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Effects of Shrimp Farming on the Hydrography and Water Quality of El Pedregal and San Bernardo Estuaries, Gulf of Fonseca, Honduras

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Abstract

Intensive data collection and a modeling study have been underway for the past several years addressing two of the channel estuaries draining into the Gulf of Fonseca, Estero El Pedregal and Estero San Bernardo. Data have been compiled on the shrimp farm configurations, exchange rates, and effluent chemistry. Temperature, salinity, and dissolved oxygen profiles have been measured in the estuary channels during both rainy and dry seasons. Physiographic, hydrographic, and meteorological data have been obtained to supplement the estuary data. This report examines the assimilative capacity of these estuaries with respect to dissolved oxygen (DO). The oxygen demand of organics is measured by biochemical oxygen demand (BOD). Shrimp farm BOD loadings were estimated from effluent data and exchange. A transport model for salinity and DO in the estuaries was applied to predict the tidal-mean and section-mean concentrations of salinity and DO. The model predictions of DO—based on 1995 BOD loadings—were satisfactory. Future loadings based upon full shrimp farm development along these two estuaries were then input to determine the resulting DO under these conditions. It was found that the 1995 configuration is already pressing the carrying capacity of both systems, and the DO will be worse at full development. Shrimp farms placed farther upstream than about 20 km from the mouth will most likely have excessive impact on the DO in the estuary, which is exacerbated under dry-season conditions. Negative impacts of a specific farm can be ameliorated by reducing or eliminating pond discharges during the dry season and by reducing the level of water exchange employed. This work needs to be extended to address additional water quality parameters and to incorporate larger spatial scales, especially to establish the interaction between different estuaries draining into Fonseca.

Introduction

The Gulf of Fonseca is a large estuarine embayment on the Pacific coast of Central America, bordered by the countries of El Salvador, Honduras, and Nicaragua. The general geography of the Gulf of Fonseca is depicted in Figure 1, showing isobaths in meters.

The Gulf of Fonseca has become the focus of the shrimp aquaculture industry in Honduras; shrimp represents the third most important export of the country after coffee and bananas. Much of the industry, representing a total of over 14,000 ha of operational pond area in 1998, is situated along the distributaries of the deltaic region in the eastern arm of Fonseca, designated Monypenny Bay in Figure 1. (There is no geographical authority for a designation of this bay in the maps and references available to us. Rather than continuing to refer to it as “the eastern arm of the Gulf,” we have adopted the name of the historical anchorage at its mouth.) The low relief of the region, consisting of salt flats fringed by mangrove swamp, is an ideal setting for the construction of shrimp aquaculture ponds, requiring only a modest investment in levee construction. Proximity of the estuarine distributaries allows them to function as both source for exchange and make-up water and as receiving bodies for pond effluent. The high tidal range and seasonal freshwater influx (see following section) provide exchange and dilution water.

The phenomenal growth of the shrimp industry in the past decade has raised concerns that the combined effluent from the ponds might accumulate in such concentration as to contaminate the estuary waters as a source of exchange influent. Such self-limiting densities from shrimp aquaculture have been exceeded in other coastal environments where intensive shrimp aquaculture has been implemented, with disastrous consequences for the industry. Therefore, there is considerable practical need for determining whether such a limiting concentration of the industry on the Gulf of Fonseca exists and to quantify its magnitude. This is referred to here as the "carrying capacity" of the system.

As noted by Ward (1995), the concept of a carrying capacity for a source of contamination in an estuary is not new. The more common term, assimilative capacity, is the basis for determining acceptable magnitudes of wasteloads, especially from municipal or industrial sources. (In the United States, the assimilative capacity for a heavily loaded system is apportioned among the sources of waste effluent in a regulatory process called a "wasteload allocation.") Ward (1995) identifies four tasks that must be carried out to determine carrying capacity:

1) Specify which water quality parameter(s) will form the basis of water quality evaluation;
2) Determine the parameter value(s) corresponding to acceptable water quality;
3) Develop and verify a model for the specified parameter(s) that is appropriate for the estuary of concern; and
4) Establish the combination of external conditions that are critical for water quality.

While it is tempting to think of carrying capacity as a single number, like passenger capacity of a bus, it is important to recognize that it is in fact a function of position, both in terms of the region of the water-course in which acceptable water quality must be maintained and in terms of the actual locations of the sources of contamination. In a demonstrative

