroves zone (fig. 16b). Porewater salinity was significantly different between dry and wet seasons (49.0 g/L ± 1.8 and 67.8 ± 3.5, respectively; $p = 0.01$). Porewater temperature did not vary significantly by sites or season; values ranged from 29ºC (dry season) to 31.7ºC (wet season).

Porewater sulfide concentrations were not significantly different between sites ($p = 0.1089$), zones ($p = 0.4319$), or season ($p = 0.2758$). Concentrations between zones ranged from 0.004 ± 0.12 mM (scrub mangroves) to 0.135 ± 0.1 mM (fringe). All sites showed the highest values during the wet season when concentrations ranged from 5.17 E-10 mM ± 0.0 mM (S4) to 0.33 ± 0.99 mM (S1) (fig. 17). The soil pH was slightly acid to neutral in all sites ranging from 5.04 (S1) to 7.29 (S2). Significant differences in soil Eh were observed between sites ($p = <0.0019$) and depths ($p = <0.0001$). In contrast, zones did not have a significant effect ($p = 0.49$) on sulfide concentrations. Soil Eh between sites ranged from +300 mv (S3, 0 cm depth) to -77 mv (S1, 45 cm depth). Seasonal differences ($p = <0.0001$) indicated higher values in all depths and sites in soil Eh during the wet season than in the dry season (fig. 18).

Mean porewater $NO_2^- + NO_3^-$ concentrations were significantly different between sites (p = 0.0372), varying between 3.9 ± 0.5 µM (S2) and 1.7 ± 0.6 µM (S4). Seasonal variation in $NO_2^- + NO_3^-$ concentrations was significant (p = <0.0001), and higher concentrations were detected in all sites during the wet season (fig. 19a). Porewater NH_4^+ concentrations varied significantly (p = <0.0001) between sites; S1 had the highest concentrations (5.2 ± 0.8 µM), while S3 had the lowest values (0.65 ± 0.1 µM). All sites had higher NH_4^+ concentrations during the dry season, (p = 0.0002; fig. 19b). Mean porewater PO_4^{3-} concentrations were only significantly different (p = <0.0001) among seasons. Higher values were reported during the dry season ranging from 3.6 ± 2.2 µM (S1) to 8.2 ± 2.1 µM (S3) and S1 (fig. 19c). In general, there were no significant differences in porewater PO_4^{3-} concentrations along transects in all sites.

Nutrient concentration in soil samples

Mean total C concentrations in the top 20 cm of mangrove soils were not significantly different between sites (15.5 mg g^{-1} ± 1.7 – 19.4 mg g^{-1} ± 1.7). However, C values decreased significantly (P = <0.0001) along each transect. The fringe zone showed the higher concentration (22.3 ± 1.6 mg g^{-1}), while the scrub mangroves zone the lower (9.8 ± 1.6 mg g^{-1}). In contrast, total N concentrations were lower in the fringe (2.5 ± 0.4 mg g^{-1}) and higher in the transition zone (3.9 ± 0.6 mg g^{-1}). P concentrations ranged from 0.5 ± 0.2 (transition) to 0.6 ± 0.2 mg g^{-1} (fringe zones) and were not statistically significant (table 9).

C:N ratios (mass) ranged from 15.2:0.97 to 7.2:5 in all sites. C:N values were higher in S1 and S2 than in S3 and S4 (fig. 20a). N:P ratios ranged from 7.7:0.43 to 1:0.6 and increased with distance from the fringe to the scrub zone at S3 and S4 (fig. 20b). This trend was also observed in S1 and S2 sites.

Discussion

Forest Structure and Soil Chemistry

The geomorphic characteristics of a coastal region, together with geophysical and biogeochemical processes, control the basic patterns in forest structure and growth (Thom, 1984). Mangrove species distribution is influenced by several environmental gradients that respond either directly or indirectly to particular landform patterns and physical processes (Woodroffe, 1992). The forcing functions that constitute the energy signature of mangrove ecosystems, such as solar radiation, wind, precipitation, river flows, and tides, determine in large part the network of energy flow and material cycling that develops within the system (Twilley, 1995). These forcing functions along with biological interactions influence ecological processes in mangrove such as productivity, biomass, succession, litter dynamics, nutrient cycling, and sedimentation (Twilley, 1995).

Zonation of mangrove species reflects ecophysiological response of the plants to one or a series of environmental gradients (Woodroffe, 1992). The combination of abiotic factors such as frequency and duration of inundation, waterlogging of substrate, porewater salinity, nutrient availability, soil redox potential and pore water sulfide concentrations (McKee and others, 1988) determine mangrove species distribution (Odum and others, 1982; Woodroffe, 1992). Other environmental factors that account for mangrove distribution are climate, soil composition, wave energy, topography, and sedimentation (Chapman, 1976; Odum and others, 1982; Woodroffe, 1992).

The structural and functional attributes of dry climate mangroves along the Pacific coast of Central America differ significantly from those described for the Caribbean region (Cintron and others, 1978). It has been recognized that the spatial distribution of mangrove forests of arid environments is the result of climatic, edaphic, and hydrologic conditions occurring along the coast (Pool and others, 1977; Cintron and others, 1978; Jimenez and Soto, 1985; Jimenez, 1990). Mangroves in the study area are exposed to long periods of dry season (six months) and seasonal rainfall that determined the arid conditions of this environment. These harsh conditions are reflected in high air temperatures and high evapotranspiration rates which increase soil salinities in inland forest. However, the amount and frequency of tidal flooding in certain mangrove areas (near to tidal creeks) decrease soil salt accumulation allowing a better structural developed of mangrove species, particularly *R. mangle*.

Significant differences in mean basal areas were found among sites (22.3 – 36.6 $m^2\ ha^{-1}$) and were associated with differences in height within species and zones. These results are similar to those reported by Pool and others (1977) in the Pacific Coast of Costa Rica. They found shorter canopies (9.5 – 10 m) and lower basal areas (23.2 – 32.9 $m^2\ ha^{-1}$) of fringe and riverine mangroves of arid environments comparable to those structurally developed mangroves in the Caribbean coast of Costa Rica with larger basal areas (96.4 m^2 ha^{-1}).

Changes in structural development of mangroves in the southern region of the Gulf of Fonseca were observed under different hydrological conditions. The interior zone (fringe) was constantly influenced by a strong tidal regime. This zone was dominated by *R. mangle* as indicated by its high basal area (34.3 m^2 ha^{-1}). In contrast, the transition and scrub mangroves zones were characterized mainly by the presence of *A. germinans* with basal areas of about 19.2 and 16.1 m^2 ha^{-1}, respectively. This species has been recognized to growth beyond a soil salinity threshold of about 90 g/L, but in a stunted growth (Lugo and Snedaker, 1974; Cintron and others, 1978).

The distribution patterns of mangroves species associated with the spatial variation of soil Eh and porewater sulfide have been largely documented (Carlson and others, 1983; Thibodeau and Nickerson, 1986; McKee and others, 1988; McKee and others, 1993). Field studies in other areas indicate that sulfide concentrations ranging from 1.5 to 4.1 mM were significant in the distribution of *R. mangle* and *A. germinans*. Results from this study show that porewater sulfide concentrations in all sites were overall < 0.63 mM, indicating low stress. Eh in the study area can be characterized as slightly reducing with mean values varying between +89 and +149 mv. These results are similar to those reported by McKee and others (1993) in soil mangrove forests of Florida concluding that . those soils were moderately reduced (Eh = 100 –300 mv). Our results indicate that sulfide concentrations do not reach stress concentrations and cannot explain the characteristic physiognomy of the mangrove vegetation in along the sampled transects.

Soil nutrient availability has been considered an important factor regulating mangrove biomass and productivity (Chen and Twilley, 1999). Porewater $NO_2^- + NO_3^-$ concentrations were < 5 µM in all sites and were higher than reported values for other mangrove soil samples along the Shark River estuary in south Florida (Chen and Twilley, 1999). Porewater NH_4^+ concentrations were in most cases < 5 µM in our study sites and similar to reported values for mangrove areas in southwest Florida (McKee and Faulkner, 2000) and the Shark River estuary in south Florida (Chen and Twilley, 1999). Inorganic PO_4^{3-} concentrations were < 9 µM in all sites and significantly higher than those to reported for southwest Florida (< 1 µM) (McKee and Faulkner, 2000) and the Shark River estuary (< 2 µM) (Chen and Twilley, 1999). In contrast, our porewater PO_4^{3-} concentrations cpmared (0.1-35.2 µM) to those reported by Boto and Wellington (1984) in pore waters of mangrove forests in Australia.

Differences in concentrations of inorganic nutrients among mangrove estuaries can be attributed to local characteristics, such as the extent of freshwater and groundwater input and the productivity of the biota (Alongi and others, 1992). It has been recognized that the concentration of dissolved phosphorus in mangrove soils decrease with increasing salinity (Wong, 1984). Our results show that porewater PO_4^{3-} concentrations were lower in the scrub mangroves zone. The low concentrations of inorganic nutrients (e. g., ammonium) found in most of the zones indicate an active nutrient cycling related to mangrove plant growth, soil reabsorption, and microorganism activity (Boto and Wellington, 1984; Alongi and others, 1992).

The C:N ratio in the study area was < 20 and compares to values reported for the lower region of the Shark River estuary (Chen and Twilley, 1999). In general, mangrove sites located along El Pedregal estuary (S1 and S2) showed lower C:N than sites along San Bernardo estuary.

The low concentrations of TP in mangrove soils along San Bernardo and El Pedregal estuaries results in N:P ratios < 20 in all sites. Similar results were reported by Chen and Twilley (1999) at the lower estuary of the Shark River estuary. These results suggest that P is a limiting nutrient in both the fringe and scrub zones. However, tree height was different between these area similar concentrations of P were found. This trend suggest that another soil stressor (e. g., salinity) may influence the spatial distribution and stature of mangrove species along the elevation gradient in the study area.

Hypersalinity has been recognized as one of the major factors limiting mangrove forest stature and growth (Lugo and Snedaker, 1974; Cintron and others, 1978). The spatial distribution of mangroves in dry environments is strongly related to soil salinity gradients (Soto and Jimenez, 1982). Also, the effects of extreme environments can limit the structural development of mangrove forest (Pool and others, 1977). High soil salinity concentrations interfere with enzymatic reactions reducing protein synthesis, alter the osmotic potential in tissue, and produce a rapid loss of photosynthetic activity (Scholander and others, 1965; Ball, 1988).

Mangrove forests in the study area present a conspicuous distribution that is related mainly to soil salinity gradients and tidal flushing. *R. mangle* was found in lower elevations near to tidal creeks and border of channels where salinities were lower (30-50 g/L), while *A. germinans* occurred in the highest elevation with higher salinities (60-140 g/L). Soto and Jimenez (1982) have reported similar results in the Pacific Coast of Costa Rica. They found *Rhizophora* species occupying the lower ridges of channels with salinities of about 57.5 g/L, while *A. germinans* occurred further away from the channels in higher elevations when the soil salinity was up to 155.

According to Jimenez (1990), there is a high variability in structural attributes within and between sites in mangroves of arid environments along the Pacific Coast of Central America. As a result of strong salinity gradients (30-150 g/L), these forests exhibit large differences in structural development. One striking characteristic is a reduction in height and basal area with distance away from the channels. These structural patterns were observed in our study area. The mangrove trees located near to the tidal creeks exhibited the best development in terms of basal area and tree height due to significant reduction of soil salinity, even during the rainy season.

Tree height ($r = -0.65$, $p = <0.0001$) and basal area ($r = -0.42$, $p = 0.0030$) were correlated negatively with soil salinity indicating a better forest development when salinity values were lower. These results are similar to those reported by Cintron and others (1978) in Puerto Rico and Soto and Jimenez (1982) in Costa Rica. They found that tree height ($r = 0.72$) and basal area were inversely related to soil salinity ($r = 0.92$).

Conclusions

The conspicuous structural patterns of mangrove forest in the southern Gulf of Fonseca are strongly related to the frequency of tidal flushing and salinity gradients, which determine the zonation and species composition within the study area.

The spatial distribution of mangrove species was significantly different along an elevation gradient; *A. germinans* was found in higher elevations associated with high salinities (>60 g/L), while *R. mangle* was located in lower elevation where salinity was lower (<50 g/L).

Salinity is a stress factor affecting growth rates and spatial distribution of mangrove forests in the study area. Salinity stress has been alleviated by hydrological modifications as a result of shrimp mariculture development in the Pedregal and San Bernardo estuaries; areas close to channels discharging effluents support high growth rates of propagules and seedlings allowing the establishment of mangrove species that are less tolerant to high salinities (i.e., *Laguncularia racemosa*).

The spatial distribution of mangrove forests can be used to establish rehabilitation, "construction," and restoration of mangrove wetlands in areas impacted by natural and human impacts in the Gulf of Fonseca, Honduras.

Temporal and spatial distribution of water quality variables.

Methods

Water quality data was obtained from the database collected and managed by La Lujosa Water Quality Laboratory in coordination with the Asociacion Nacional de Acuicultores de Honduras (ANDAH) and the Pond Dynamics/Aquaculture Collaborative Research Support Program project (Auburn University). In general, the data set used in this report includes weekly nutrient concentrations (total ammonia, total phosphorous, soluble reactive phosphorous, nitrate, nitrite), chlorophyll a, and salinity values for 12 sites (Figure 21) located in estuaries of the southeastern region of the Gulf of Fonseca for the period 1993-2001. No data was collected during 1998 when Hurricane Mitch affected Honduras and part of Central America. Water quality information (salinity, total suspended sediments, total phosphorous and nitrogen) was also obtained from water samples collected simultaneously in effluents from 5 shrimp farms and their adjacent estuaries in the period 2000-2001 (La Lujosa Laboratory, database, Delia Martinez). Protocols for sampling and chemical analyses are described in Teicher-Coddington (1995) and Teichert-Coddington and others (2000). Values for each of the environmental variables analyzed here are monthly means of 2-4 weekly measurements.

Results and Discussion

Long term data sets

There is a strong seasonal pattern of salinity associated to precipitation. Low salinity values were registered during the rainy season (April-October), whereas high salinities occurred during the dry season (November-May) (fig. 22). The lowest salinites during the dry season were observed in the site Aquacultivos #2 where salinities were < 25 ppt and salinities close to zero were more frequent during the dry season (fig.22). An increase of salinity during the dry season was observed for the years 2000-2001. Salinities were significantly higher in January and February 2001 (45 ppt) than in 2000 (35 g/L) in estuaries north and south of Punta Guatales (fig. 22).

Soluble reactive phosphorous (SRP) was generally higher in estuaries north of Punta Guatales (fig. 23). The highest concentrations were measured in the El Garcero site located in the Pedregal estuary (fig. 23). Mean concentrations in this site ranged from 1.5 to 20 µM, with the highest values generally observed during the dry season throughout the sampling period (fig. 23). Another site with similar trend was La Lujosa, where high mean values were also measured during the dry season. This site is located along the Rio Choluteca more than 30 km upstream from the Pedregal estuary (fig. 24). Mean values higher that 10 µM were commonly observed from1993 to 1997, decreasing between 3-8 µM beginning in 1999. This reduction in concentration might be related to hydrological changes caused by Hurricane Mitch (1998), which diverted part of the water flow towards the west (old tributary, Gulf of Fonseca) decreasing freshwater flow into the Pedregal estuary through the La Jagua estuary. Sites located south of Punta Guatales (GSMB #2, BIOMAR, CUMAR, CRIMASA) also showed a strong seasonally of SRP values, although mean values were lower than 8 µM during (fig. 23). Higher values in these sites were consistently observed during the dry season.

The SRP values observed in all sites are among the highest reported for coastal ecosystems (table 10). Although values > 5 µM have been reported, these concentrations are generally associated to human activities such as agricultural and industrial development. For example, long-term data obtained in Perdido Bay, Florida, USA (Livingston, 2001) show a range of SRP values of 0.02-7.2 µM. One station (station P22; table 10) in this estuary was located next to a pulp mill and averaged a value of 3.87 µM (range: 0.32-7.74). Similarly, the coastal lagoon Cienaga Grande de Santa Marta (CGSM), Colombia, shows high SRP values where maximum values of 5-14 µM were measured (table 10). The hydrology of the CGSM estuary is regulated by a large watershed covered with extensive commercial agriculture (Rivera-Monroy and others, 2001). High loading rates of nutrients during the rainy season control chlorophyll *a* concentrations and water column primary productivity; due to its high productivity, this coastal lagoon is considered a hyperthrophic coastal ecosystem (Nixon, 1995; Rivera-Monroy and others, 2001). Mean SRP values for several coastal systems range from 0.3 to 3.87 µM, with a grand mean of 1.06 (± 0.19) and a median of 0.78 µM (table 10). Values measured in the sites throughout the Pedregal and San Bernardo estuaries are significantly higher than those values.

The high SRP concentrations registered in the La Lujosa station indicate that historically the Choluteca River has been an important source of SRP. The current hydrological changes caused by Hurricane Mitch may reduce this input and at the same affect the hydrology of the estuaries around Punta Guatales. The high concentrations observed in El Garcero after 1998 (presence of Hurricane Mitch) suggest that there are other sources of SRP within the study area. El Garcero is located in the upper reaches of the Pedregal estuary, and given the dimension of the channel and probably a high water residence time during the dry season, this area might retain a high percentage of the nutrient loading from shrimp farms around the area. Teichert-Coddington (1995) classified this area as an “embayment” (lack of direct influence by a river) due to its location upstream the mouth of the Pedregal estuary. He analyzed the same information presented in this report for the period April 1993-July 1994 and concluded that most of the embayment systems in the area were “pristine” compared to riverine estuaries and the Choluteca River. This conclusion contrast with the long-term trend showed in fig. 23 and emphasizes the role of the hydrology interannual variability. It is important that these same sites are continued being monitored to assess long term consequences as result of the current hydrological modifications occurring at the landscape level.

Other processes such SRP deadsorption due to low oxygen concentrations in sediments might also contribute to the observed high SRP concentrations. Mean oxygen concentrations measured (Green, unpub. data) along the Pedregal estuary, up to the intersection with El Garcero estuary (fig. 21) in the rainy season (June 2000) was 1.51

mg/L (± 0.08, N = 64). This value was higher that the concentration measured during the dry season (March 2000; 0.37 mg/L ± 0.07, N = 42). These oxygen concentrations are < 2 mg/L, which is considered the upper limit to define hypoxic conditions in coastal waters; under this low oxygen water column level, SRP could be released from the sediments into the water column. Further work needs to be developed to characterize this process in the San Bernardo and Pedregal estuaries.

Nitrate plus nitrite (N + N) concentrations were also consistently higher in all the sampled sites. Similar to the SPR trends, higher values and higher variances were observed in sites north of Punta Guatales (fig. 25). The values in this area ranged from 0.1 to > 80 µM. Some seasonal pattern was apparent in few sites particularly for La Lujosa station, La Jagua, and GSMB #1. The wide variation for most of these sites indicates the complex hydrological and geochemical process regulating N+N concentrations, particularly the input of nitrogen from the Choluteca River and the effluents from shrimp ponds. The seasonal variation is apparent for all the stations south of Punta Guatales (fig. 25).

There is a cyclic pattern where higher concentrations are observed during the rainy season throughout the entire sampling period (fig. 25). Generally, in river-dominated systems, nitrate concentrations increase as result of river discharge during the rainy season, but in this case, there is the presence of higher concentrations is during the dry season. This pattern might be the result of the low dilution due to the lack of freshwater input during periods of shrimp pond effluent discharge. In contrast to SRP concentrations, N + N values have increased since 1999, indicating that N + N concentrations might no be controlled by freshwater discharge but more directly by local biogeochemical processes. N + N concentrations in all sites are within the range reported for other estuarine systems (table 10). The range of concentrations found in both dry and rainy season suggests that primary productivity might not be N limited in the region.

Mean total ammonia (TA) concentrations follow the same pattern that N+N; higher values and variances are found in the northern sites than in the southern sampling sites (fig. 26). Seasonal changes are also observed in the southern area; however, the higher values were registered in the rainy season. Because of the high variability within each month, it is difficult to differentiate temporal patterns for each station, indicating that TA is controlled for several factors at small temporal scales.

Despite the high concentration of total suspended sediments, chlorophyll a (Chla a) values can reach high concentrations (60-70 µg/L) (fig. 27), particularly during the dry season. This increase might be due to less turbulence in the water column due to low river discharge increasing the photic zone in all sites. Chla a concentrations are similar in all sites north of Punta Guatales, with an average annual mean of 35 µg/L. In contrast to the northern site in the Pedregal and Jagua estuaries, mean annual Chla a concentrations are < 15, although higher values can be observed (50-60 µg/L). These values compared to chla a concentrations reported for other coastal ecosystems (table 11). Seasonal variation is apparent in all years, and no long-term changes were observed as result of lower river discharge and a longer dry season in 2001.

Effluents vs. estuarine concentrations.