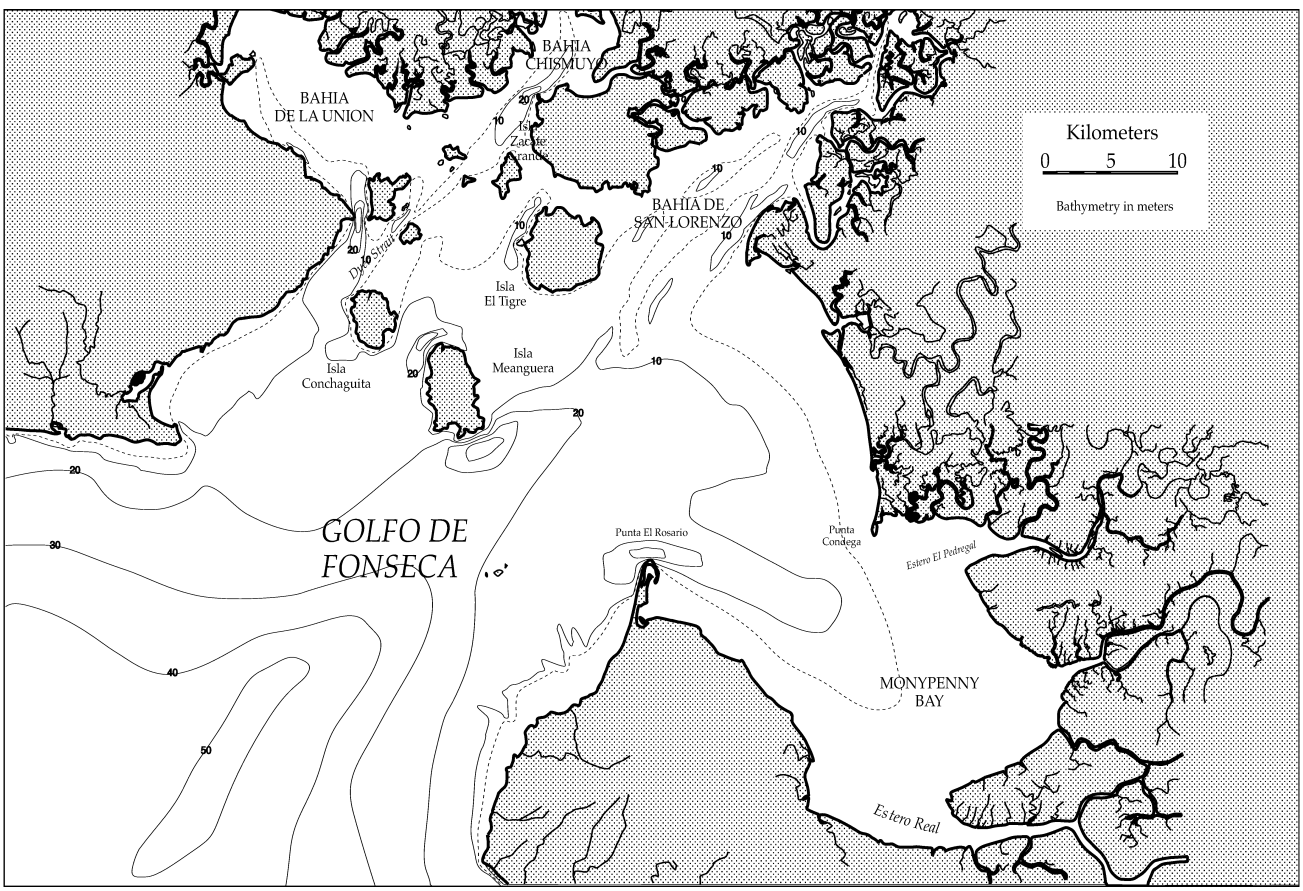

Figure 1. Gulf of Fonseca.

experiment with a water quality model, Ward (1995) showed that the intensity of the impact on water quality from a single shrimp farm could be altered by changing farm location, even though its effluent discharge remained the same. There is not a unique value for carrying capacity in terms of numbers and sizes of discharges (e.g., of shrimp farms) because different spatial distributions in combination with different effluent loads can result in the same impact on water quality. On the other hand, if the combined effect of a collection of effluent loads is to reduce water quality below its acceptable value, then it can be said that the carrying capacity of the system has been exceeded.

In this paper we address the effect of shrimp aquaculture on the single parameter of dissolved oxygen (DO) and focus on two channel estuaries, the Estero El Pedregal and the Estero San Bernardo in the Monypenny Bay region, which are the sites for the greatest development of shrimp farming operations in the country (Figure 2). While there is no single value of DO that demarcates acceptable from unacceptable water quality, as a general rule concentrations in excess of 5 mg l^{-1} assure maintenance of almost all aerobic species, and only the most resilient can survive with DO values below 2 mg l^{-1} for long periods of time. Occurrence of DO concentrations below 1 mg l^{-1} for any substantial period of time will eliminate most higher aerobic organisms.

Implicit in the above four tasks for carrying capacity determination is the availability of a database from the estuary, including water quality parameter(s) of potential concern and the factors and variables that influence those parameter(s). During the past five years data collection efforts have been underway in Honduras to quantify pond metabolism, receiving water quality, and basic hydrography of the distributaries of the Monypenny Bay region, especially Pedregal and San Bernardo. This work was accomplished under the aegis of the Auburn University/Honduras project of the Pond Dynamics/Aquaculture Collaborative Research Support Program and involved collaboration among universities, the private sector, and the public sector. A full description of the program is given in Green et al. (1997). Since 1995, annual summaries of estuarine water quality findings have been published (Teichert-Coddington, 1995; Teichert-Coddington et al., 1997; Green et al., 1998; Green et al., 1999).

Also implicit in the above tasks is the availability of a model to predict estuary response to various scenarios. Much of the purpose of this study was to develop and apply suitable models for evaluating the effects of shrimp farms on the quality of the estuaries. The management of estuaries involves being able to determine the effect on estuary circulation or constituent concentrations, or upon elements of the estuary dependent upon these (such as biological communities) that results from a specific event or external control. This general statement includes a wide range of causes and effects, both natural and man-made: discharge of wasteloads, spills of hazardous or toxic substances, floods, reductions of freshwater inflow, construction of reservoirs, shoreline development, channelization, installation of hurricane barriers, and alterations in land use in the watershed.

For some of these, especially natural events, the determination of effects can be based entirely upon data collection and analysis. Many of these, however, entail management situations in which a human activity must be evaluated before it is implemented; these require some sort of predictive capability. The standard methodology is to apply a predictive model. Models assume a variety of forms, including scaled hydraulic models, laboratory cultured ecosystems (microcosms), and statistical regressions, but in the present context we apply the term strictly to some sort of mathematical formulation of a physical relationship. The attributes that all of these types of models share are that 1) each is a simplified depiction of reality and 2) each is quantitative.

The problem of shrimp aquaculture development requires an evaluation of water quality in the area of the proposed aquaculture operation, especially how that water quality is influenced by the anticipated wasteloads from the shrimp farm itself and from other wastewater discharges in the region. Therefore, a model is needed of the space-time distribution of concentration of controlling parameters in the estuary. The concentration of a constituent is governed by transport processes (including mixing) and kinetic processes, so the model must include a determination of hydrodynamic transports as well as a mass balance of the water quality constituents. This is true whether the watercourse is a river, lake, aquifer, or estuary. For an estuary, however, the complex geometry and complicated hydrodynamics make model formulation especially difficult. For this reason, the special topic of estuary modeling has long received concentrated attention (e.g., nearly 25 years ago, Ward and Espey, 1971) and is supported by extensive literature (Ward and Montague, 1996). Also, this is why the hydrography of an estuary must be understood in order to evaluate its water quality.

Frequent reference is made to the "flushing time" of an estuary. This is defined (e.g., Officer, 1976) as:

$$T = V/Q$$

where

V = the volume of the estuary and

Q = the long-term average river inflow.

This concept is applied to lakes and rivers, but has also been adapted for use with estuaries. Flushing time, also referred to as "renewal time" and "replacement time," is the time required for the freshwater inflow to replace the volume of water in the estuary. It is directly related to the degree of dilution with "new, uncontaminated" water. Used for this purpose in water quality assessments, flushing time is a useful parameter for lakes and rivers, in which the river inflow is virtually the only supply of new water. In an estuary, however, the parameter is nearly useless, for two reasons. First, dilution varies strongly as a function of position in the estuary. A single number attempting to characterize the entire system is useful only for gross, relative comparisons between estuaries, not for any absolute characterization of the estuary's ability to assimilate wasteloads. Second, and more importantly, there are other mechanisms of dilution and water replacement operating in an estuary in addition to river inflow. Most important among these are tides, meteorological flushing, and the influx of more saline water driven by density currents. As an example, in longitudinal estuaries (river-channel estuaries), the flow associated with the density current circulation (see Ward and Montague, 1996) is nominally an order of magnitude greater than the river flow that maintains the salinity gradient. The flushing time for the Gulf of Fonseca based upon mean inflow is estimated to be on the order of five years, from which one would conclude that the gulf is too poorly flushed to sustain any intensive wasteloading, including wasteloads from shrimp aquaculture. This is clearly incorrect. Moreover, this single number gives us no information about whether a specific farm at a definite position will experience water quality problems. As shown by Ward (1995), the same farm placed in different positions within the estuary will either be well flushed and experience no problems or be so poorly flushed that it far exceeds the assimilative capacity of the estuary. This problem of the variability of assimilative capacity as a function of spatial position and external conditions is the fundamental reason for development of a model.

Hydrography of the Gulf of Fonseca

The Gulf of Fonseca is a type of estuary. There are several definitions of an estuary, of which the important feature is that an estuary is a complex watercourse that is transitional between a purely riverine system and one that is purely marine. Therefore, an estuary is governed by hydrographic processes that are both riverine and marine, e.g., floods and tides, respectively. It is also subject to processes that are unique to the estuarine environment, originating from the interaction of marine and riverine influences and its semi-enclosed morphology. Estuaries tend to be broad, well-circulated systems. There is usually a clear zonation in morphology and habitats with distance out to sea—from deep, saline, well-aerated watercourses near the main inlet to the sea to shallow, brackish, poorly flushed systems in the upper reaches.

By definition, an estuary includes a riverine inflow. Inflow affects the hydrography of the estuary by establishing a gradient of salinity across the system and further influences the water quality by its associated influx of constituents of terrestrial origin, frequently including human wasteloads. Both the magnitude and time sequence of inflow are important in the overall estuarine hydrography. Some systems have highly time-variable inflows, ranging over several orders of magnitude, while others have relatively steady inflows. When there is a prominent seasonality in the freshwater inflow, the character of the estuary can change greatly from the low-flow to the high-flow season. An extreme example is the estuaries of India (e.g., the Vellar), which shift with the monsoon from pure freshwater systems to pure seawater. It is important to recognize that the inflow feature of an estuary enlarges the geographical area of concern to encompass the entire watershed of the feeding rivers, which may entail a completely different hydroclimatology than that of the coastal region in which the estuary is located.

Morphologically, the Gulf of Fonseca is a tectono-bay (Ward and Montague, 1996), a flooded coastal indentation formed by faulting and vulcanism, and part of the Great Rift Valley, some 3,000 km^2 in area and 17 km^3 in volume (Figure 1, p. 2). The Gulf of Fonseca has a free connection with the Pacific of approximately 30 km width and 20 m average depth. The inland reaches of the Gulf of Fonseca exhibit features of a drowned-river-valley-type estuary, with extensive mud shoals and deltaic-like shoal areas, especially in its eastern arms. The overall volume of the gulf is estimated to be about 1.7 x 10^{10} m^3. Its coastal physiography consists of tidal flats, tidally flushed mangrove swamps fringing largely unvegetated tidal flats, and low relief "sweetland" punctuated by steep igneous formations. Several submerged pinnacle rocks lie in the open waters of the gulf and comprise a hazard to navigation.

As an estuary, the Gulf of Fonseca is subject to both marine and terrestrial influences, the former including tides and oceanic salinity, the latter freshwater runoff and fluvial sediment influx. It is also subject to factors unique to the coastal zone, including littoral sediment transports, density currents arising from salinity gradients, and physiographic controls on circulation (Ward, 1995; Ward and Montague, 1996).

Several major rivers drain into the gulf, especially in its eastern arm (Figure 2), where fluvial sediment deposition has created an extensive deltaic system. The primary riverine inflow to this region from Honduras is the Río Choluteca, but there are also several other major riverine drainageways into this deltaic region from Honduras and Nicaragua, most prominent of which is the Estero Real, whose watershed extends to Lago de Managua. Numerous distributaries lace this deltaic region, most carrying freshwater throughflow deriving from local runoff, but each in itself is a channel estuary.

Hydrology is dominated by the terrestrial environment. Hydrography, on the other hand, is decidedly marine. Most important is the effect of tides. The average tidal range on the Pacific coast of Central America increases from about 0.5 m around Acapulco to about 2.5 m off Panama and is about 1.8 m off