Temporal salinity trends in five farms show significant seasonal variations. As in the case of the long-term data set, salinity is low (< 8 ppt) during the rainy season in the estuaries, particularly during the months of August, September, and October (fig. 28). It is apparent in all shrimp farms that salinity is higher in the effluent during the dry season. The higher difference between effluent and estuarine waters during this season was observed in Cadelpa and Sea Farms of Honduras (fig. 28; see fig. 21 for location of farms). This increase of salinity is the result of high evaporation during the dry season. It is expected that a net salinity export is occurring given the constant water exchange by the farms and the high evaporation rates in the region. This salt excess might contribute to an increase in salinity in the receiving waters during this part of the year.

In general, total nitrogen (TN) and phosphorous (TP) concentrations in effluents is higher in comparison to estuarine waters in all farms (figs. 29 and 30). Values for both nutrients range from 10 to 270 µM for TN and from 1 to 20 µM for TP. Seasonal differences are not apparent for both nutrients. The largest difference between effluent and estuarine water for TN was observed in the Crimasa and for TP in Sea Farms of Honduras (fig. 30). These data suggest that nutrient export from shrimp farms into the adjacent estuaries might be influencing nutrient concentrations around Punta Guatales as discussed in the previous section.

Total suspended sediments (TSS) were consistently higher in the estuarine waters that in pond effluents, particularly during the dry season (fig. 31**)**. TSS concentrations during the dry season ranged from 400 to 1000 mg/ L; the highest values were observed in Aquacultivos, Cadelpa, and Sea Farms of Honduras. Lower values of TSS in the pond effluents indicate that shrimp ponds are a net sink of sediments in the region.

An assessment to use mangrove forest to reduce nitrogen

and phosphorous concentrations in pond effluents in Honduras:

A preliminary estimation using conceptual models

We evaluated the potential utilization of mangrove forest in Honduras to ameliorate the impact of shrimp pond effluents on the water quality of adjacent estuaries in the Gulf of Fonseca. Although this method has been proposed (Robertson and Phillips, 1995; Rivera-Monroy and others, 1999; Gautier and others, 2000) for tropical coastal ecosystems where shrimp mariculture is an important economic activity, there are practically no studies directly measuring the different biogeochemical processes involved in the assimilation and recycling of P and N in mangrove forests. Given the high nutrient concentrations observed in the waterways of the Gulf of Fonseca, it is critical to explore different technologies that can contribute to avoid “self-pollution” (Csavas, 1994). This strategy should be considered as a part of a set of alternatives and “best management practices” (use feeding trays, avoid fertilization particularly during the dry season, reduce water exchange rates, etc.) that aim to reduce nutrient, sediment, and salt loading into estuarine waters (Boyd and others, in press).

As mentioned above, there is limited information of the processes regulating nitrogen cycling in mangrove forests (Twilley and others, 1999). Yet some information from different geographical regions can be used to evaluate the potential capacity of mangroves to treat shrimp pond effluents. For example Rivera-Monroy and others (1999) estimated using a mass balance approach the potential role of mangroves sediments as a sink for dissolved inorganic nitrogen (DIN). In this study, they determine that the reduction of DIN in pond effluents by preliminary diversion of outflow to mangrove wetlands rather than directly to estuarine waters would be = 190 mg N m^{-2} d^{-1}. They proposed based on this estimate that between 0.04-0.12 ha of mangrove forest was required to completely remove the DIN load from effluents produced by a 1 ha pond. In this report, we used the same approach to evaluate the mangrove forest area needed to treat shrimp pond effluents based on loading rates estimated for shrimp farms in Honduras under current management practices.

Methods

Loading rates for different farm sites were estimated using a linear programming model (Valderrama, unpub. Data; Valderrama and Engle, in review) based on nutrient budgets for semi-intensive shrimp farms located in the same areas where the water quality variables were obtained (Teichert-Coddington and others, 2000). Loading rates were estimated assuming a 5% daily water exchange (table 12). To estimate the potential treatment capacity of nitrogen (DIN and TP) and phosphorous (SRP and TP) of mangrove forest in Honduras, we used the same nutrient fluxes estimated for mangrove forest summarized by Rivera-Monroy and others, (1999). Treatment capacity here is defined as the residual of the subtraction of the total outputs from total inputs (figs. 32 and 33). A positive number indicates that the forest has an excess capacity for nutrient assimilation and utilization.

Results and Discussion

Potential treatment capacity were positive for both N and P. The magnitude of the treatment capacity of DIN was high (2.08 kg ha^{-1} d^{-1}) and compares to estimates for Colombia where semi-intensive shrimp farm practices are common (figs. 32 and 33). It is apparent that most of the DIN entering into the mangrove forest could be processed through denitrification alone. Treatment capacity for TN was also high (1.49 kg ha^{-1} d^{-1}), although this result has limited interpretation due to the processes involved in the assimilation of particulate and organic N forms by mangrove trees. One process that is directly linked to the "utilization" of TN inside mangrove forests is the accumulation rate of N in sediment. This accumulation is generally in the form of organic N and is strongly associated to sedimentation rates. Sedimentation rates in mangrove forests are relatively high, and it is expected that N accumulation through this process will be significant in mangrove forest around the Gulf of Fonseca. A mass balance of TN indicates that of the 0.64 kg ha^{-1} entering the mangrove forest per day, 30% could accumulate in the soil. And depending of the percentage of organic nitrogen remaining in the residual 70%, mineralization of this organic N will be necessary to be loss or incorporated by denitrification or plant uptake, respectively.

Rivera-Monroy and others (1999) pointed out that it was not possible to evaluate the potential contribution of mineralization and denitrification on the transformation of dissolved organic nitrogen and particulate nitrogen from pond effluents, since few studies of mineralization and nitrification rates in mangrove sediments have been conducted. Thus further information is needed to evaluate the effect of high concentrations of DON and PN in effluents on N cycling of in mangrove forests in the Gulf of Fonseca. Long-term effects are linked to the potential increase of coupled-nitrification-denitrification as a result of the increase of NH_4^+ though mineralization of DON and PN.

Potential treatment capacity of SRP (0.66 kg ha^{-1} d^{-1}) and TP (0.61 kg ha^{-1} d^{-1}) is high. SRP and TP loading rates are equivalent to daily requirements for plant uptake and much lower that for accumulation in soils. Thus, based on these rates, it is apparent that P in pond effluents could be readily taken up by mangrove forest. Although P cycling is not as complex as N cycling (for example, there is not a gaseous phase), P could be bounded to sediments in different soil fractions. The capacity of sediment to remove P from the effluents will then depend of the saturation capacity of mangrove soils in the Gulf of Fonseca region. Measurements of total phosphorous and soil pore waters (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales) in mangrove soils in the Pedregal and San Bernardo estuaries show that total P is within the range found in other mangrove forests. N:P ratios in all the areas range from 1.6 to 17.9 (mass ratio) (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales). Recent studies show that phosphorous is a limiting nutrient in mangrove carbonated (Twilley, 1995; Feller, 1995) and organic sediments (Chen and Twilley, 1999). Application of fertilizer (P) to individual mangrove trees resulted in a dramatic increase in growth rates, suggesting that P could be a potential limiting nutrient. An increase of P concentrations could also promote tree growth particularly in areas where scrub mangroves are dominant in the Gulf of Fonseca. Yet, results from field studies (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales) show that salinity has a major role as a stressor negatively affecting mangrove forest growth and regeneration rates in the Gulf of Fonseca (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales). Further work is needed to evaluate how the interaction between P concentrations and salinity in mangrove soils regulate the distribution and growth rates of mangrove forest in the Gulf of Fonseca.

We estimated the amount of mangrove forest area needed to treat farm effluents from a large farm. We used Granjas Marinas San Bernardo as a case study since water quality sampling was carried on this farm to estimate nutrient budgets by Teichert-Coddington and others (2000). In addition, mangrove field studies were performed in the mangrove forests surrounding the farm (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales), allowing an extensive characterization of mangrove forest structure. Total mangrove and pond areas for the year 2000 were estimated using the methods described in Background (figs. 10 and 13). A land use map previous to the construction of the farm (1954) is shown in fig. 9. To evaluate changes in mangrove areas, maps were developed using Landsat images for 1985,1993,1995, 1997, 1999, and 2000 (figs. 3 and 8).

Current mangroves and shrimp pond area in GMSB is 4,034 and 2,965 ha, respectively (table 4). Mangrove area estimates were partitioned in two categories based on canopy height (< 3 m and = 3m). These categories reflect mangrove development controlled by environmental factors in the region, mainly salinity. Most of the mangrove area is distributed around the south and northwestern region of Punta Guatales (fig. 10). Oxidation ponds are used to reduce sediment and nutrient loads and are located in close proximity to the Pedregal estuary (fig. 21).

Based on treatment capacities for the different type of nutrients considered in the analyses, we estimated the area needed to treat pond effluents estimated for the entire farm (2,965 ha; table 13). The largest and smallest areas were estimated to treat TN and DIN, respectively. Wetland pond ratios were less than 1 for all cases and compared to values reported by Rivera-Monroy and others (1999). Engineering designs and costs should be evaluated to provide optimum delivery of pond effluents for specific environmental settings, mangrove ecological types, and management practices. The strong seasonally in precipitation and tidal regime should be considered to select the optimal sites to divert pond effluents toward the adjacent mangrove forests. The hydroperiod in the region is very dynamic (tidal range is 1-5 m), and therefore, the selection of mangrove areas has to consider the frequency and duration of inundation to estimate the optimal residence time of pond effluents on the mangrove soil.

Effluent dispersion should be designed as a continuous sheet flow to allow homogenous interaction between the sediment and effluent, while avoiding erosion and significant changes in the in the microtopography of the forest floor (Rivera-Monroy and others, 1999).

One way to reduce engineering costs is to use selected areas within already built oxidation ponds and plant mangrove seedlings. This strategy will not have the same treatment capacity as an "mature forest" but will allow to test different combination of species and densities to evaluate optimal curves for treatment removal.

Seedling planting experiments carried on during this project indicate that growth rates are high (5-9 cm month^{-1}) for the mangrove species *Avicennia germinans* and *Laguncularia racemosa*. Growth rates under a combination of salinities values <50 g L^{-1} and high nutrient enrichment can produce trees up to 10-15 m height and a diameter at breast high of 19.7 ± 0.3 in 15 years (e.g., Canal Norte). These planted areas could be harvest periodically allowing (thinning at 9 years) wood production for local consumption within the farm or to be sold for profit in local communities. Mixed shrimp farming-mangrove forestry practices have been applied in Asia (Robertson and Phillips, 1995; Clough and others, 1999) for several years but there are not published cases in Latin America.

Differences in hydroperiod are evident around the Gulf of Fonseca as indicated by the presence of extensive salt flats along the coast as result of high evaporation, higher elevation, and salt transport by spring tides. Mangrove spatial distribution is strongly influenced by salinity in the soil and this effect is reflected on its canopy development and height. These structural features define different mangrove ecological types. For example, *Rhizophora mangle* is generally observed in the category >3 m, but it is absent in areas where mangroves are =3 m in Punta Guatales (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales). *Avicennia germinans* is the only species growing in inland areas in Punta Guatales due to its tolerance to higher soil salinities (>60 g L^{-1}). This differential distribution of mangrove species might have an effect on the capacity to treat pond effluents as result of the different physiological state of these two mangrove species. It is required to evaluate the efficiency of these two mangrove “classes” in removing pond effluents to select the most efficient mangrove vegetation ecotype in the region.

Due to potential improvement of water quality of shrimp pond effluents as result of the use of mangrove forest in the region, it is strongly recommended that this treatment be considered as a best management practice. The advantage of considering mangrove wetlands as an integral part of shrimp pond management is twofold. The first advantage is related to the development of sustainable practices that could allow the shrimp industry to take active part in the conservation of mangrove forests in coastal areas while benefiting from the ecological services that these wetlands provide. The hydrological and climatic conditions in the Gulf of Fonseca allow for the actual “construction” of mangrove forests as shown by the net increase of mangrove area in the shrimp farm Granjas Marinas San Bernardo, particularly from 1999 to 2000. Temporal analyses of the changes in mangrove areas for the period of 1954-2000 indicate that the mangrove area has increased from 3,673 ha in 1954 to 4,043 in 2000 (figs. 8 and 9). This increase is explained in part by the hydrological modifications associated to pond construction in salt flats and water diversions. Changes in hydrological conditions have modified the salinity regime by reducing salt content in the soil. This reduction, in combination with high nutrient concentrations in water inputs and effluents, has released the physiological stress of mangrove trees, allowing mangrove species generally not found in areas around Punta

Guatales (e.g., *Laguncularia racemosa*) to grow along channels and around ponds edges. Tree height of individuals of the species *Laguncularia racemosa* measured in the channel “Canal Norte” can reach up to 10 m (figs. 34 and 35); this channel did not have any vegetation in 1985 (fig. 36). Similarly, high natural regeneration rates of this mangrove species, which is low tolerant to high salinities and intolerant to low light levels, are observed in oxidation ponds close to the Pedregal estuary (see Forest Structure and Soil Properties of mangrove forests in Punta Guatales) (fig. 37).

Another advantage of incorporating mangrove wetlands as a tertiary treatment of pond effluents in farm operations is the opportunity to develop further techniques to improve uptake of nutrients through research activities. Although shrimp farm operations are strictly a profit oriented industry, it is possible to develop research programs that could allow the participation of higher education and government institutions and NGOs to financially support research addressing specific problems in mangrove ecology, particularly questions related to mangrove ecophysiology and nutrient cycling. Although these research areas are critical to develop mangrove conservation plans in the Gulf of Fonseca region, they are the less understood in mangrove ecology. There are indications that the shrimp industry is interested in the development of active mangrove conservation programs (Rivera-Monroy and others, 2001). And this interest can be translated into economic incentives within the industry to promote a rational use of mangrove forest and develop regional plans toward the sustainability of the shrimp industry not only in Honduras but also in tropical regions.

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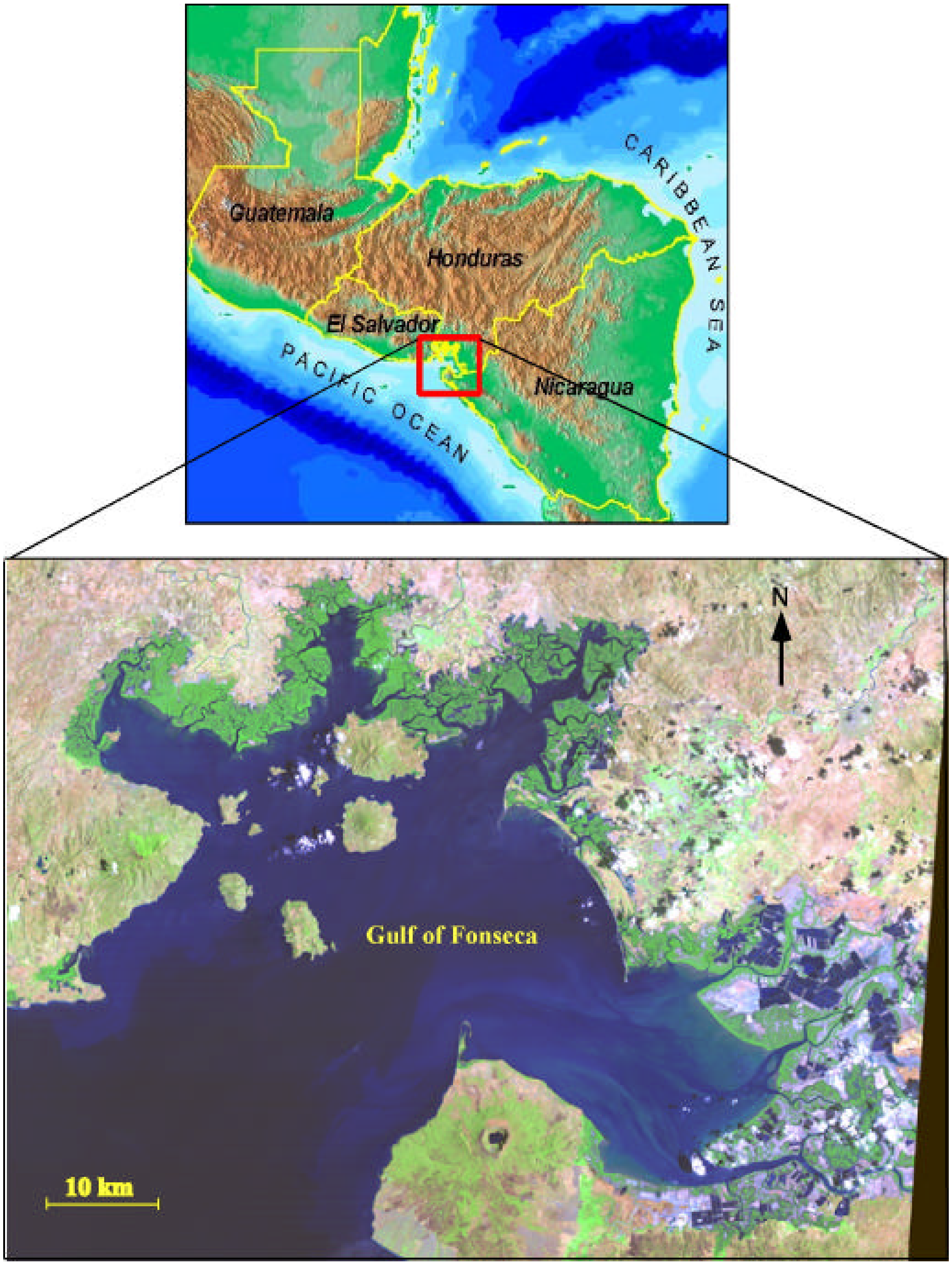


Fig. 1. Location of the Gulf of Fonseca along the Pacific Coast of Central America.

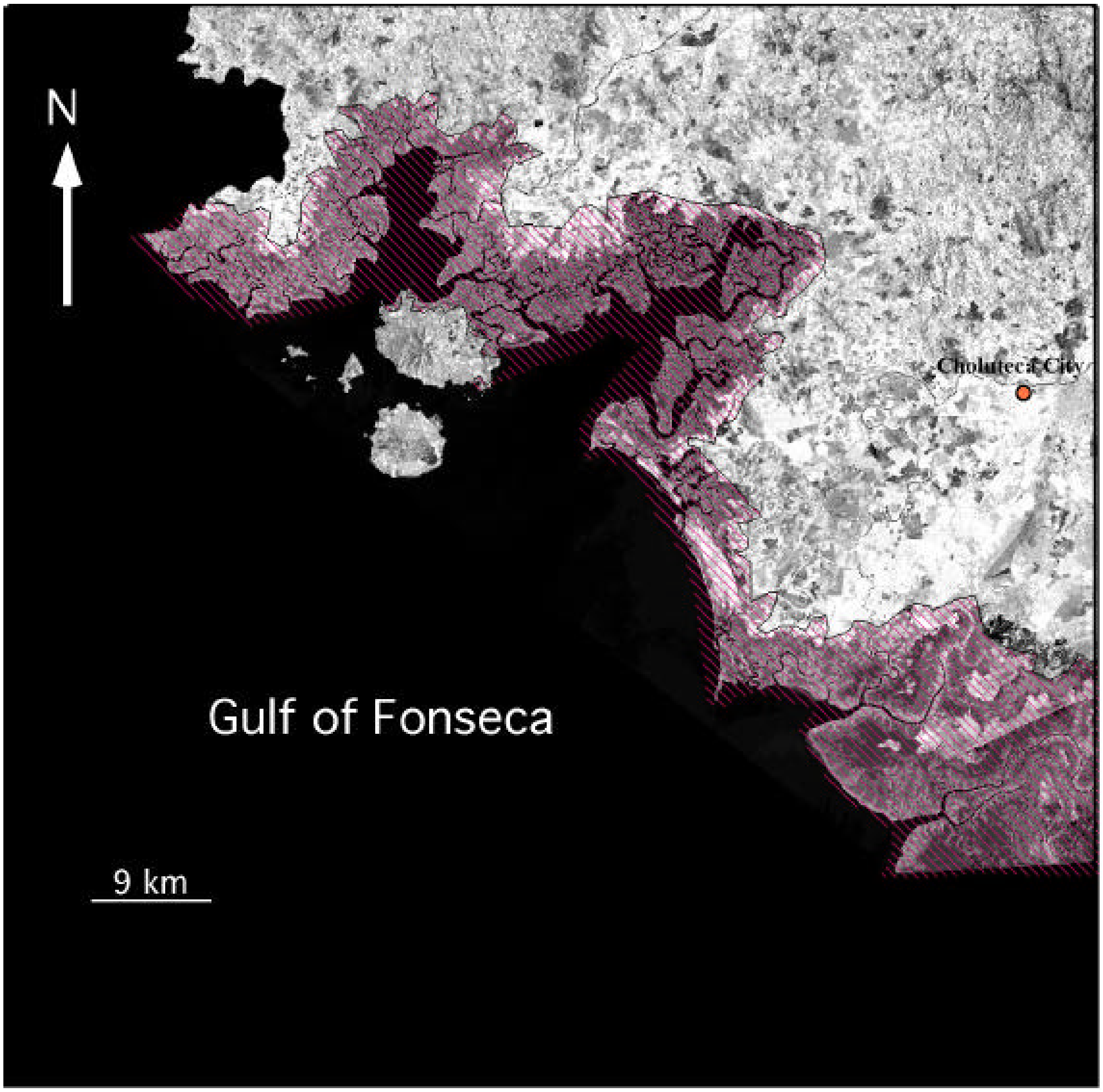


Fig. 2. "Target" area selected along the coastal region of Honduras, Gulf of Fonseca to estimate mangrove distribution and area.

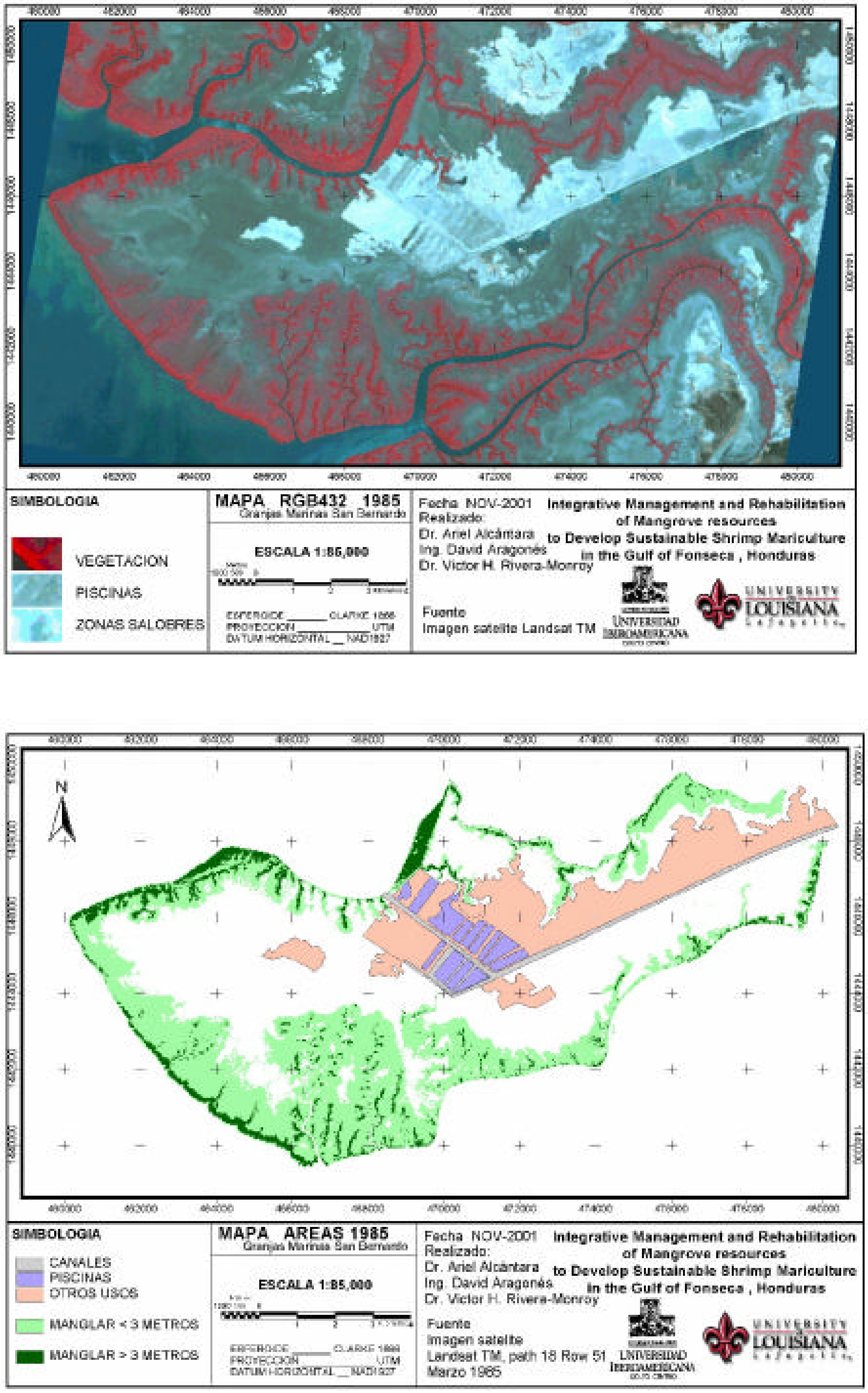
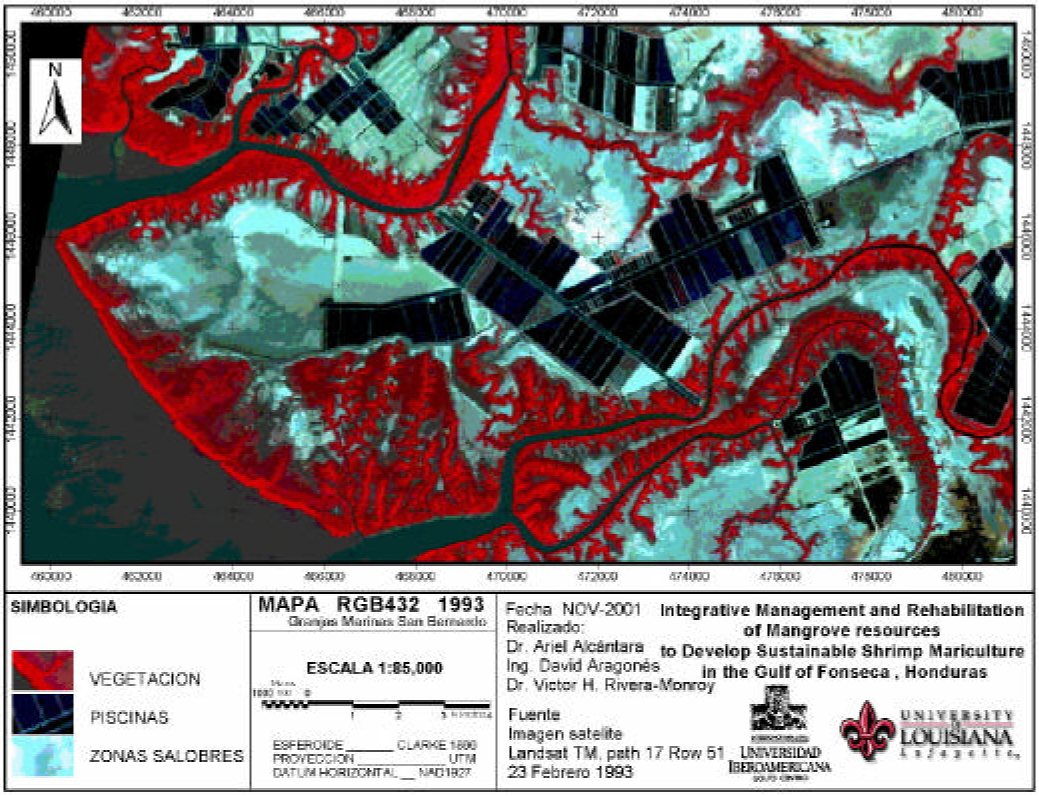


Fig. 3. Land use map for Punta Guatales, Gulf of Fonseca, Honduras (1985).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Ing. David Aragonés
Dr. Victor H. Rivera-Monroy
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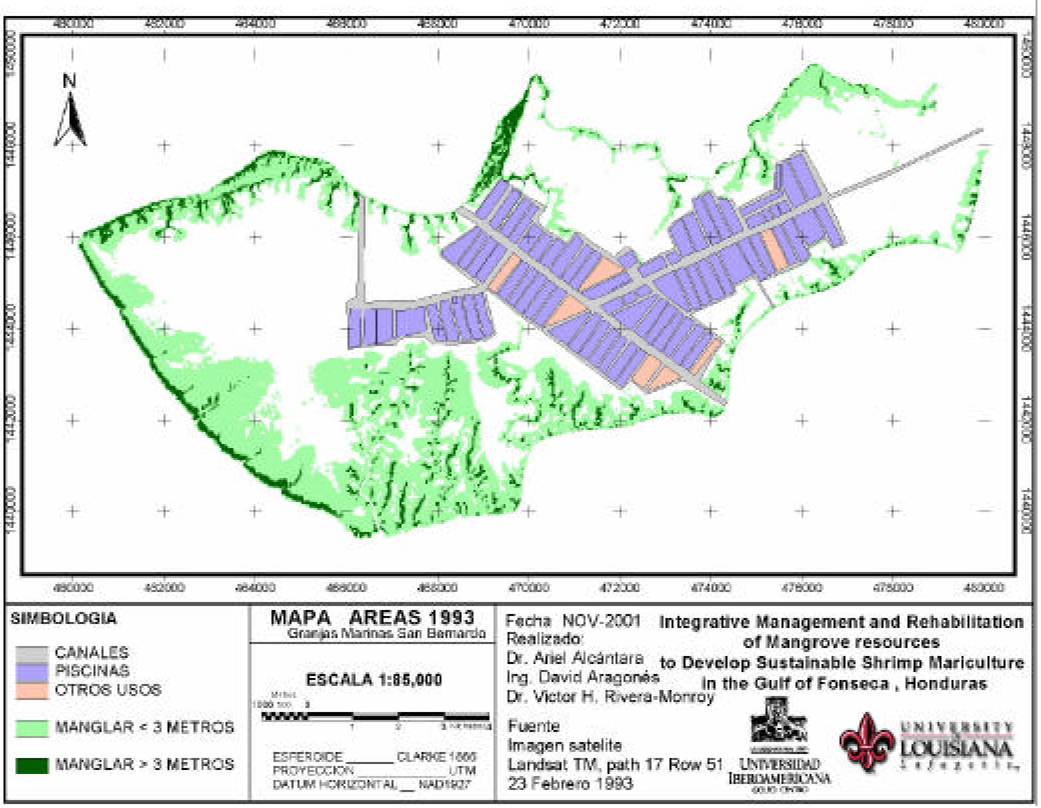
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Fecha NOV-2001
Realizado:
Dr. Ariel Alcántara
Ing. David Aragonés
Dr. Victor H. Rivera-Monroy
Fuente
Imagen satelite
Landsat TM, path 17 Row 51
23 Febrero 1993
Integrative Management and Rehabilitation of Mangrove resources to Develop Sustainable Shrimp Mariculture in the Gulf of Fonseca , Honduras
UNIVERSIDAD IBEROAMERICANA
UNIVERSITY LOUISIANA Lafayette

Fig. 4. Land use map for Punta Guatales, Gulf of Fonseca, Honduras (1993).

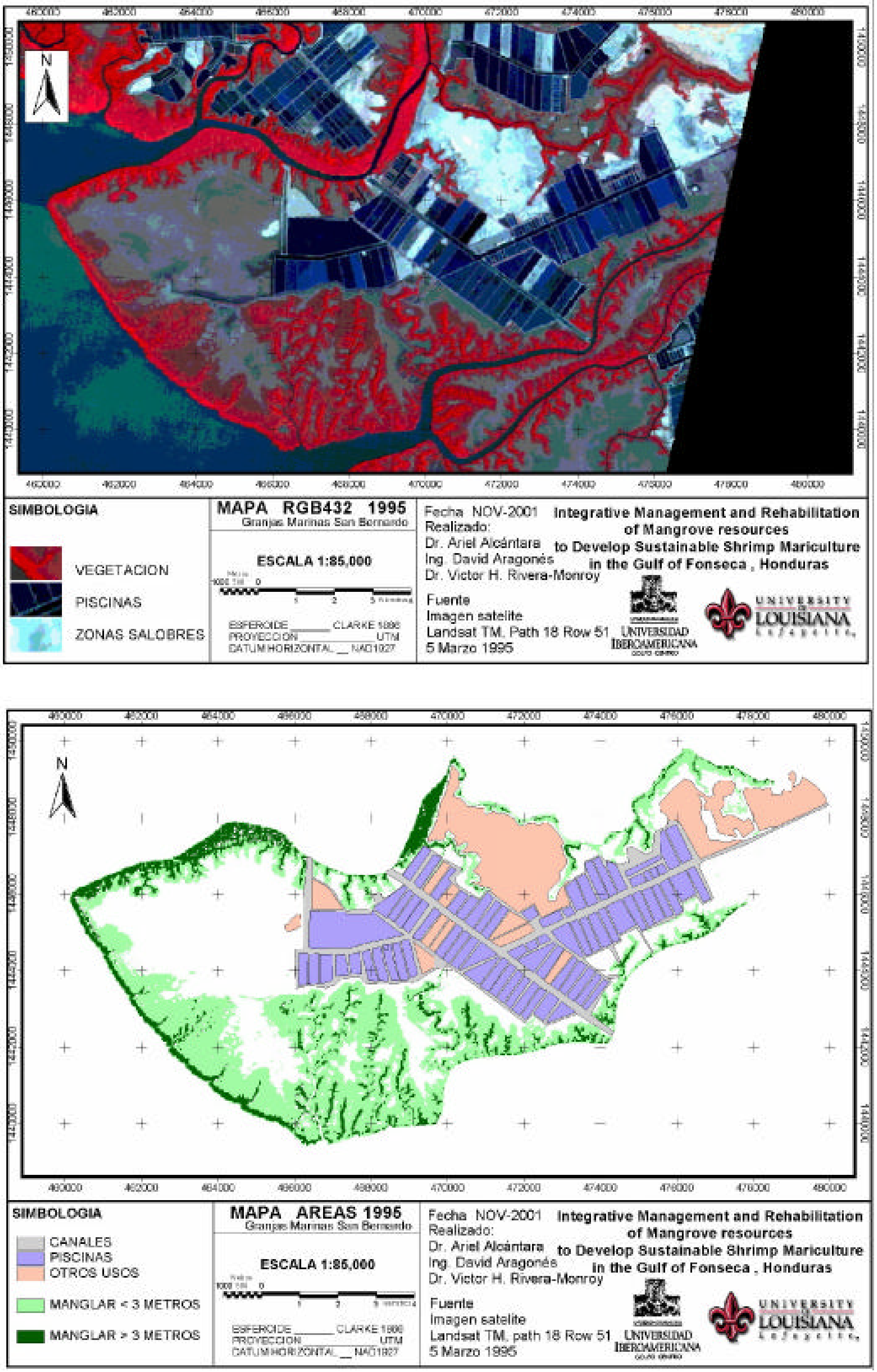
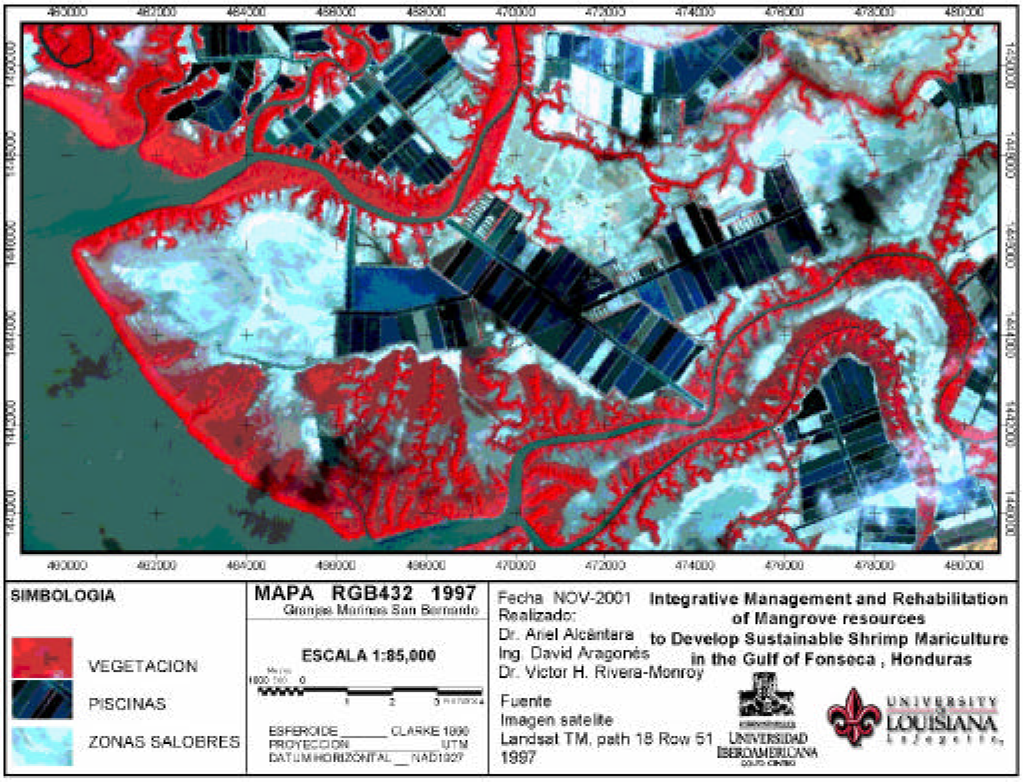


Fig. 5. Land use map for Punta Guatales, Gulf of Fonseca, Honduras (1995).



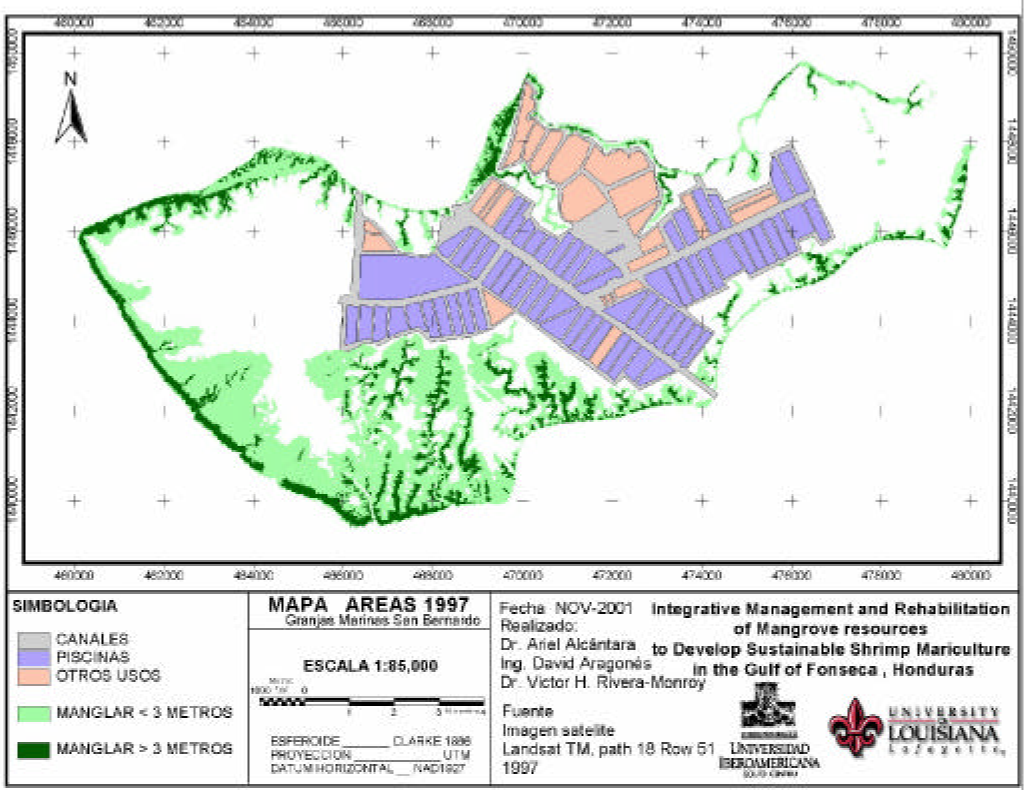


Fig. 6. Land use map for Punta Guatales, Gulf of Fonseca, Honduras (1997).