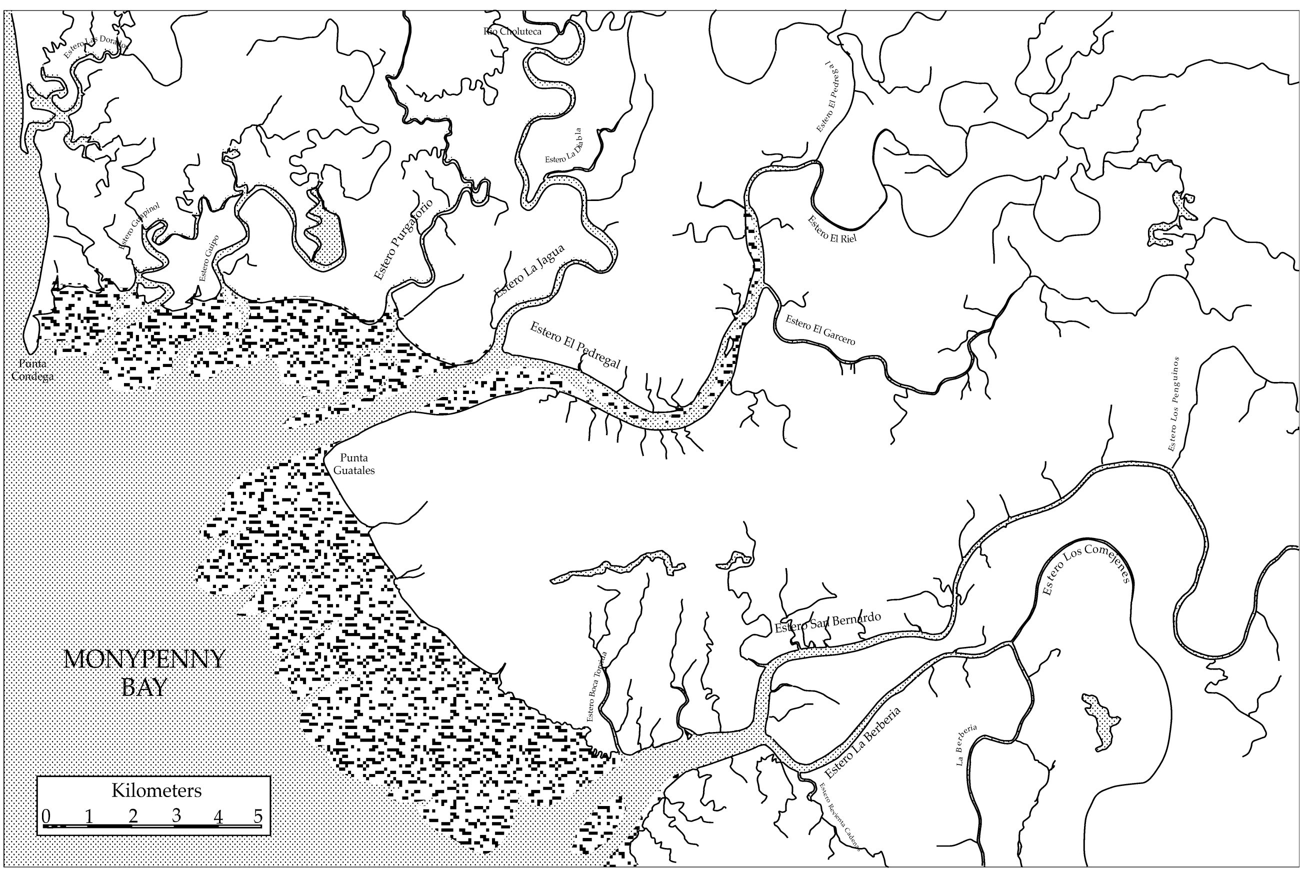

Figure 2. Study area: Esteros El Pedregal and San Bernardo.

Honduras. (Variability with the relative alignment of sun and moon and with lunar declination is more than 25% about this mean value.) The effect of the coastal bight and the morphology of the gulf is to amplify this tide within the gulf, so that the mean amplitude is over 2.5 m at La Union and over 3 m in many of the coastal inlets and channel estuaries. Moreover, this is a semidiurnal tide (period 12.4 h), so the movement of water with the flood and ebb is substantial. Tidally scoured channels are evident in the bathymetry of the gulf (Figure 1, p. 2), one adjacent to Isla Meanguera in the center of the gulf being marked in excess of 50 m and several others with depths exceeding 20 m. The great depths in Estero Real, which make it navigable by oceangoing vessels in its lower 50 km, are most likely due to tides rather than river flow; coastal pilots note tidal currents exceeding three knots (Admiralty, 1951). (A typical tide variation in Estero El Pedregal is shown in Figure 3.)

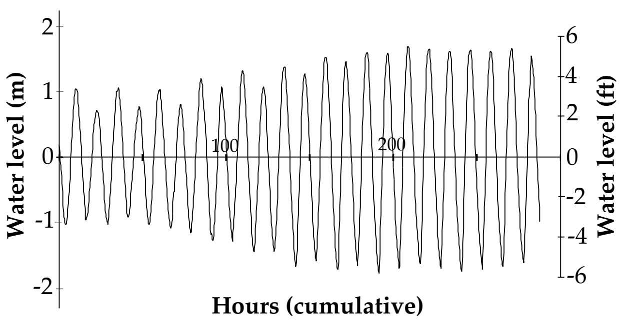

Figure 3. Observed tide variation at Granjas Marinas intake, Estero El Pedregal, 1–12 August 1994.

Offshore, this region of the Pacific is dominated by the Costa Rican Current, setting to the northwest and fed by the Equatorial Countercurrent, which is strongest when the Intertropical Convergence Zone (ITCZ) is in its most northerly position (corresponding to the rainy season) (e.g., Wyrtki, 1966). During this same season, waters with lower near-surface salinities (on the order of 30‰) are advected northward from the Gulf of Panama region. With the seasonal weakening of the Equatorial Countercurrent in February to April, these near-surface salinities rise to the order of 34‰. The region has a characteristically shallow thermocline (~50 m). Through the mixed layer, salinities increase slightly and temperatures decrease slightly down to the level of the pycnocline. At depth in the eastern tropical Pacific is the subsurface oxygen-minimum layer found in the equatorial zones of the subtropical gyres in all oceans, formed by a combination of high productivity and poor circulation. Off Central America, the minimum dissolved oxygen concentrations drop as low as 0.5 mg l^{-1} in this deep hypoxic layer of perhaps 1 km thickness, but the layer's vertical extent seems to be capped by the pycnocline, so it is unlikely that it influences the Gulf of Fonseca. The net effect of these processes is an offshore water mass of 30 to 34‰ salinity that is fairly well mixed within the range of depths of the entrance to the Gulf of Fonseca, whose waters are continuously replaced by the longshore current. By estuarine standards, this is a fairly homogeneous seaward boundary. By far, the more important sources of variability in salinity and chemical quality within the gulf are due to its internal hydrography and terrestrial influences.