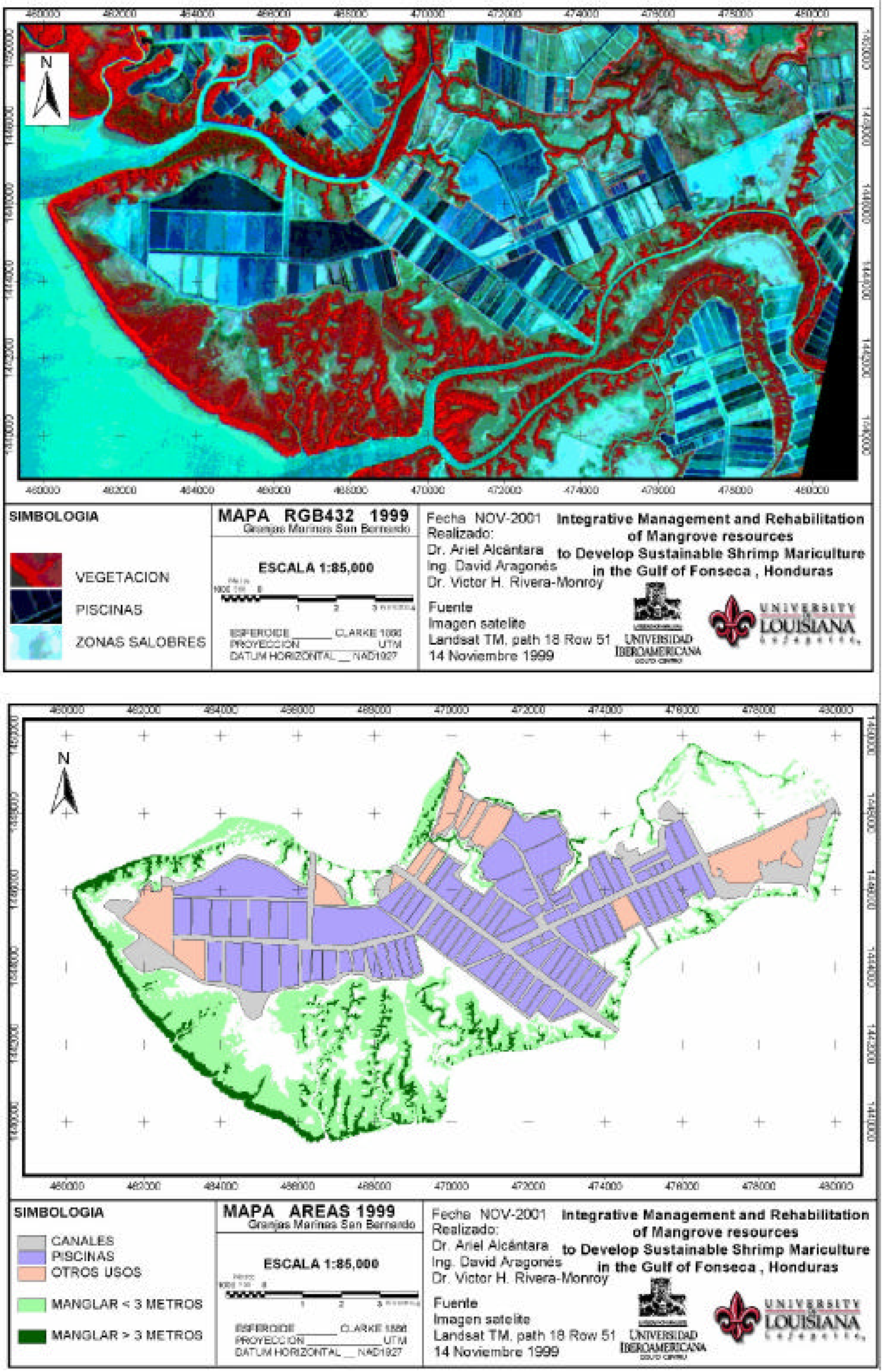


Fig. 7. Land use map for Punta Guatales, Gulf of Fonseca, Honduras (1999).

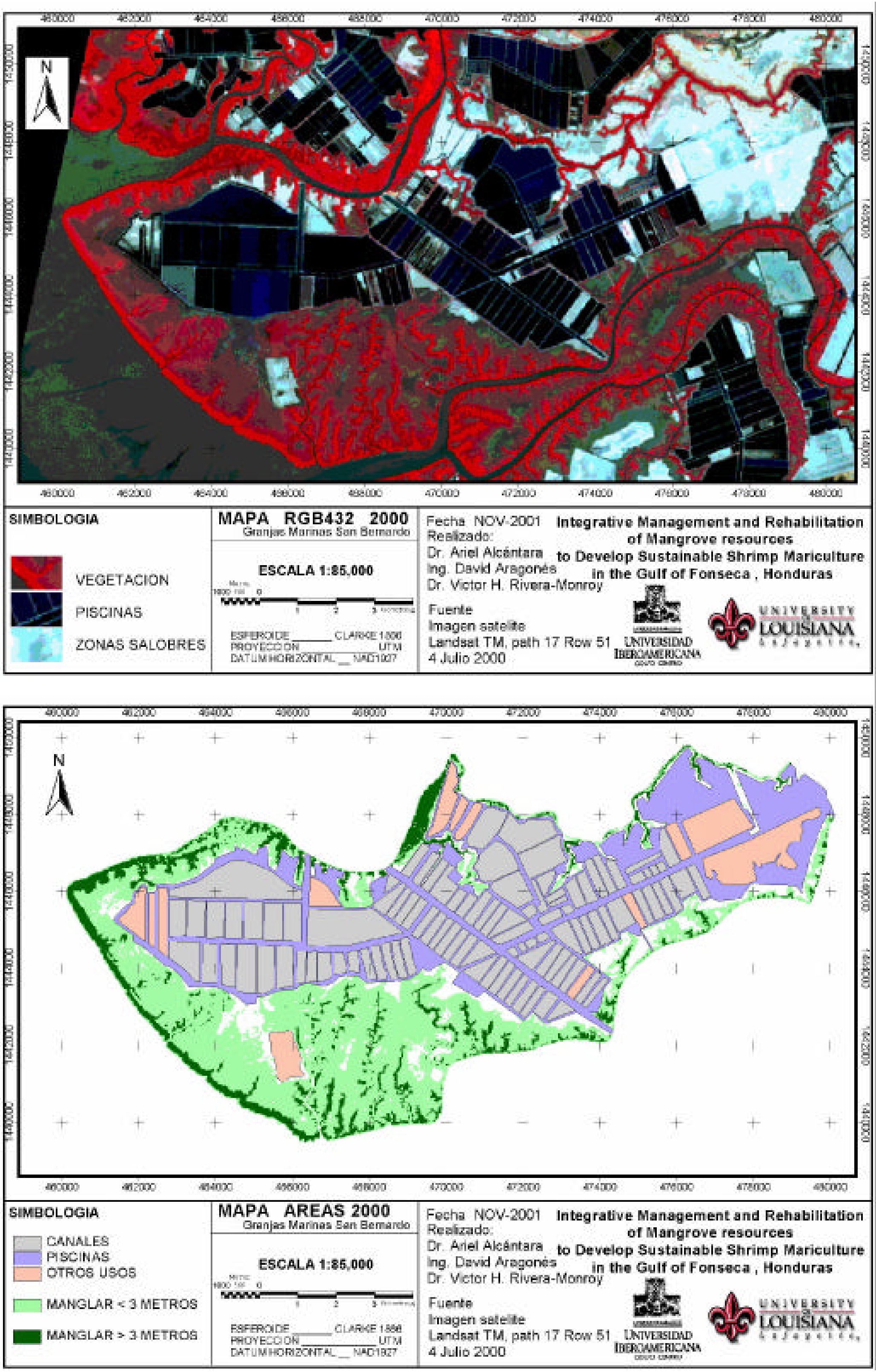


Fig. 8. Land use map for Punta Guatales, Gulf of Fonseca, Honduras (2000).

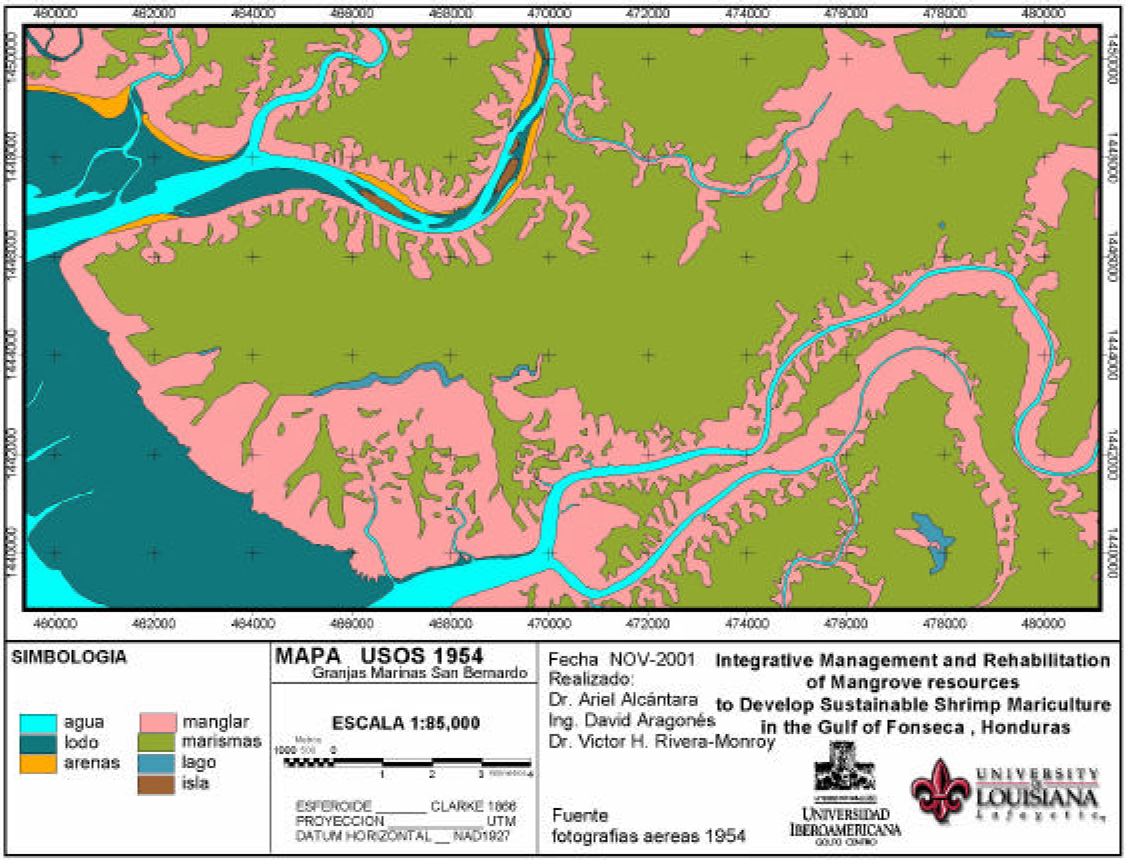


Fig. 9. Land use map for Punta Guatales, Gulf of Fonseca, Honduras (1954).

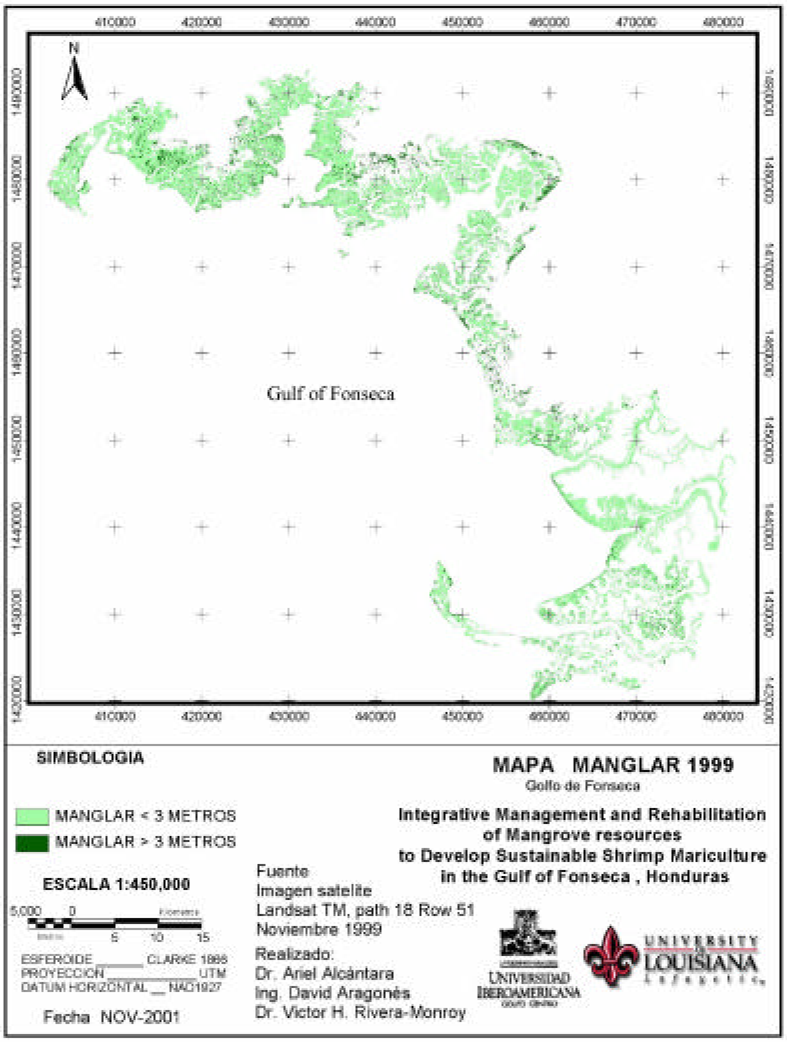


Fig. 10. Mangrove distribution in the Gulf of Fonseca (1999).

Fig. 11. Location of shrimp farms along the coast of El Salvador, Gulf of Fonseca (1999).

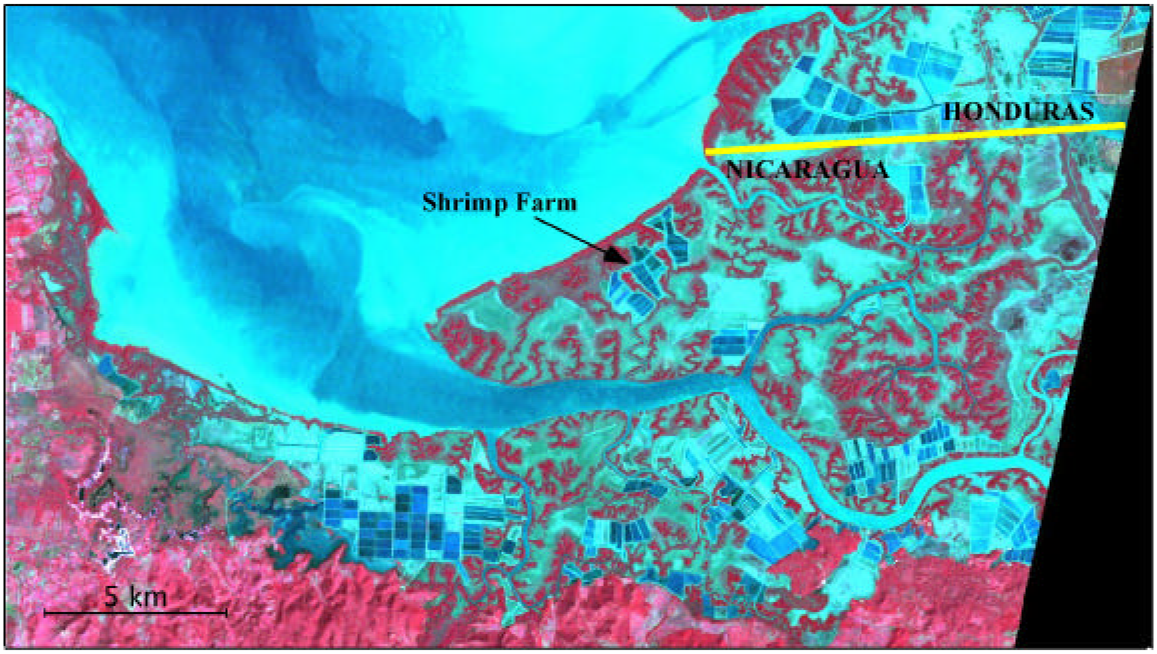


Fig. 12. Location of shrimp farms along the coast of Nicaragua, Gulf of Fonseca (1999).

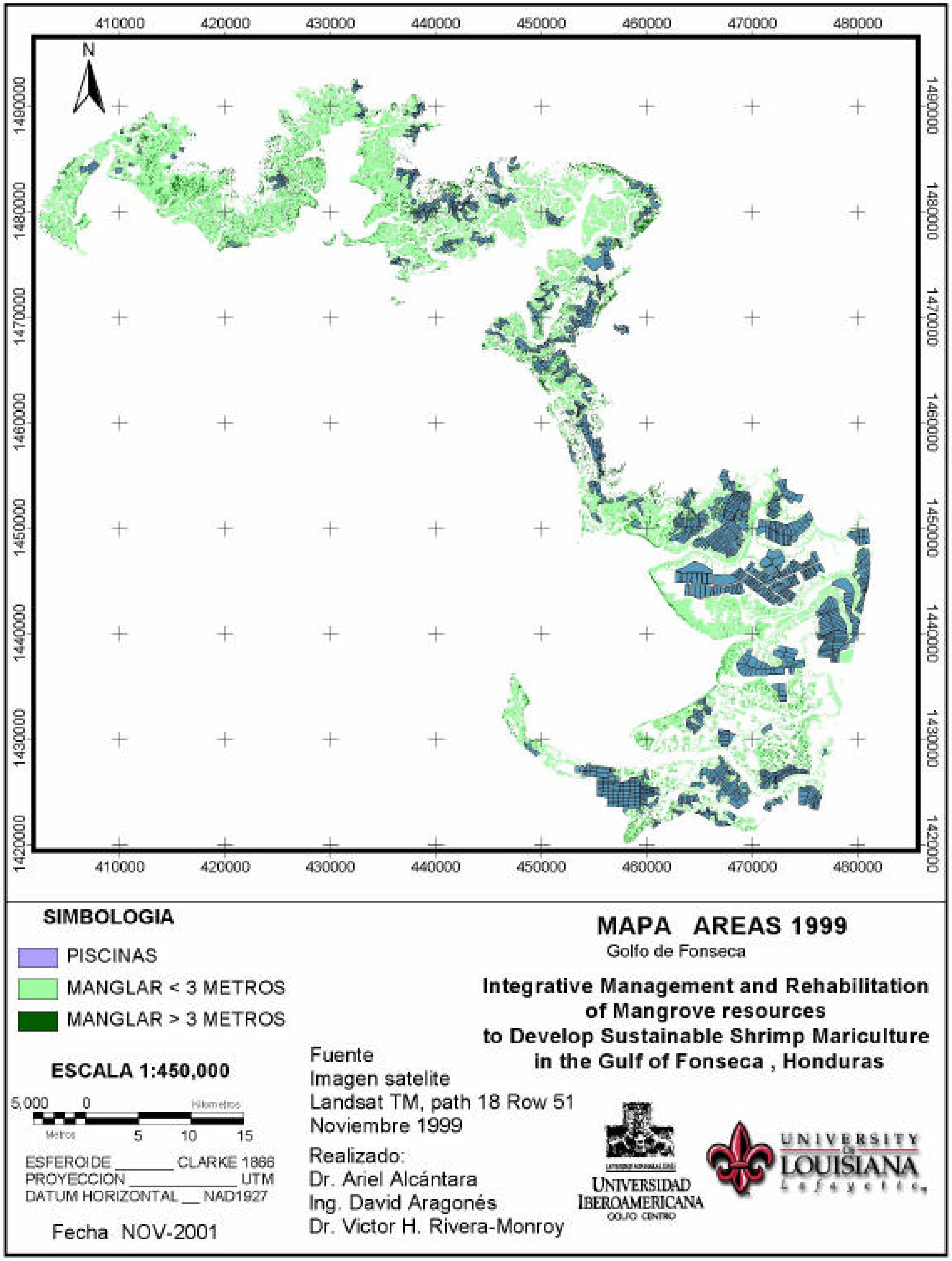


Fig. 13. Spatial distribution of shrimp ponds in the Gulf of Fonseca (1999).

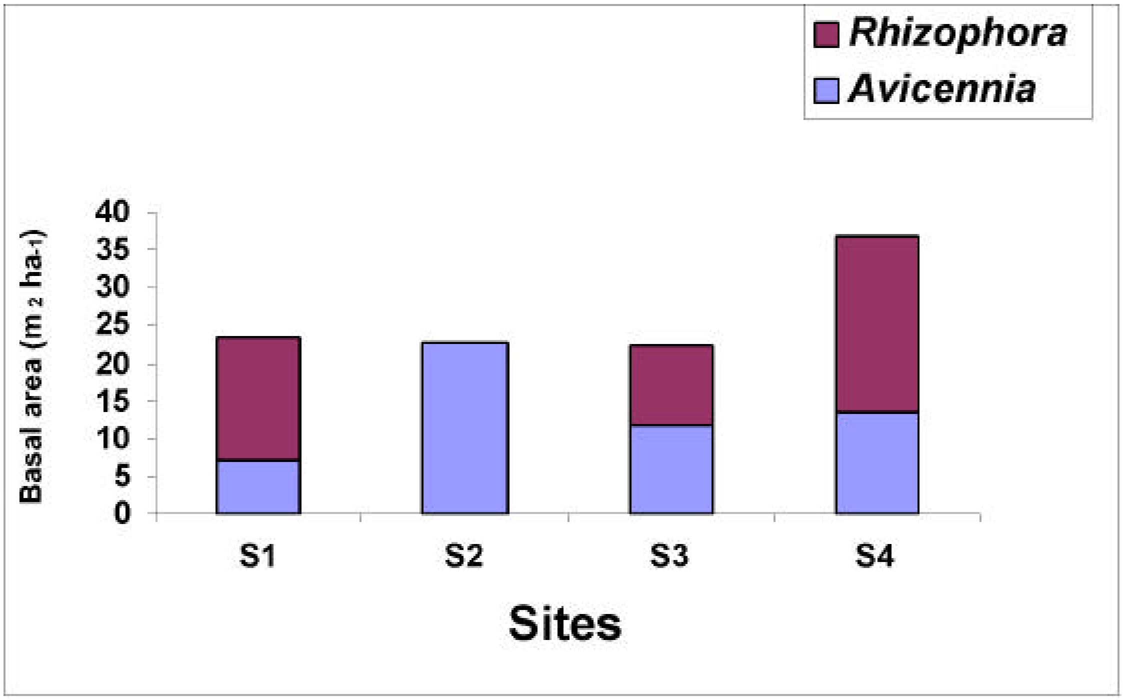


Fig. 14. Mean basal area by species at mangrove sites along the San Bernardo and El Pedregal estuaries in the southern region of Gulf of Fonseca.