Salinity is a fundamental estuary measurement for several reasons:

1) Salinity can be measured quickly and inexpensively. Moreover, there are devices for rapid measurement of salinity in the field, most important of which is the profiling conductivity meter.
2) Because the salinity of fresh water is nearly zero, the value of salinity at a point in an estuary can be interpreted as a measure of the proportion of seawater at that point.
3) Salinity is virtually conservative and therefore can be used as a water tracer.
4) In an estuary, the variation of water density is dominated by salinity and is little, if at all, affected by temperature. Therefore, salinity is an important indicator of hydrodynamic processes affected or controlled by density, such as turbulence and density currents.
5) Salinity is one of the key variables determining habitat due to the varying osmoregulatory capabilities of estuarine organisms.

In an estuary, salinity has both horizontal and vertical structure, and the distribution of sampling points must be capable of resolving both of these. This requires numerous horizontal stations, sufficient to map the principal gradients of salinity through the system, and vertical profiles to depict the stratification of salinity.

One inference from the density-current structure of an estuary is that in a system with a predominant longitudinal dimension (e.g., a coastal-plain or river-channel estuary) where the density current is directed upstream in the lower layer, any dissolved or suspended constituent that is introduced or settles into the lower layer will tend to migrate upstream. In association with vertical mixing, this leads to a considerable upstream and downstream dispersion of matter in the saline intrusion reach. It also implies that sediments subject to gravitational settling in the water column will be carried upstream with salinity intrusion. This will result in a zone of lower-level convergence of sediments in the upper reaches of the estuary and greatly enhanced siltation. This is the basic mechanism creating the turbidity maximum in an estuary (Ward and Montague, 1996).

The greatest deficiency in information about the Gulf of Fonseca concerns its salinity regimes. Given the relatively low freshwater inflows, the broad, deep physiography of the gulf, the large tidal range, and the high, stable salinities of this region of the Pacific, we judge that the main body of the gulf itself should be relatively saline, probably near oceanic values most of the time (i.e., 35 to 36‰). The role of the high evaporation rate in the area is unknown. Conceivably, this could produce salinities within the gulf in considerable excess of oceanic values. But this would be accompanied by sinking of the denser water, and the large volume of the gulf and its free exchange with the sea would suggest that substantial increases in salinity above oceanic values are unlikely, at least in the main body of the system. Of greater importance in the present context is the information that salinity could give us about circulation and exchange. The magnitude and location of salinity gradients would be a direct indicator of the influence and magnitude of peripheral runoff and the principal density currents. The time-variability in salinity would provide insight into intrusion and extrusion of seawater and the effectiveness of this as a diluting and renewal mechanism for the gulf. It would also indicate to what extent the river-channel estuaries exchange with the open gulf waters near their mouths.

Climatology and Hydrology

The climatology of Honduras meets the United Nations Educational, Scientific and Cultural Organization (UNESCO) definition of "humid tropics" and is characterized by two distinct seasons in the year, the dry season and the rainy season (Chang and Lau, 1983). These seasons are keyed to the annual movement and intensity of the ITCZ. In fact, a closer examination of precipitation and streamflow data reveals that 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. This produces a characteristic bimodal shape to the annual pattern of precipitation. Daily precipitation measurements from Marcovia and Yusgüare (which approximately span the watersheds of Esteros El Pedregal and San Bernardo) were compiled for the period 1973–96 and accumulated monthly. The corresponding annual patterns for each year are superposed in Figure 4. The summer minimum month of July should be noted. The 1973–96 average pattern, shown as the bold line in Figure 4, is clearly bimodal.

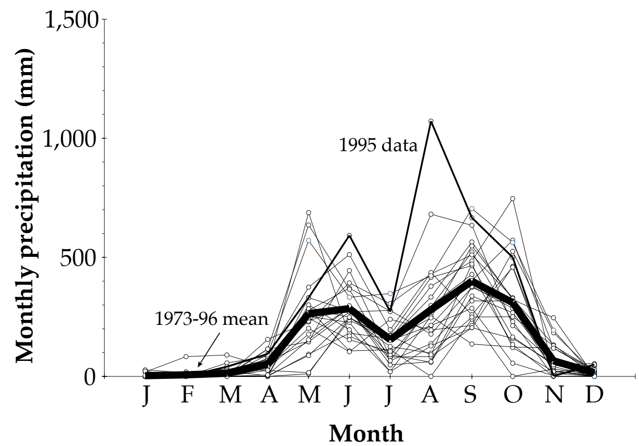

Figure 4. Average monthly precipitation at Marcovia and Yusgüare, 1973–1996.

The primary riverine inflow to this region from Honduras is the Río Choluteca, but there are also several other major riverine drainageways into this deltaic region from Honduras and Nicaragua, most prominent of which is the Estero Real, whose watershed extends to Lago de Managua. Numerous distributaries lace this deltaic region, most carrying freshwater throughflow derived from local runoff, but each in itself is a channel estuary.

The Río Negro enters the head of the San Bernardo from its watershed in Nicaragua. This appears to be a major source of inflow for San Bernardo, moreover it is evidently water of high oxygen content, originating in a mountainous basin in Nicaragua. The Río Negro, fed by the Río Gausale, Río El Gallo, and Río Los Quesos, also drains from the continental divide. The actual divide seems to be a prominent ridge, including Quiabu paralleling the Carretera Panamericana just to the west of Estili. We estimate the watershed area of the Río Negro to be about 15,000 ha.

Although the Estero Real inflow does not directly influence the channel estuaries addressed in this study, it certainly has an effect upon the Monypenny Bay area. The drainage into the Estero Real also extends to the continental divide and is the confluence of two main forks. The southern drainage includes the Río Tecomapa and its tributary the Río Olomega. The northern fork is the Río Villa Nueva, which is fed by the Río Achuapita, the Río Grande, and the Río El Portillo, extending into the mountains around Estelí.