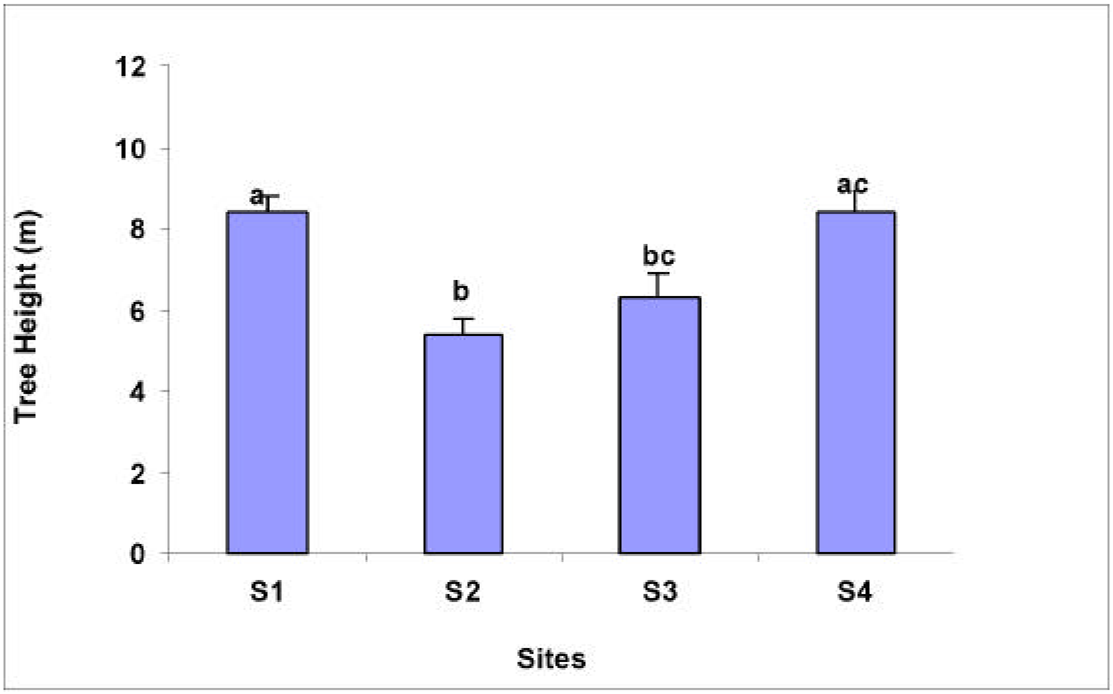


Fig. 15a. Mean tree height of mangrove forest sites along the San Bernardo and El Pedregal estuaries. Values with the same letter are not significant among sites (P > 0.05).

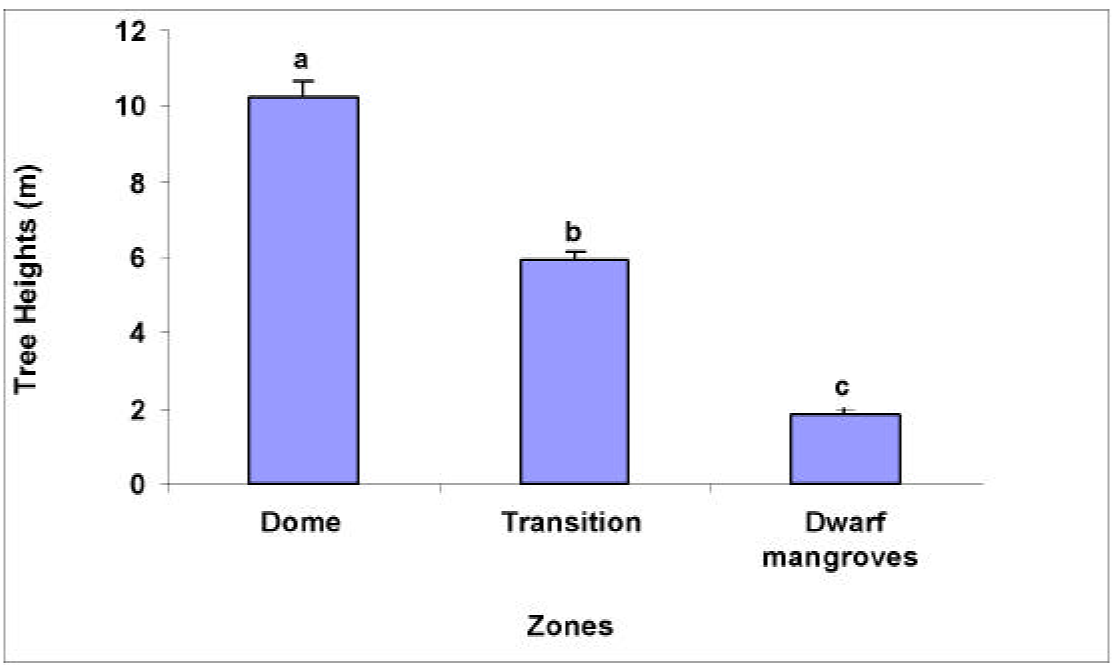


Fig. 15b. Mean tree height along the elevation gradient at sites along the San Bernardo and El Pedregal estuaries. Values with the same letter are not significant among sites (P > 0.05).

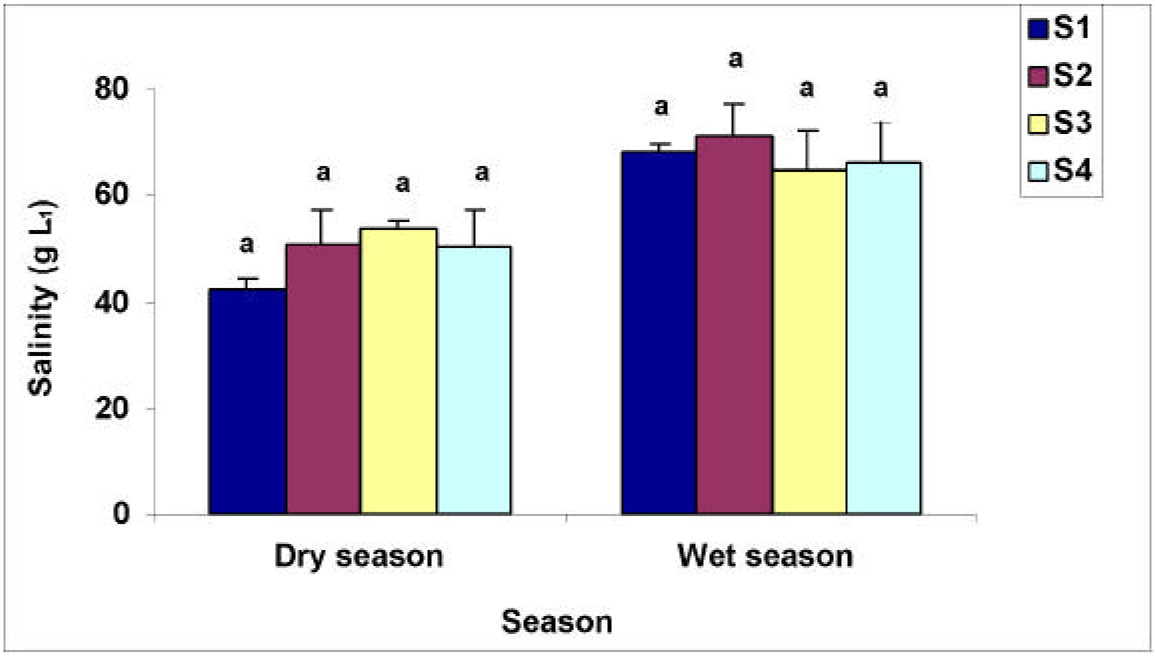


Fig. 16a. Spatial and temporal variation of porewater salinity along the San Bernardo and El Pedregal estuaries during February and August 2001. Values with the same letter at each season are not significant among sites (P > 0.05).

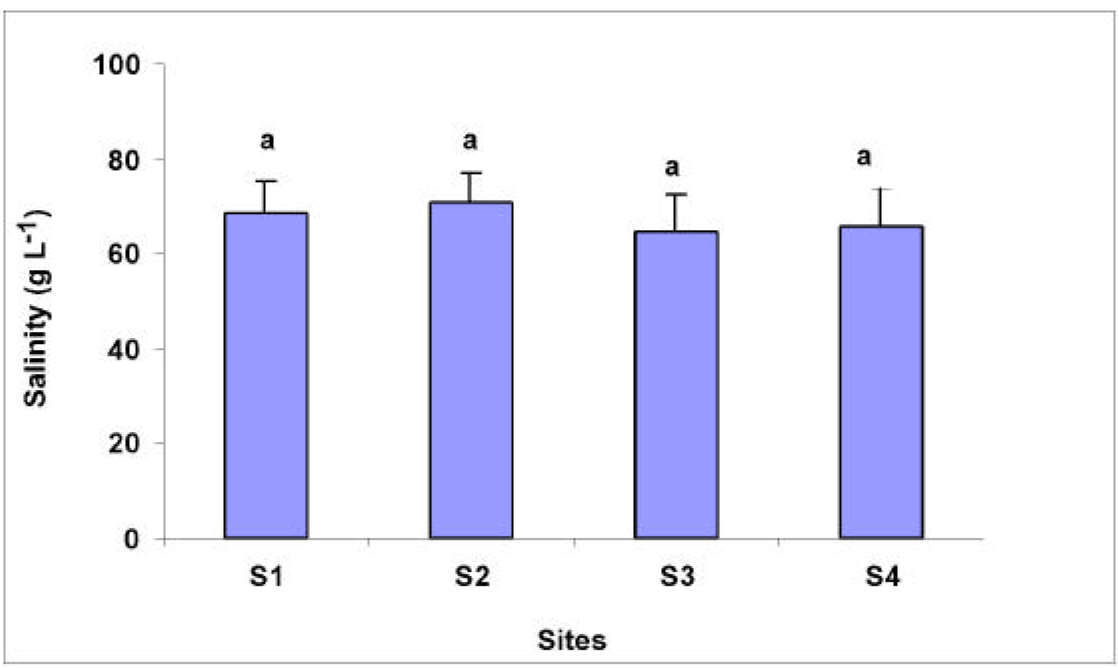


Fig. 16b. Variation of porewater salinity along the elevation gradient in mangrove sites. Values with different letter are significant among zones (P < 0.05).

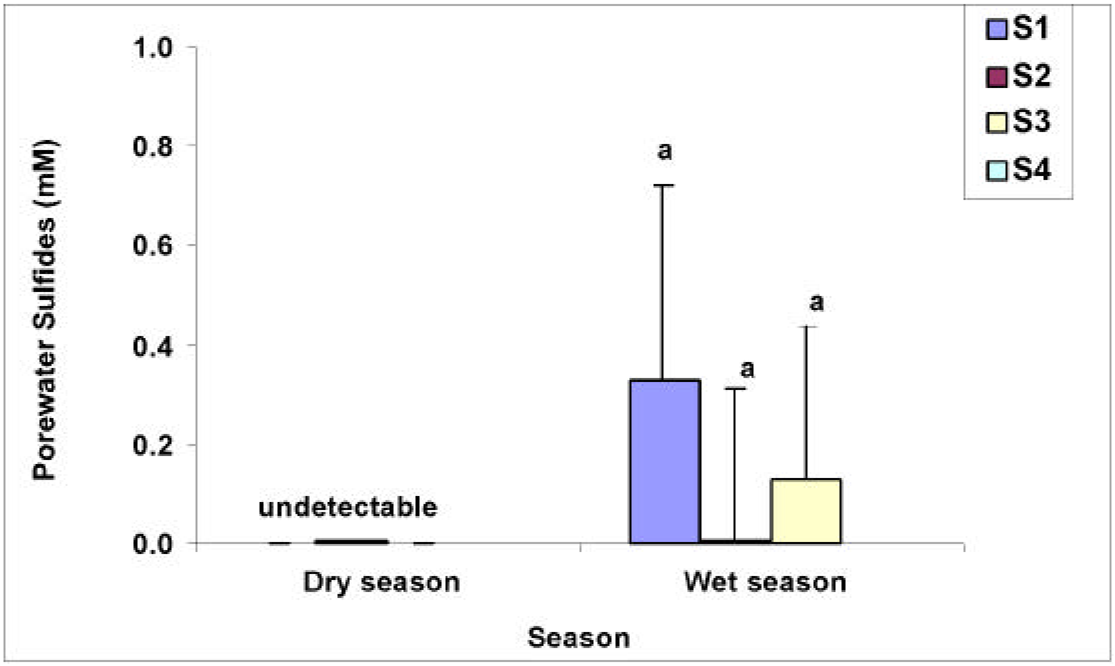


Fig. 17. Mean porewater sulfide concentrations measured in mangrove sites along the San Bernardo and El Pedregal estuaries during dry and wet season. Values with same letter at each season are not significant among sites ($P > 0.05$).

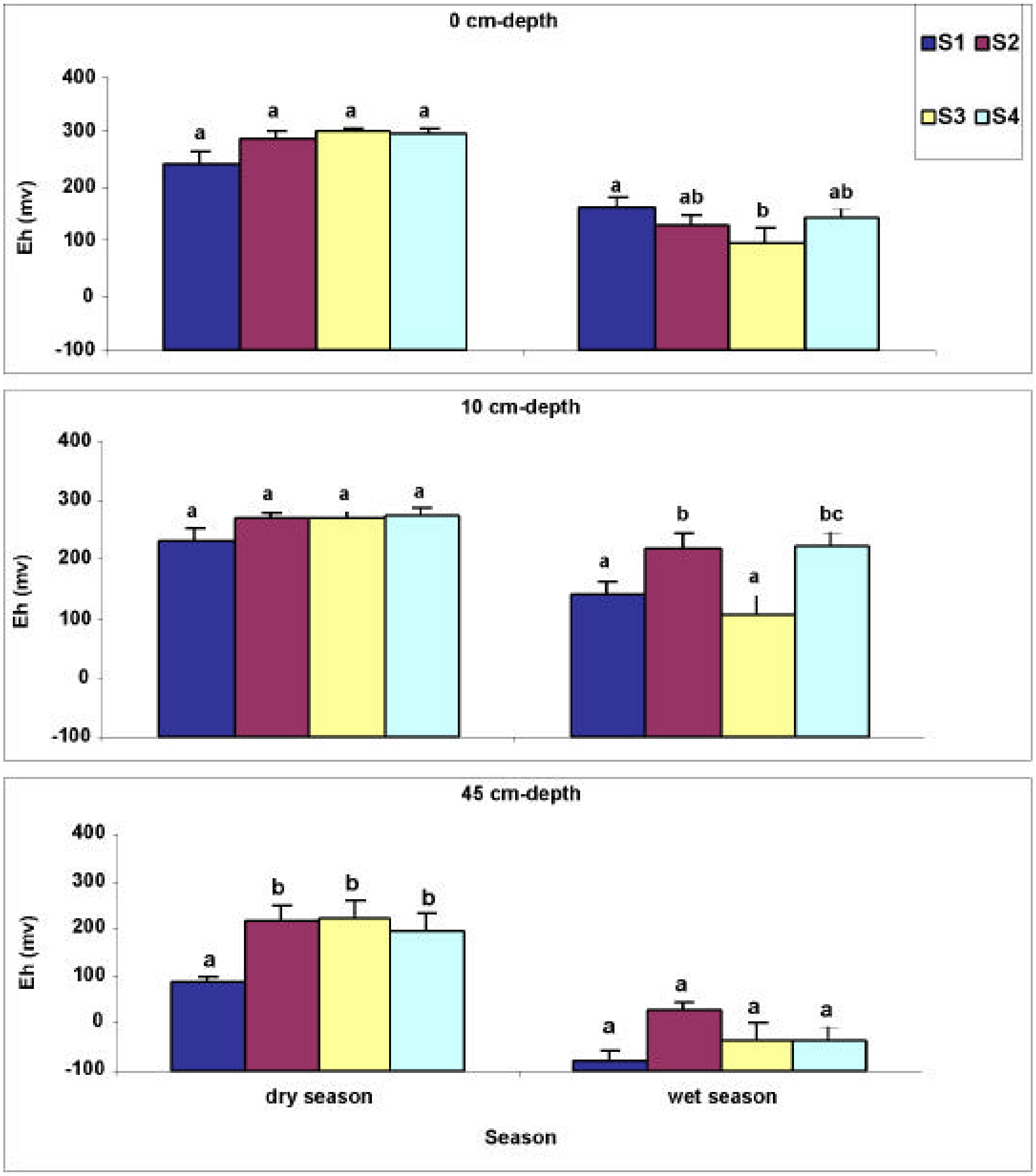


Fig. 18. Mean variation of soil Eh measured at different depths (0, 10, and 45 cm) in mangrove sites along the San Bernardo and El Pedregal estuaries during two climatic seasons. Values followed by different letters at each season are significant among sites (P < 0.05).

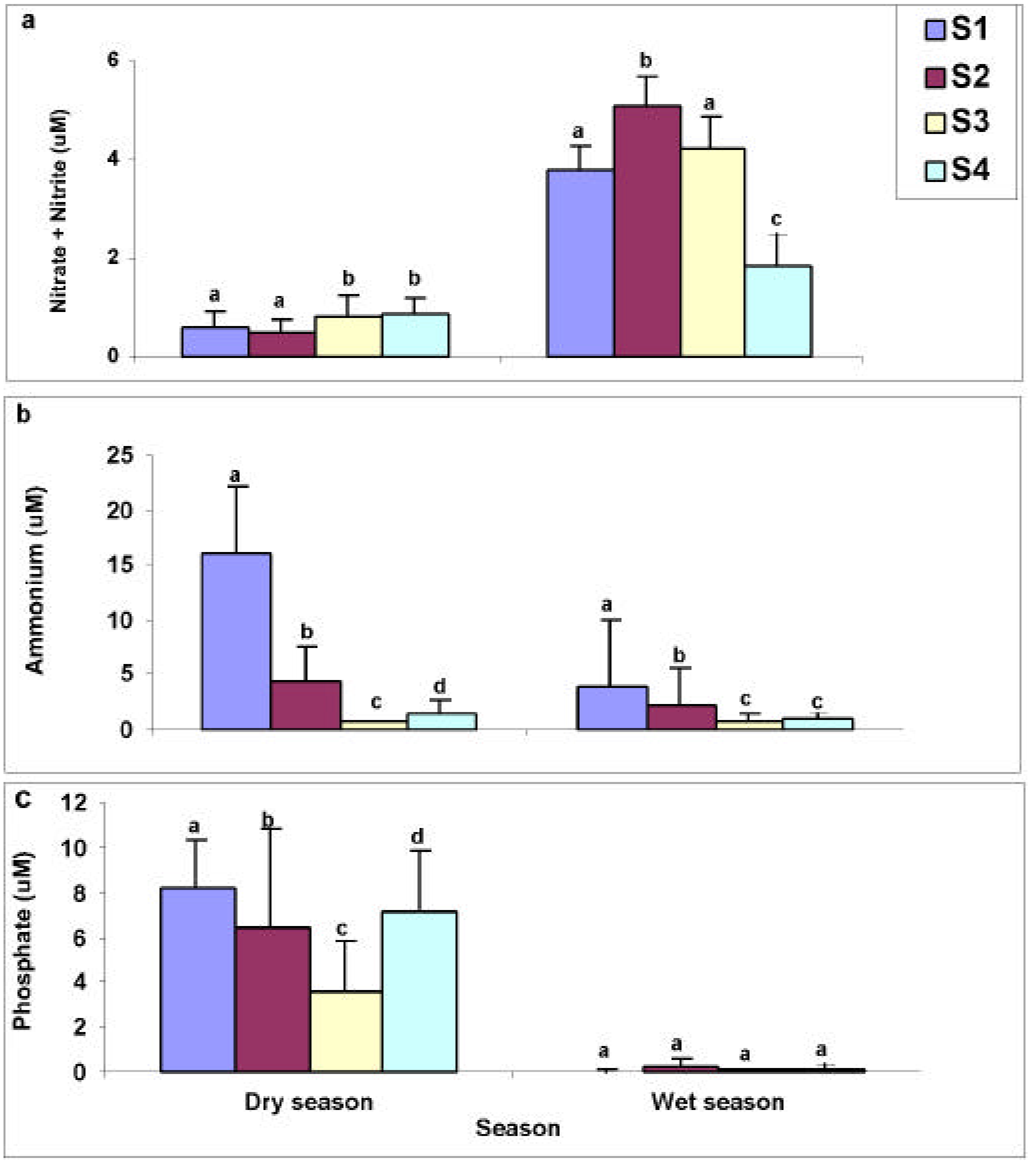


Fig. 19. Mean values of porewater inorganic nutrients including (a) nitrate + nitrite, (b) ammonium, and (c) phosphate measured in mangrove sites during February and August 2001 along the San Bernardo and El Pedregal estuaries. Values followed by different letters at each season are significant among sites ($P < 0.05$).