The only one of these inflows for which we have gauged data is the Río Choluteca. Figure 5 displays the annual patterns of flow in the Río Choluteca as gauged at Choluteca. These were constructed from 1979–1990 data provided by the Departmento de Servicios Hidrológicos y Climatológicos. These data display considerable year-to-year variability in river flow, with mean annual inflow ranging from 19.2 (1986) to 90 (1980) $m^3 s^{-1}$, averaging (for the 1979–90 period) 45 $m^3 s^{-1}$. Figure 5 exhibits the average behavior of river flow, showing the 1979–90 average for each day, further smoothed by a seven-day running mean to filter out most of the hydrograph peaks. Runoff is clearly bimodal, with two high-flow seasons, spring and fall, separated by dry seasons of winter and summer, as is evident in the precipitation data. The winter dry season typically extends from November through May, during which the region becomes quite arid, exacerbated by high evaporation rates due to the high temperatures, high winds, and reduced humidities during this season. The "little dry season" of summer is usually only a two-month interruption of the thunderstorm season. Average annual evaporation

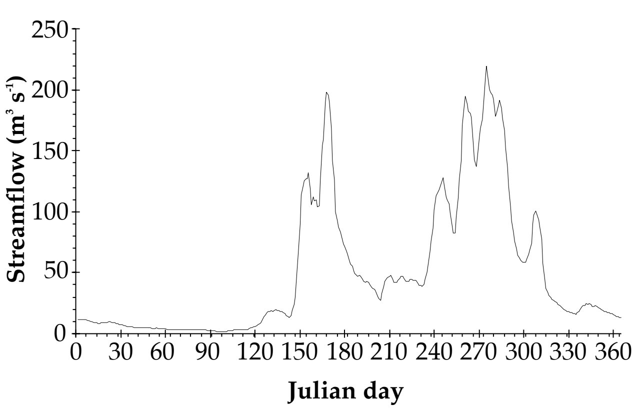

Figure 5. Average daily flows, seven-day running mean, Río Choluteca, 1979–1990.

rates are estimated at 200 cm yr^{-1}, and of course the dry-season rate is even higher. The river flow regime during the dry season becomes five months of virtually steady flow on the order of 5 to 10 $m^3 s^{-1}$.

Figure 5 emphasizes the long-term average behavior of the gauged flow. In fact, the high-flow seasons are "flashy," made up of individual storm hydrographs. This is not unexpected in view of the fact that precipitation in this area of Central America is driven by intense local thunderstorms embedded in tropical depressions.

The Río Choluteca also drains the urban areas of Tegucigalpa and Choluteca in Honduras and receives the wastewater from both of these municipal areas, which account for about 25% of the population of the country. Assuming a combined population in the watershed of one million, with a per capita biochemical oxygen demand (BOD) of 0.1 kg d^{-1}, the total load would be on the order of 100,000 kg d^{-1}. The data of Teichert-Coddington (1995) from the river downstream from Choluteca (and above tidal influence) show relatively low values of BOD but elevated concentrations of inorganic nitrogen (~0.5 ppm) and filterable phosphate (~0.25 ppm), which suggests that most of this wasteload is stabilized in its transit down the river channel. It is probable (though no data are yet available to confirm this) that the gauged flows in the dry season are predominantly wastewater return flows.

The estuarine data analyzed in this paper were collected in 1995. The rainy season in 1995 was exceptional, the regional precipitation being the highest in the 1973–96 record, as shown in the annual rainfalls of Figure 6. August 1995, in particular, logged the highest rainfall for any month in this entire period by a substantial factor. (The monthly rainfall graph for 1995 is identified in Figure 4. The prominence of the August 1995 value is evident.)

Hydrology is a key component of transport processes in an estuary, providing a source of dilution and throughflow, contributing an influx of terrestrial-derived constituents, and maintaining the seaward gradient in salinity. In order to close a mass budget on the channel estuaries of Monypenny Bay, it is necessary to include the inflows in the tributaries. Unfortunately, the only extant gauge is that on the Río Choluteca at Choluteca, which until recently flowed into Estero La Jagua, a tributary of El Pedregal. Inflows therefore had to be estimated from the precipitation data at Marcovia and Yusgüare, using the approximate drainage areas of

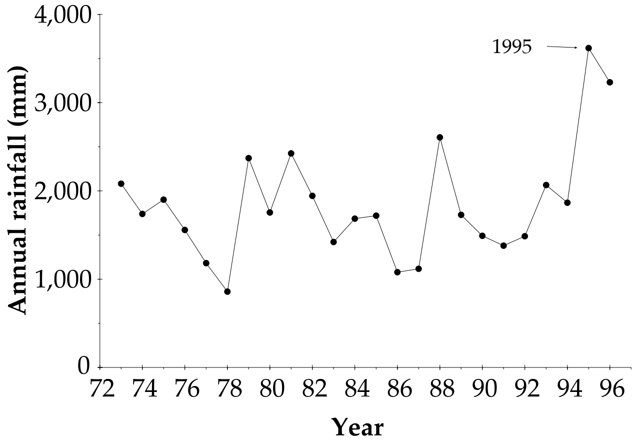

Figure 6. Average annual rainfall at Marcovia and Yusgüare, 1973–1996.

the tributaries and a coefficient of runoff. The coefficient of runoff, the ratio of runoff to rainfall, was taken to be 0.4 for El Pedregal and San Bernardo watersheds, based upon tropical watersheds of similar morphology and hydroclimatology (e.g., Instituto Geográfico Nacional, 1986; Larsen and Concepción, 1998), but the true value could lie anywhere in the range 0.3 to 0.7, with a corresponding error for the estimated inflows.

Shrimp Farms

Because the field data employed in this analysis were collected mainly in 1995, it was necessary to characterize the state of shrimp farm operation at this time. A concerted effort was made to obtain data on pond area, exchange, and management from the various farm operators with installations on the El Pedregal or San Bernardo systems. Figure 7 maps the shrimp farm deployment in this region as of 1995. The bold outlines approximate levee locations or other physical barriers (roads or infrastructure). Locations of pumps and drains are indicated by the respective symbols with regard to distributaries. Additional detail on the physical facilities at the farms is given in Table 1.