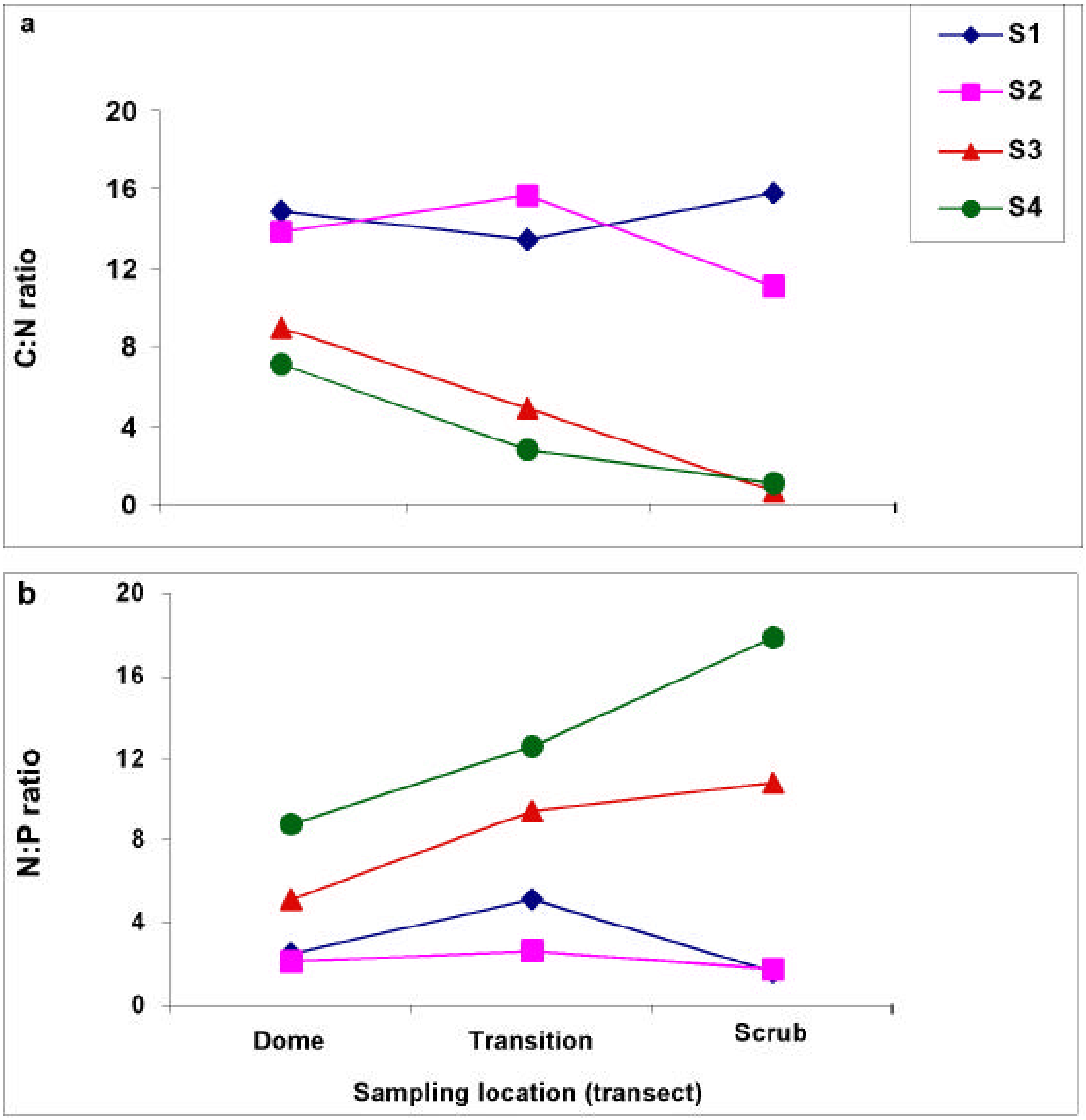


Fig. 20. Changes of soil carbon, nitrogen and phosphorus expressed as C:N (a) and N:P (b) ratios in the top 20 cm of soils along the elevation gradient in mangrove sites.

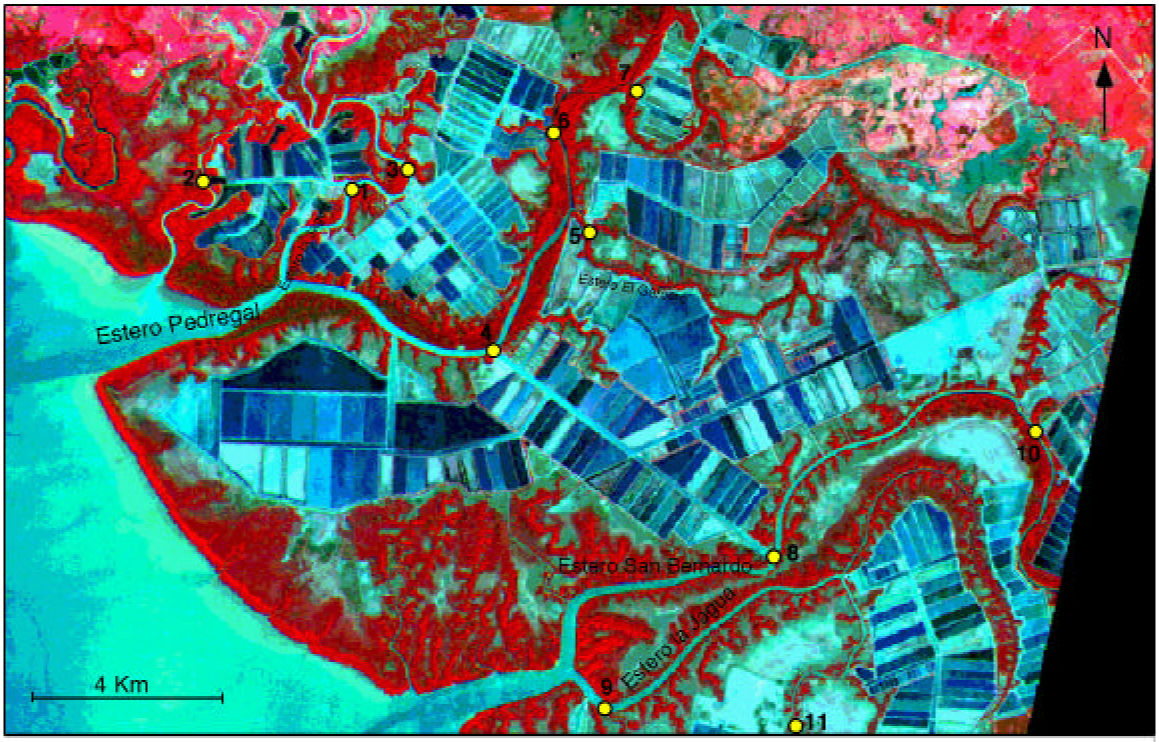


Fig. 21. Location of water quality sampling stations. (1) Aquacultivos 1; (2) Aquacultivos 2; (3) La Jagua; 4) GMSB 1; (5) El Garcero; (6) Cadelpa; (7) Promasur; (8) GMSB 2; (9) Biomar; (10) Cumar; (11) Crimasa.

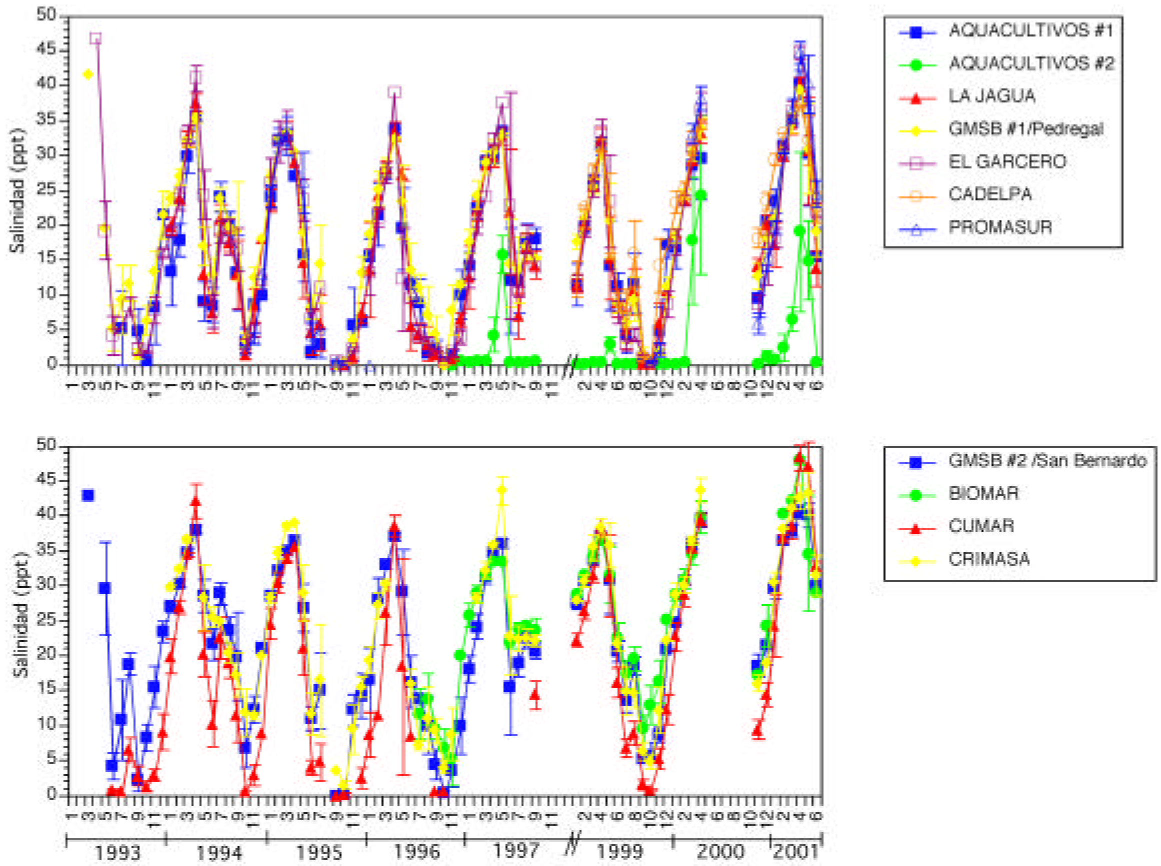


Fig. 22. Salinity values (mean ± SE) in stations located north and south of Punta Guatales (1993-2001). No data is available for 1998. Hurricane Mitch indirectly impacted the region in October-November 1998.

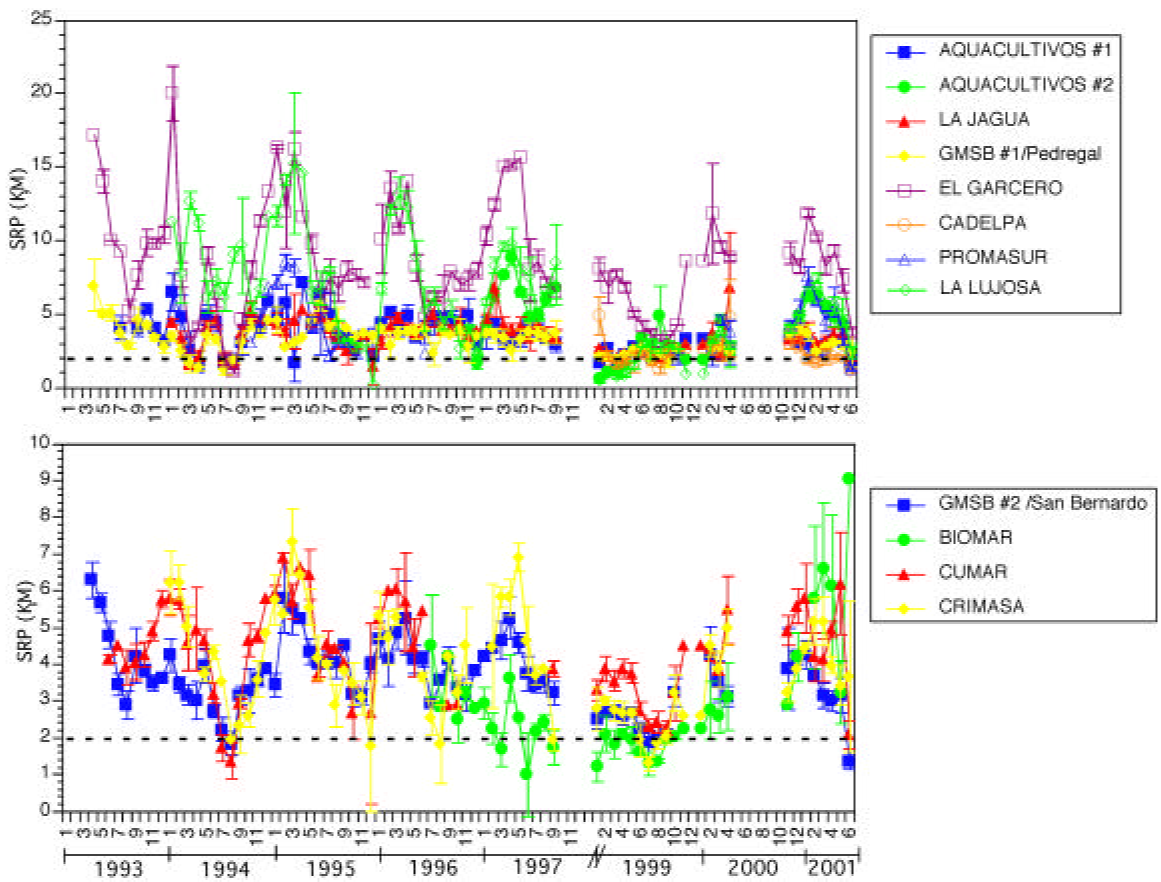


Fig. 23. Soluble reactive phosporous (SRP, µM) values (mean ± SE) in stations located north and south of Punta Guatales (1993-2001). No data is available for 1998. Hurricane Mitch indirectly impacted the region in October-November 1998.The dotted line indicates the mean value generally found in coastal ecosystems.

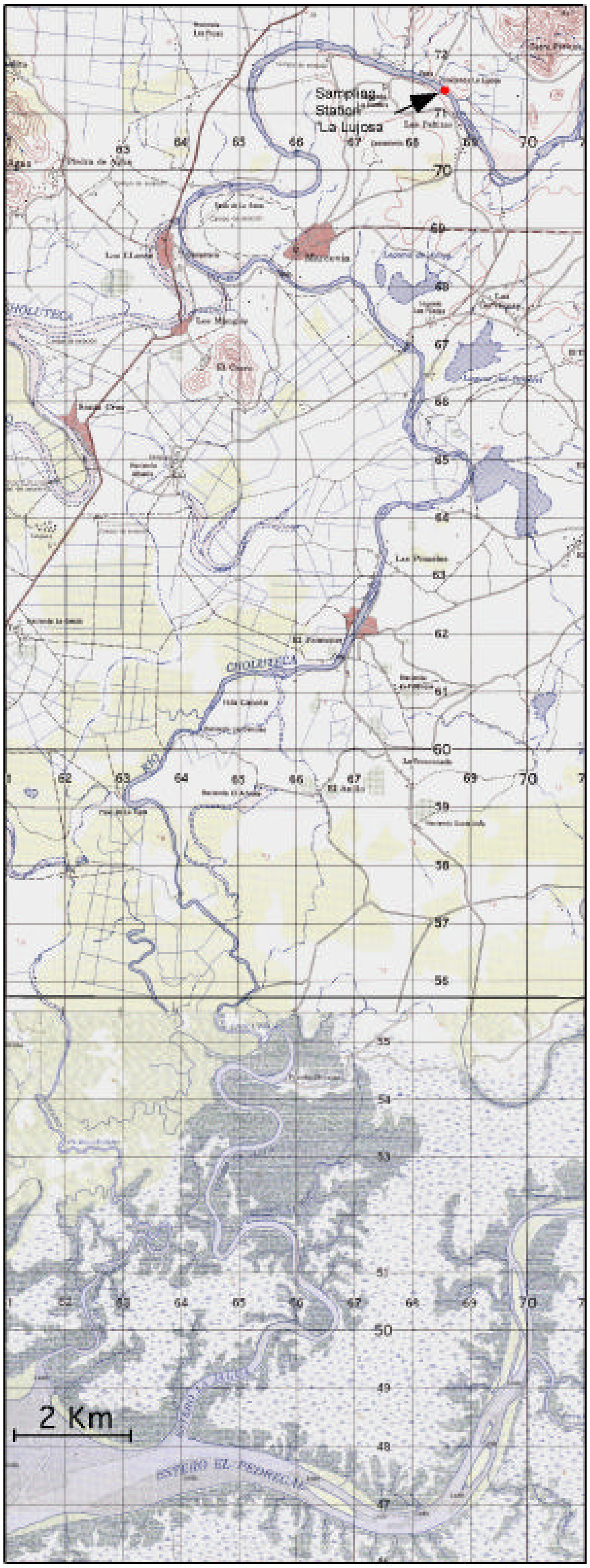


Fig. 24. Location of La Lujosa sampling station in the Choluteca River.

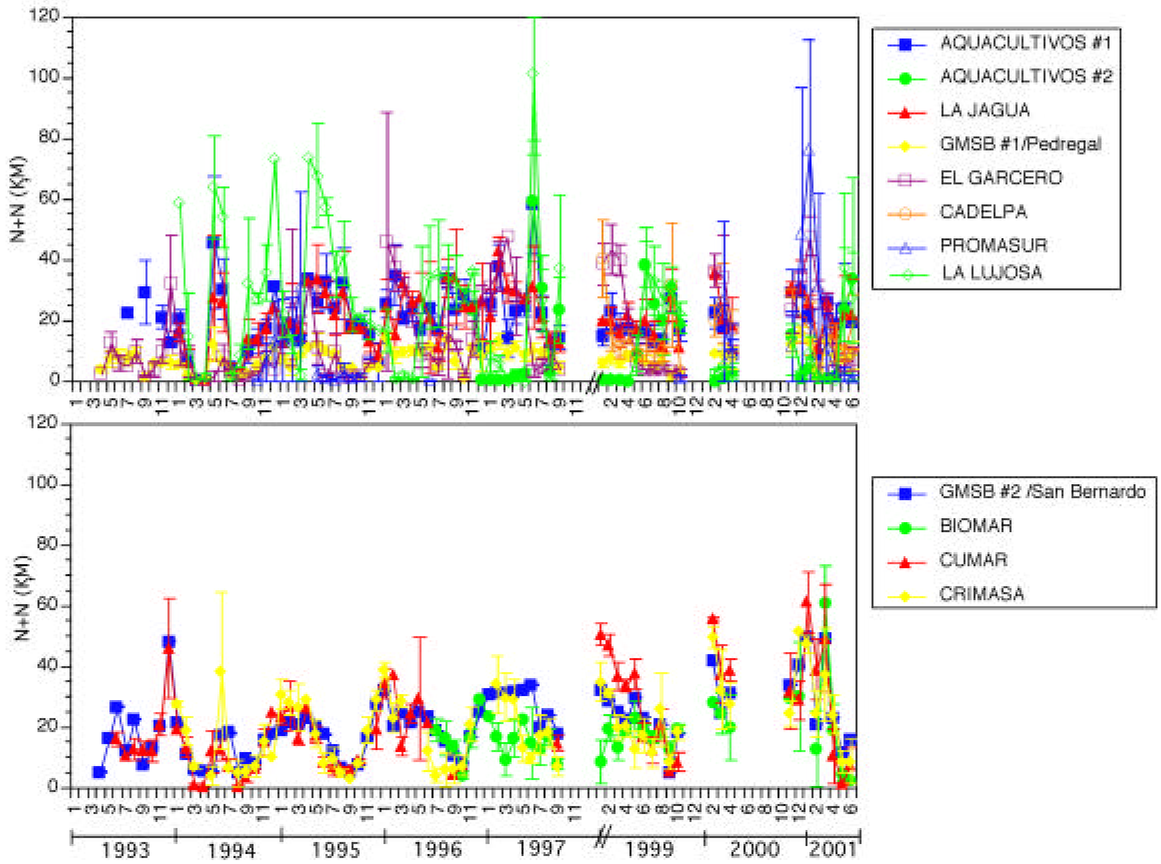


Fig. 25. Nitrate plus Nitrate (N + N, µM) values (mean ± SE) in stations located north and south of Punta Guatales (1993-2001). No data is available for 1998. Hurricane Mitch indirectly impacted the region in October-November 1998.

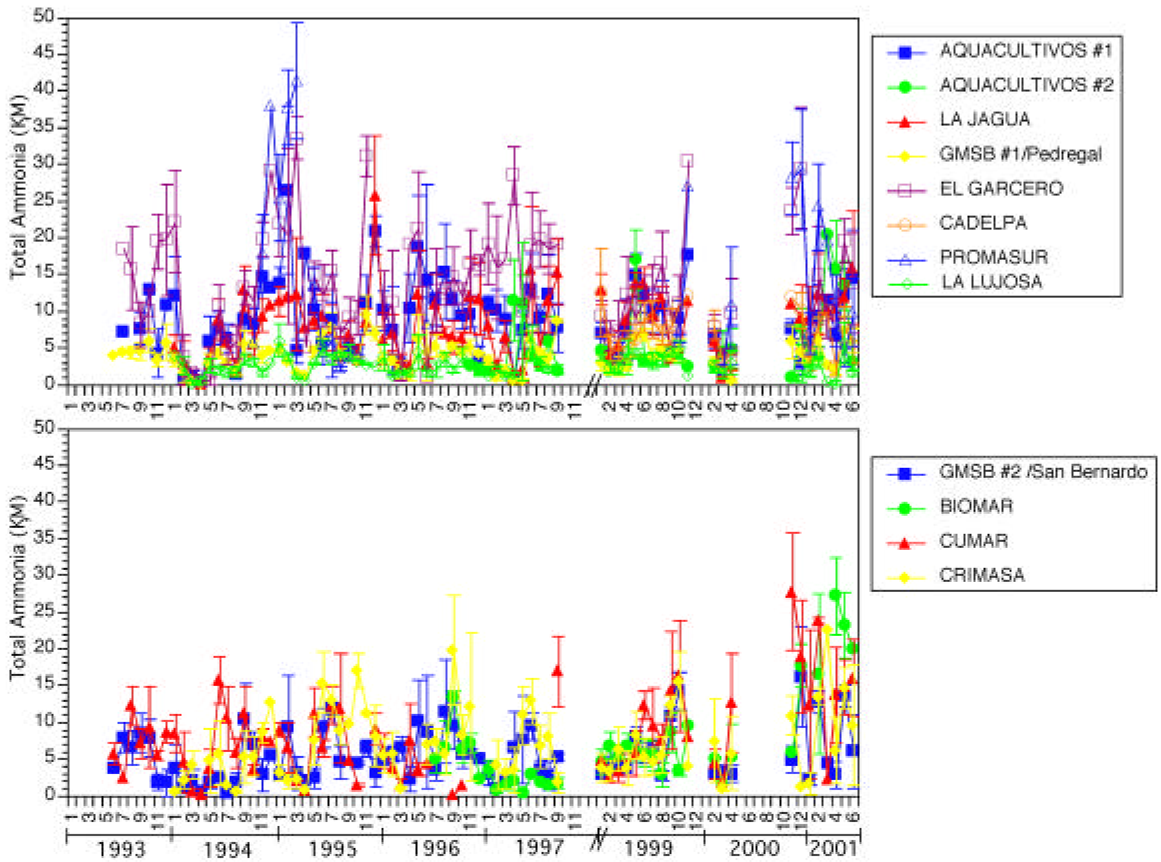


Fig. 26 Total Ammonia (TA) values (mean ± SE) in stations located north and south of Punta Guatales (1993-2001). No data is available for 1998. Hurricane Mitch indirectly impacted the region in October-November 1998.

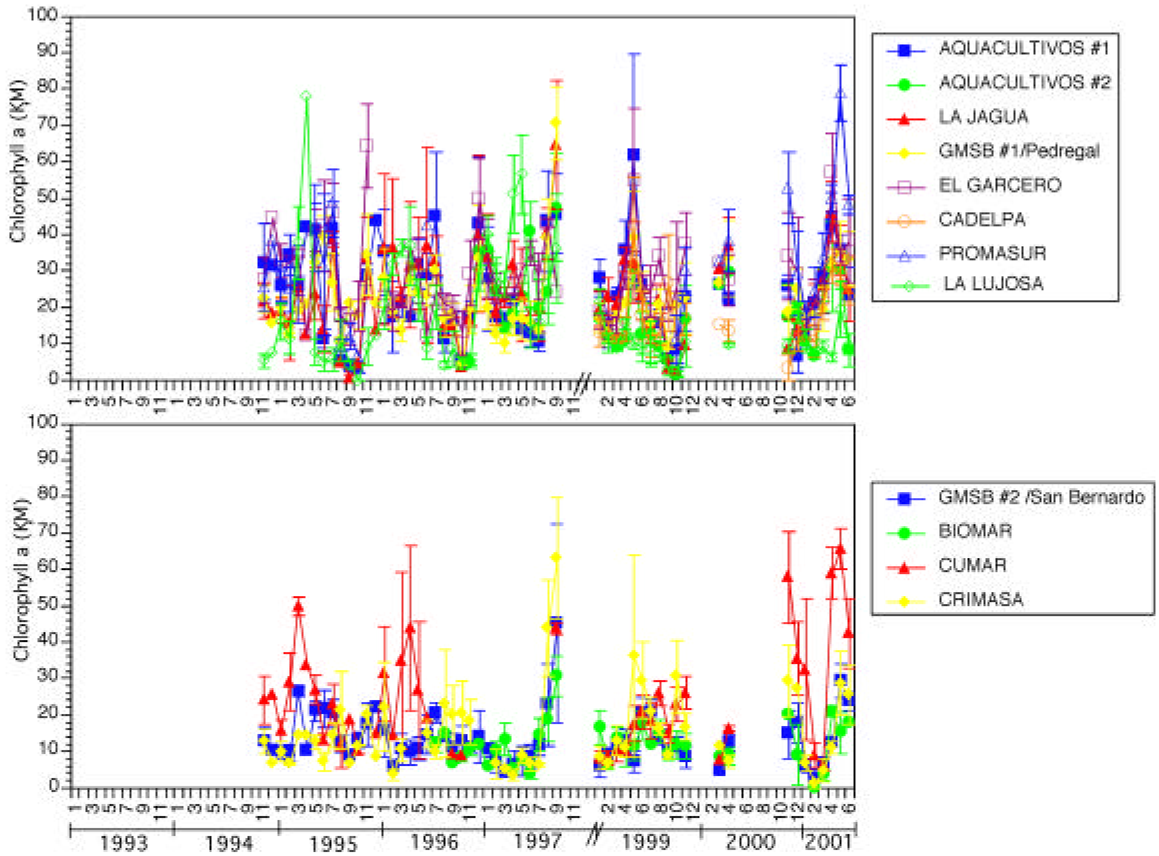


Fig. 27. Chlorophyll a values (mean ± SE) in stations located north and south of Punta Guatales (1993-2001). No data is available for 1998. Hurricane Mitch indirectly impacted the region in October-November 1998.

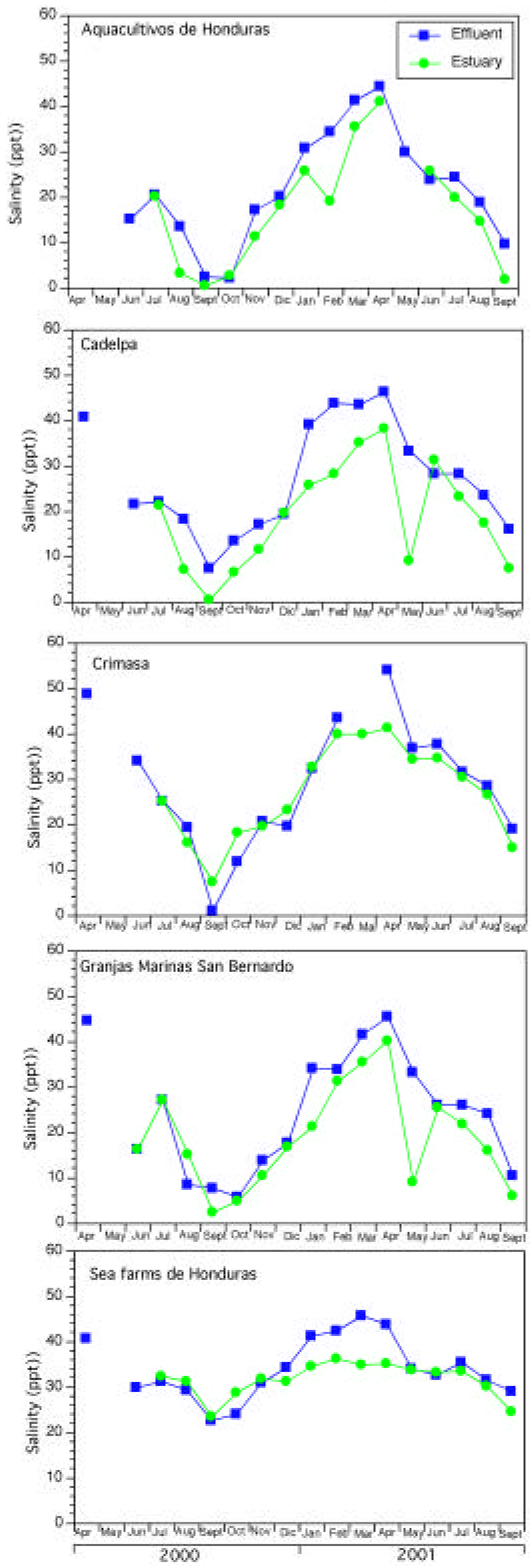


Fig. 28. Salinity values in shrimp pond effluents and estuaries in Punta Guatales, Gulf of Fonseca, Honduras.

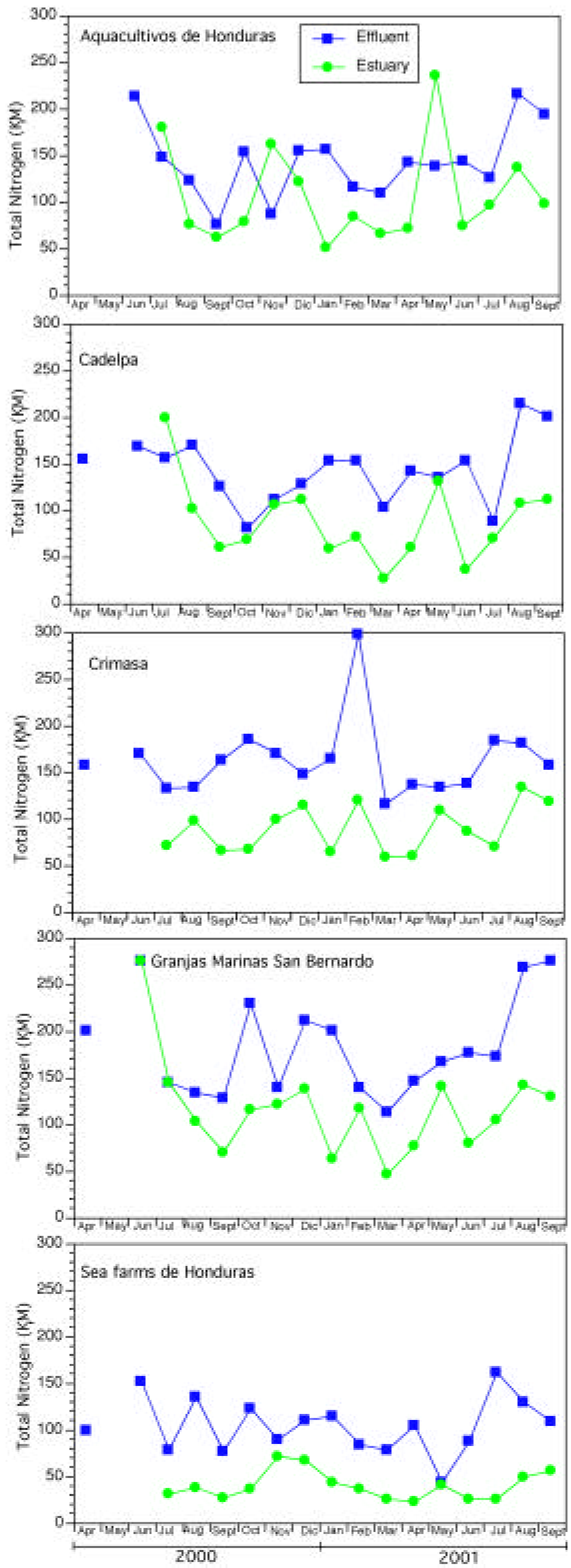


Fig. 29. Total nitrogen concentrations in shrimp pond effluents and estuaries in Punta Guatales, Gulf of Fonseca, Honduras.

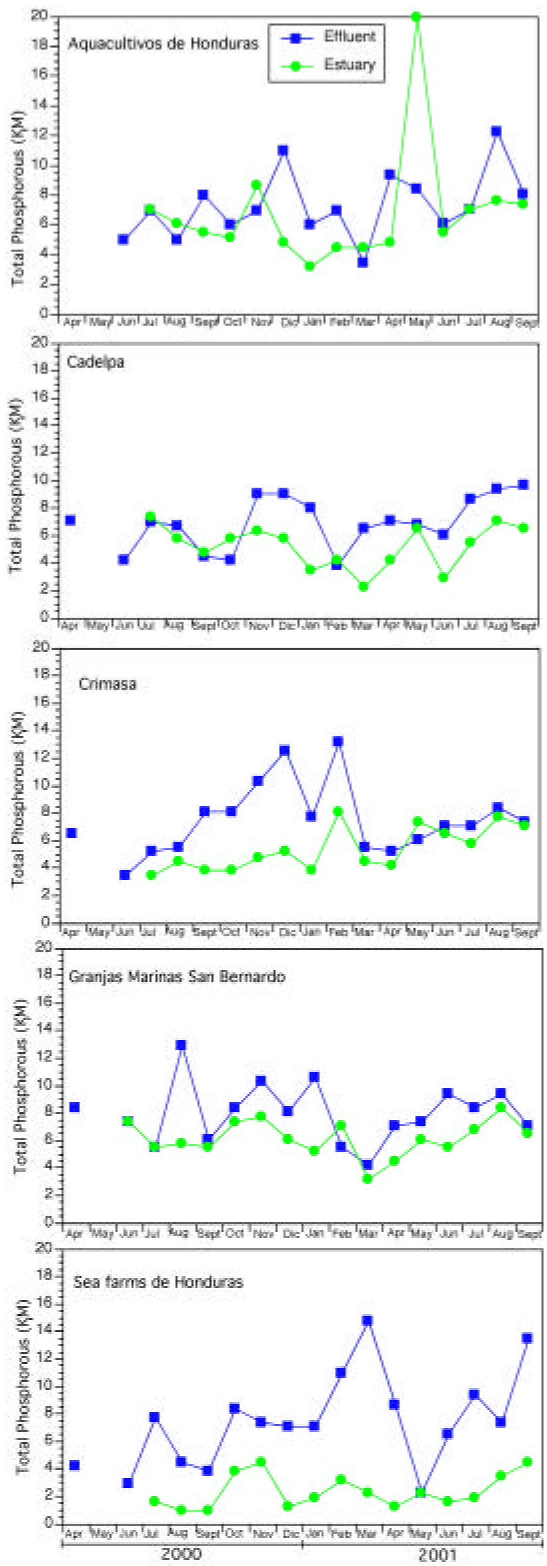


Fig. 30. Total phosphorus concentrations in shrimp pond effluents and estuaries in Punta Guatales, Gulf of Fonseca, Honduras.

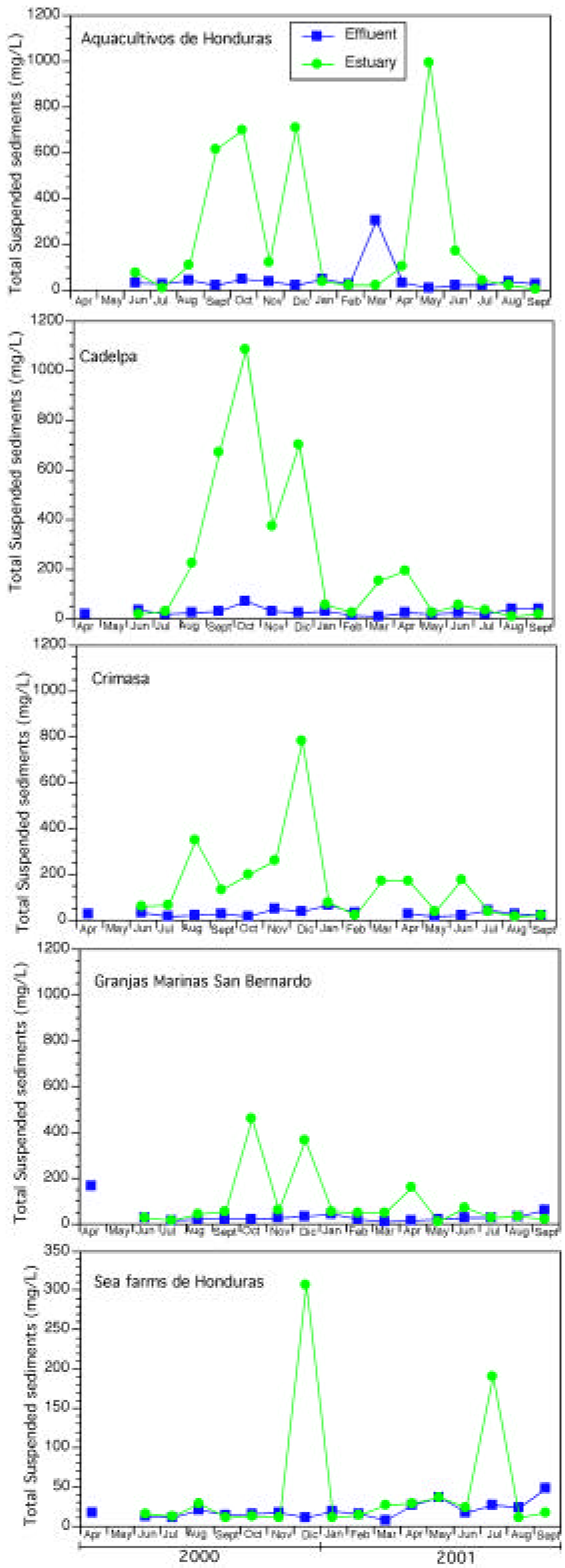


Fig. 31. Total suspended sediments concentrations in shrimp pond effluents and estuaries in Punta Guatales, Gulf of Fonseca, Honduras.

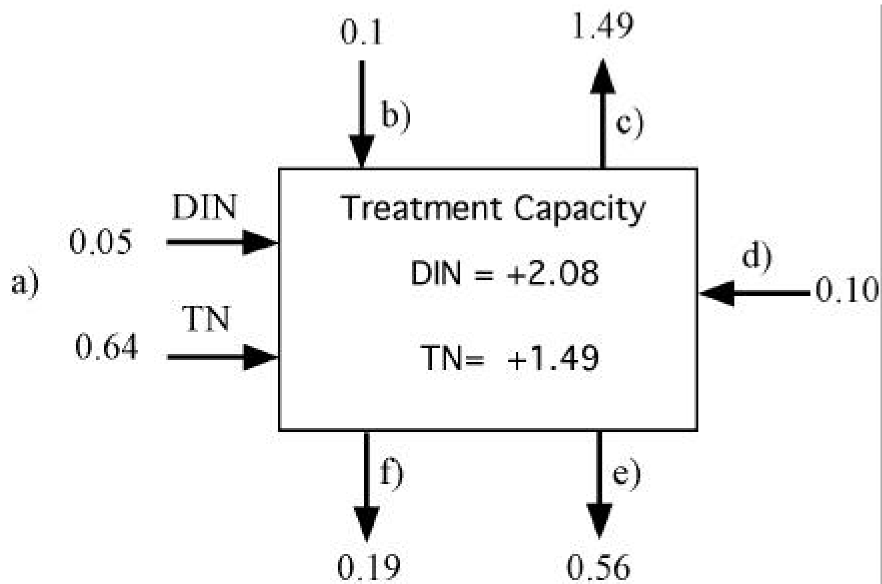


Fig. 32. Estimated potential nitrogen treatment capacity (kg ha^{-1} d^{-1}) of mangrove forest receiving effluents from shrimp aquaculture ponds in Honduras. DIN= dissolved inorganic nitrogen; TN= total nitrogen;. (a) pond effluent; (b) nitrogen fixation; (c) denitrification; (d) tidal inundation; (e) plant uptake; and (f) N accumulation in soil. Nitrogen fluxes are from Rivera-Monroy and others (1999) except pond effluents estimates (Valderrama, unpublished results).