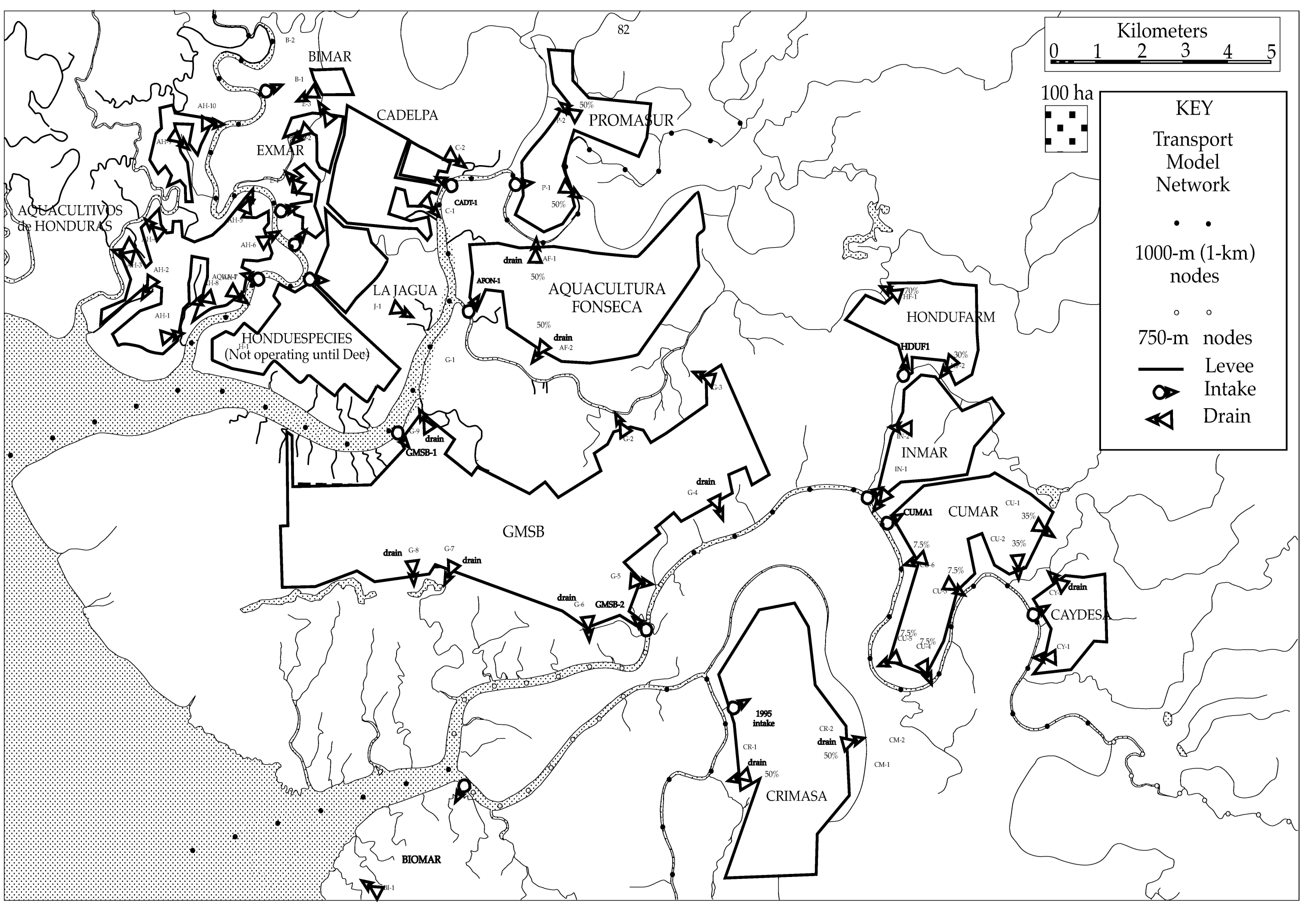

Figure 7. Study area showing 1995 shrimp farm configuration.

Table 1. Physical data on shrimp farms, 1995.

Name	Areas		Intakes			Drains			Exchange (% d^{-1})
	Ponds (ha)	*Infrastructure** (ha)	*ID***	*Location*	*Proportion* (%)	*ID***	*Location*	*Proportion* (%)	
Acuacultura Fonseca	682	110	AFON-1	El Garcero	100	AF-1	El Riel	50	2
Aquacultivos de Honduras	600	250	AQUA-1	Jagua		AH-1	Jagua	20	8
Year-round operation						AH-2	Purgatorio	8	
						AH-3	Purgatorio	7	
						AH-4	Purgatorio	5	
						AH-5	Jagua	10	
						AH-6	Jagua	10	
						AH-7	Jagua	10	
						AH-8	Jagua	10	
						AH-9	Jagua	15	
						AH-10	Jagua	5	
BIMAR	47	10		Jagua		B-1	Jagua	100	10
Dries out January–March									
CADELPA	312	93	CADT-1	Pedregal		C-1	Pedregal	80	3
Intake moved from Pedregal to Jagua in 1995; year-round operation									
CAYDESA	110	5		Bernardo		CY-1	Bernardo	50	10
Year-round operation									
CRIMASA	1,068	185		Berberia		CR-1	Berberia	50	6
Intake moved in 1998; year-round operation						CR-2	Comejenes	50	

* Roads, levees, fast land
** See Figure 7

Table 1. Continued.

Name	Areas		Intakes			Drains			Exchange (% d^{-1})
	Ponds (ha)	*Infrastructure** (ha)	*ID***	*Location*	*Proportion* (%)	*ID***	*Location*	*Proportion* (%)	
CUMAR	600	100	CUMA-1	Bernardo	100	CU-1	Quebrechal	35	7
Before 1996; year-round operation						CU-2	Quebrechal	35	
						CU-3	Bernardo	7.5	
						CU-4	Bernardo	7.5	
						CU-5	Bernardo	7.5	
						CU-6	Bernardo	7.5	
EXMAR	149	25		Jagua		E-1	Diabla	50	8
Dries out January–April						E-2	Diabla	25	
						E-3	Diabla	25	
GMSB	1,987	779	GMSB-1	Pedregal	75	G-2	Garcero	30	6
			GMSB-2	Bernardo	25	G-3	Garcero	10	
						G-4	Bernardo	10	
						G-5	Bernardo	15	
						G-6	Bernardo	15	
						G-7	Bernardo	10	
						G-9	Pedregal	10	
HONDUESPECIES	357	116		Jagua		n/a			0
Closed 2–3 years prior to December 1995									
HONDUFARM	350	50	HDUF-1	Pinguinos		HF-1	Garcero	70	10
						HF-2	Pinguinos	30	
INMAR	238			Bernardo		IN-1	Bernardo	30	12
Includes CULMASA-2						IN-2	Pinguinos	70	
La Jagua	203	50		Jagua		J-1	Pedregal	100	2
PROMASUR	378	135		Pedregal		P-1	Pedregal	50	6
Operates March–December, dries out January–April						P-2	Pedregal	50	