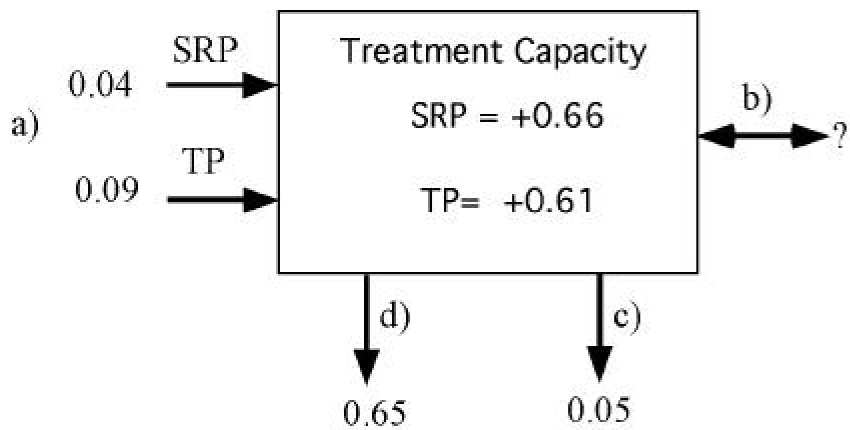


Fig. 33. Estimated potential phosphorous treatment capacity (kg ha^{-1} d^{-1}) of mangrove forests receiving effluents from shrimp aquaculture ponds in Honduras. SRP= soluble reactive phosphorus; TP= total phosphorus.; (a) pond effluent (Valderrama, umpublished results), (b) tidal inundation, (c) plant uptake (Robertson and Phillips, 1995); and (d) P accumulation in soil (Lynch, 1989).

Fig. 34. Tree of *Laguncularia racemosa* growing along Canal Norte, Granjas Marinas San Bernardo. On the forefront there is a sapling of the same species (October 2000).

Fig. 35. Mangrove trees along Canal Norte, Granjas Marinas San Bernardo. Honduras. This vegetation colonized the channel after 1984. Tree height is > 8 m (October 2000).

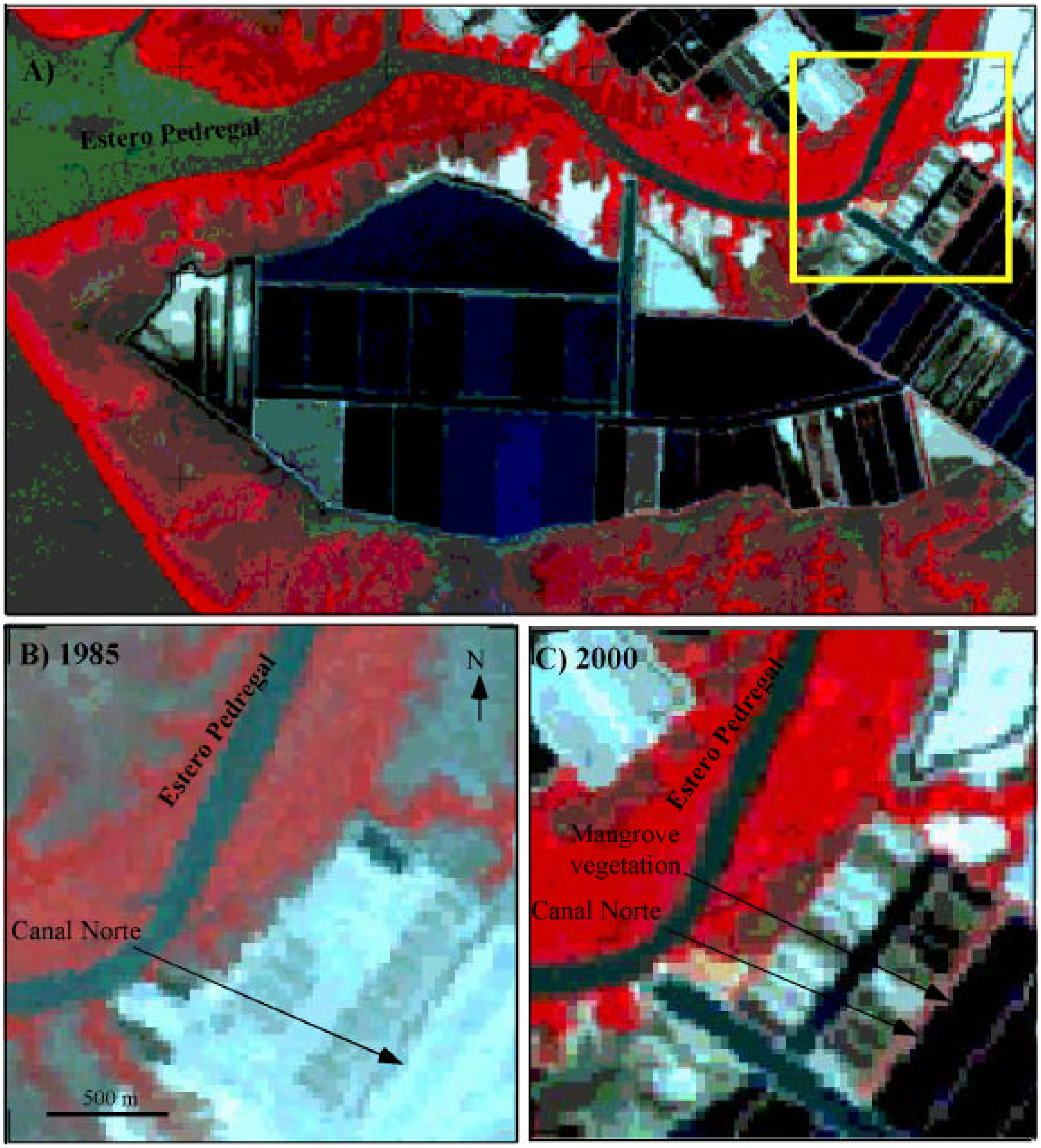


Fig. 36. Satellite images (Landsat) of Granjas Marinas San Bernardo, Honduras, and location of Canal del Norte (A). Frames B and C show landscape details of Canal Norte in 1985 and 2000. The arrow in 2000 shows new mangrove vegetation cover (red) along the channel.

Fig. 37. *Laguncularia racemosa* and *Avicennia germinans* saplings at the edge of the oxidation pond located next to El Pedregal estuary (February 2001).

TABLES

TABLES

Table 1. Information summary of Landsat Thematic Mapper (TM) data sets used to estimate mangrove a shrimp pond areas in the Gulf of Fonseca.

LABEL	Product number	WRS	Acquisition _date/time	Satellite	Satellite _instrument	Sun elevation	Sun_azimuth
LT5017050009305410	1199030102910010	017/050	022393/15273279	LANDSAT_5	TM	46.18	121.77
LT5017051009305410	1198110700420000	017/051	022393/15275659	LANDSAT_5	TM	46.74	120.32
LT5018051009406410	1199033000850000	018/051	030594/15325541	LANDSAT_5	TM	48.66	115.66
LT5018050009406410	1199030102910010	018/050	030594/15323174	LANDSAT_5	TM	48.22	117.26
LT5017050009905510	11010104008600000	017/050	1999-02-24T15:44:49Z	LANDSAT_5	TM	49.22	123.82
LT5017050009915110	11010104008600000	017/050	1999-05-31T15:43:42Z	LANDSAT_5	TM	60.87	70.59
LT5018051009931810	11010104008600000	018/051	1999-11-14T15:47:52Z	LANDSAT_5	TM	48.9	138.69
LE7017051009931950	11010109008700000	017/051	1999-11-15T15:58:52Z	LANDSAT_7	ETM+	51.27	143.72
LE7018050000008950	11010109008700000	018/050	2000-03-29T16:04:17Z	LANDSAT_7	ETM+	60.54	108.58
LE7017051000009850	11010109008700000	017/051	2000-04-07T15:58:23Z	LANDSAT_7	ETM+	62.39	99.48

Table 2. Mangrove and shrimp pond total surface (ha) in the coastal zone of the Gulf of Fonseca, Honduras.

	Mangrove surface (ha)			Ponds	
Year	Total	> 3 m	= 3 m	Number	Surface
1985	42,215	8,308	33,907	58	845
1994	37,788	10,560	27,228	691	10,040
1999	35,375	4,169	31,206	1,022	15,580

Table 3. ID values for classes selected to determine mangrove surface in the Gulf of Fonseca, Honduras.

Year	> 3 metros	= 3 metros
1985	144, 145, 151, 180,181	72, 73, 78, 79, 108, 109, 114, 115,
1994	144, 145, 151, 180, 181, 187	72, 73, 79, 80, 108, 109, 115
1999	144. 145, 150, 151, 152, 180	72, 73, 78, 79, 80, 108, 109, 114, 115, 116

Table 4. Mangrove and shrimp pond total surface (ha) in Granjas Marinas San Bernardo, Punta Guatales, Honduras (1954-2000).

	Mangrove			Ponds		Other uses		Channels
AÑO	TOTAL	> 3 m	= 3 m	Number	Surface	Number	Surface	Surface
1954	3,673			0	0	0	0	0
1985	3,946	653	3,293	13	298	8	1,598	207
1993	3,416	494	2,922	76	1,442	7	152	866
1994	3,498	787	2,711	76	1,618	13	1,256	948
1997	3,144	758	2,386	68	1,577	24	623	1,218
1999	3,083	635	2,448	99	2,823	12	735	1,645
2000	4,034	853	3,181	103	2,965	13	789	2,317

Table 5. ID values for classes selected to determine mangrove surface in Punta Guatales, Gulf of Fonseca, Honduras.

Year	> 3 m	= 3 m
1985	144, 145, 151,180,181	72, 73, 78, 79, 108, 109, 114, 115
1993	144, 145, 180	72, 73, 78, 79, 80, 108, 109, 115
1994	144, 145, 180, 181	72, 73, 79, 80, 108, 109, 115,
1997	144, 145, 180, 181	36, 72, 73, 79, 80, 108, 109, 115, 116, 151, 152, 187, 188
1999	144, 145, 150, 180	72, 73, 78, 79, 80, 108, 109, 114, 115
2000	144, 145, 151, 180	72, 73, 78, 79, 80, 108, 109, 115

Table 6. Summary of mangrove surface (ha) estimates for the Gulf of Fonseca, Honduras. Modified from Sánchez (1998) and Vergne and others (1993). INA= information no available in sources .

Year	Source	Method	Scale	Total (ha)
1954	Prats (1958)	Aerial photos (1954)	1:64,000	28,000
1962	FAO (1965)	Aerial photos (1962)	1:40,000	91,800
1973	Vergne et al. (1993)	Aerial photos (1973)	(INA)	30,697
1977	Rollet (1986)	(INA)	(INA)	32,000
1985	This study	Landsat 85	1:100,000	**42,215**
1987	Oyuela (1997)	No available	(NA)	46,710
1989	Sanchez (1998)	Landsat 89	1:100,000	46,890
1992	This study	Landsat 94	1:100,000	**37,788**
1995	Oyuela (1997)	(INA)	(INA)	41,320
1995	AFE-COHDFOR	Landsat 93 ?	1:100,000	47,200
1995	Sanchez (1998))	Landsat 95	1:100,000	41,900
1999	This study	Landsat 99	1:100,000	**35,375**

Table 7. Summary of forest structural characteristics at four sites (S1, S2, S3, and S4) located along the San Bernardo and El Pedregal estuaries in the southern region of the Gulf of Fonseca, Honduras. The importance value (I_v) for each species was calculated as the sum of relative density, relative dominance, and relative frequency.

	S1	S2	S3	S4
Mean Basal area (m^2 ha^{-1})				
A. germinans	7.0	22.6	11.6	13.3
R. mangle	16.4	0.0	10.8	23.4
	23.4	22.6	22.4	36.7
Tree density (stems/ ha)				
A. germinans	851.2	2395.1	1818.4	1064.0
R. mangle	655.8	0.0	762.5	731.5
	1507	2395	2581	1795
Relative density (%)				
A. germinans	56.5	100.0	70.5	59.3
R. mangle	43.5	0.0	29.5	40.7
Relative dominance (%)				
A. germinans	30.0	100.0	51.8	36.2
R. mangle	70.0	0.0	48.2	63.8
Relative frequency (%)				
A. germinans	58.8	100.0	69.2	63.2
R. mangle	41.2	0.0	30.8	36.8
Importance value (I_V):				
A. germinans	145	300	192	158
R. mangle	155	0	108	142

Table 8. Importance values (I_v) for mangrove species at three vegetation zones along the elevation gradient in mangrove sites along the San Bernardo and El Pedregal estuaries.

	Site	Dome	Transition	Dwarf mangroves
A. germinans	S1	0	216	300
	S2	300	300	300
	S3	0	296	300
	S4	0	222	300
R. mangle	S1	300	84	0
	S2	0	0	0
	S3	300	4	0
	S4	300	78	0

Table 9. Soil carbon (C), nitrogen (N), and phosphorus (P) concentrations along the elevation gradient at mangrove sites at the San Bernardo and El Pedregal estuaries. Values are the mean ± 1SE (n = 32). Values followed by the same letter in each row are not significant ($P > 0.05$).

	Dome	**Transition**	**Dwarf mangroves**
Total C (mg g^{-1})	22.3 ± 1.7a	19.6 ± 1.6b	9.8 ± 1.6c
Total N (mg g^{-1})	2.5 ± 0.4a	3.9 ± 0.6b	3.7 ± 0.8b
Total P (mg g^{-1})	0.6 ± 0.2a	0.5 ± 0.2b	0.5 ± 0.2c

Table 10. Annual and seasonal mean concentration and range of dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) in several neotropical estuaries. Modified from Pennock and others 1999)

Location		DIN (µM)	SRP (µM)
Florida Bay, USA			
	Eastern region	3.81 (0.4-92.1)	0.03 (0.01-0.51)
	Central region	5.56 (0.02-125.7)	0.04 (0.01-0.84)
	Western region	0.24 (0.02-14.5)	0.03 (0.01-0.39)
Apalachicola Bay, USA		9.0 (1.3-43.0)	0.16 (0.1-0.62)
Mobile Bay, USA			
	Northern region	9.8 (0.0-57.0)	0.58(0.05-1.4)
	Central region	6.4 (0.0-32)	0.46 (0.05-1.7)
	Southern region	5.1 (0.0-30.0)	0.38 (0.05-1.6)
Corpus Christi Bay, USA		7.25 (2-30.3)	1.99 (0.6-4.6)
Perdido Bay, USA			0.32 (0.16-0.48)
Elevenmile Creek (station P22)/Perdido Bay			3.87 (0.32-7.74)
Celestun, Mexico		12.72 (1.9-91.5)	0.82 (0.02-7.2)
Chelem, Mexico		9.2 (2-44)	0.41 (0.1-6.0)
Dzilam, Mexico		10.7 (2-25)	1.45 (0.2-8.1)
Rio Lagartos, Mexico		9.1 (2.0-2.21)	1.55 (0.3-11.0)
Laguna de Terminos, Mexico		4.5 (6.51-27.2)	0.2 (0.01-0.64)
Cienaga Grande de Santa Marta, Colombia			
	1993	4.82 (0.02-56.39)	0.73 (0.0-5.8)
	1994	18.53 (0.8-67.73)	1.78 (0.1-13.8)
	1995	6.03 (0.02-80.48)	1.46 (0.00-11.91)
	1996*	1.94 (0.01-20.23)	1.45 (0.00-10.83)
	1997**	3.0 (0.10-16.00)	24.0***(20.00-32.00)
	1998	3.72 (0.03-21.33)	0.38 (0.10-0.93)
	1999	6.78 (0.20-25.13)	0.78 (0.00-7.26)
Guayas River Estuary, Ecuador			
	Dry season	16.98 (0.0-54.92)	2.65 (0.78-5.15)
	Rainy season	18.26 (4.28-35.50)	2.13 (0.69-3.79)
Gulf of Guayaquil, Ecuador			
	Dry season	8.26 (3.77-18.30)	1.71 (1.08-2.32)
	Rainy season	3.29 (1.72-5.43)	1.36 (0.61-1.82)
	Grand mean (± SE)	7.9 (1.02)	1.06 (0.19)
	Median	6.78	0.78

(= 4 months; **=includes October 1996 and March, June, and September 1997;***=value was excluded to calculate mean value)*

Table 11. Mean annual chlorophyll *a* values in different coastal ecosystems (from Rivera-Monroy and others 2001).

Location	Latitud	Chlorophyll *a* ($\mu g\ L^{-1}$)
Port Hacking Basin	-34	3.50
Rio de la Plata Estuary, Argentina	-34	5.00
Tweed	-28	3.5
Brunswick	-28	12.5
Richmond	-28	19
Clarence	-29	11.35
Bellinger	-30	5.15
Nambucca	-30	4
Hastings	-31	5.5
Macleay	-30	5.5
Manning	-32	5.55
Cananeia, Brazil	-25	7
Guayas River Estuary, Ecuador	-2	10.00
Orinoco River Mouth, Venezuela	9	1.00
Cochin Backwater	10	8.00
Cienaga Grande, Colombia	11	110.00
Cienaga Grande, Colombia (Promedio, valores minimos)	11	50.00
Porto Novo Estuary, Brazil	11	11.50
Chantuto-Panzacola	15	21.50
Goa	15	8.75
Laguna de Terminos, México	18	4.10
Celestum, México	21	5.8
Chelem, México	21	2.8
Dzilam, México	21	2.7
Río Lagartos, México	21	4.9
Tai Shui Hang Stream	22	30.14
Shing Mun River	22	51.18
Lam Tsuen River	22	38.51
Tai Po River	22	30.44
Florida Bay, USA	25	8.90
Estero La Cruz	28	3.53
Barataria Bay, USA	29	12.50
Fourleague Bay, USA	29	19.00
Waccasassa River	29	4.40
Apalachicola Bay, USA	30	7.00
Altamaha River Mouth	31	3.75
Funka Bay	33	0.50
Beaufort Sound	34	6.25
Lower Pamlico River,USA	35	21.00
Pamlico River Estuary, USA	35	37.50

Chincoteague Bay	37	24.00
San Francisco Bay, USA	37	8.75
Mid-Chesapeake Bay, USA	38	8.00
Mid-Patuxent River, USA	39	22.50

Tabla 11. Cont.

Location	Latitude	Chlorophyll *a* (μg L^{-1})
Upper Chesapeake Bay, USA	39	16.00
Upper Patuxent River, USA	39	23.00
Hudson River, USA	40	3.00
Raritan Bay, USA	40	23.50
Long Island Sound, USA	41	7.50
Narragansett Bay, USA	41	8.00
Peconic Bay	41	3.50
Southern Long Island estuaries	41	5.50
Vostok Bay	42	3.50
St. Margarets Bay	44	2.00
Venice Lagoon	45	3.70
Bedford Basin	46	3.40
Columbia River	46	9.00
Boughton	46	1.40
Brudenell	46	1.95
Cardigan	46	1.55
Darnley Basin	46	1.37
Dunk	46	5.92
Foxley	46	2.03
Grand	46	1.45
Murray River	46	1.30
Mill River	46	3.85
North Lake	46	4.65
Percival	46	1.10
Rustico	46	1.99
St. Peters	46	1.71
Savage	46	1.57
Wilmot	46	3.48
Duwamish River	47	9.50
Burrard Inlet	49	8.60
St. Lawrence River	49	2.50
Strait of Georgia	49	2.10
Victoria Harbor	49	3.15
Roskeeda Bay	53	1.30
Western Wadden Sea	53	12.00
Fraser River	56	1.65
Loch Awe	56	5.80

Table 12. Annual estimates of total and net discharge quantities of selected compounds from shrimp farms in Honduras. Daily water exchange rate is 5%. (Valderrama, unpublished results).

Farm-size scenario	Variable (kg/day)								Average discharge volume (m^3 water/day)
	TN		TP		SRP		BOD		
	Total discharge[1]	Net discharge[2]	Total discharge	Net discharge	Total discharge	Net discharge	Total discharge	Net discharge	
Small farms (= 73 ha)	48.08	21.29	7.32	1.47	3.58	0.13	277.42	74.18	30,795
Medium farms (= 293 ha)	187.85	92.61	27.41	6.61	12.87	0.61	1,074.72	352.16	109,479
Large farms (= 966 ha)	623.88	307.54	91.16	22.07	42.77	2.04	3,579.82	1,179.95	363,616

[1] Refers to the total load of nutrients discharging into the mangrove forests, including the initial load of intake water.
[2] Refers to the portion of total discharge attributable to shrimp pond operation, i.e., addition of feed, water exchange, pond drainage, etc.

Observations

1. No estimates of net DIN discharge are reported because shrimp ponds operate as sinks of DIN, according to Teichert-Coddington and others (2000). However, total discharge quantities can be assumed to be 4.19, 14.89, and 49.45 kg/day for the small, medium, and large farms, respectively.
2. The above estimates correspond to farms managed without fertilization. Net and total discharges of TP and SRP would be significantly higher in farms with fertilization regimes.

Table 13. Potential treatment capacity of mangrove forest estimated for the shrimp farm Granjas Marinas San Bernardo, Punta Guatales, Honduras. Total area = 2,965 ha (2000).

Nutrient	Treatment capacity (kg ha^{-1} mangrove day^{-1})	Mangrove area needed for treatment (ha)	Wetland:pond area ratio
DIN	2.08	73.0	0.02
TN	1.49	1285.2	0.43
SRP	0.66	198.9	0.07
TP	0.61	460.91	0.16