* Roads, levees, fast land

** See Figure 7

From the standpoint of determining the impact of shrimp farm operations on receiving-water quality, the important data are the flow out of the pond, its oxygen-demanding constituents, and the point at which pond effluent enters the watercourse. The flow out of the pond is estimated by the volume of the operating ponds and the daily exchange (given in Table 1 as a percent of pond volume per day). The constituent concentration is determined by the labile organic matter in the effluent, whose effect on dissolved oxygen is discussed below. Most farms have multiple drains, which are shown in Figure 7 and summarized in Table 1. The distribution of effluent into the separate drains from a farm was based upon estimates provided by the farm manager. Estimates were made by inspecting the pond areas serviced by the drain or, when no further information was available, by assuming an equal distribution of effluent into the available drains.

Pond effluent contains a varied mixture of mineral and organic byproducts, including organic carbon compounds and inorganic and organic species of nitrogen, phosphorus, and silicon in both suspension and solution, as well as a variety of microorganisms. Some of these constituents directly consume DO in chemical reaction, in respiration, or in bacterial degradation of complex organics, while others have an indirect effect on DO in stimulating the growth of photosynthetic or respiratory organisms in the watercourse. As a gross measure of the oxygen-demanding potential of the effluent, we employ BOD, the oxygen consumed in a specified time period per unit volume of water in the absence of photosynthesis. Samples of farm effluent were initially saturated with DO and maintained in sealed, dark bottles in a stable laboratory environment, and measurements of DO in the samples were performed at regular intervals. Details are given in Teichert-Coddington et al. (in prep.). The DO consumption over time increases rapidly at first then approaches an asymptotic value. This asymptote, termed the ultimate BOD, denoted BODu, is conceived to be the maximum potential oxygen demand from the labile constituents of the sample. In terms of BODu, the BOD progression is analyzed as an exhaustion process, following the relation:

$$L = Lu \exp(-kt)$$

$$D = Lu - L$$

where

D = cumulative oxygen depleted after time t (in days) and

Lu = BODu.

A least-squares-fit simultaneous solution for rate coefficient k and BODu was determined for each DO depletion series, for which three or four replicates were used and measurements were made at days 2, 5, 7, 14, and 21. An example of the data and best-fit BOD-reaction is shown in Figure 8. Because no bacterial "seed" is added, the BOD so determined is the result of the normal bacterial community in the pond effluent, so the reaction rate k is a good measure of what would prevail in the watercourse. Samples from both influent and effluent were analyzed in this manner, so the incremental BODu added by the pond to BODu already in the estuary water could be isolated.

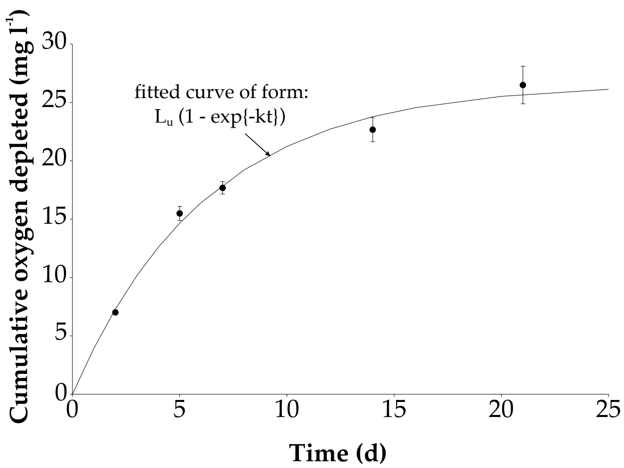

Figure 8. Oxygen depletion data and fitted exhaustion curve, GMSB discharge, 4 January 1995.

The only farm from the El Pedregal/San Bernardo region for which such detailed BOD analyses were performed is Granjas Marinas San Bernardo (GMSB). The incremental BODu added by the farm was determined to be about 12 mg l^{-1} and the rate coefficient (at 20°C) to be about 0.15 d^{-1}. (The order of magnitude of BODu was checked against computations of carbon oxidation assuming typical stocking and grow-out data for the Honduran farms.) Part of the organic byproducts is oxidized in the pond itself and the remainder rejected to the receiving watercourse in the effluent. The relative proportion of the two depends upon the exchange rate of the pond. The data from GMSB were scaled to the other farms in the El Pedregal/San Bernardo study area according to their individual exchange rates (Table 1). The product of the resulting effluent BODu concentration and the discharge flow

Table 2. Estimated BOD loads for 1995 shrimp farms.

Farm	Pond Area (ha)	Exchange (% d^{-1})	Throughflow (Mm^3 d^{-1})	BODu (mg l^{-1})	BOD Load (kg d^{-1})	(lb d^{-1})
Acuacultura Fonseca	682	2	0.136	4	546	1,200
Aquacultivos de Honduras	600	8	0.48	16	7,680	16,896
BIMAR	47	10	0.047	20	940	2,068
CADELPA	312	3	0.0936	6	562	1,236
CAYDESA	110	10	0.11	20	2,200	4,840
CRIMASA	1,068	6	0.641	12	7,690	16,917
CUMAR	600	7	0.42	14	5,880	12,936
EXMAR	149	8	0.1192	16	1,907	4,196
GMSB	1,987	6	1.192	12	14,306	31,474
HONDUESPECIES	357			*(Not operating in 1995)*		
HONDUFARM	350	10	0.35	20	7,000	15,400
INMAR	238	12	0.286	24	6,854	15,080
La Jagua	203	3	0.0609	6	365	804
PROMASUR	378	6	0.227	12	2,722	5,988

* For pond volume calculations, mean pond depth was estimated to be 1.0 m.

(with appropriate units conversions) is the BODu load (mass per unit time) tabulated in Table 2.

Hydrography of the Channel Estuaries

Hydrography of the channel estuaries was addressed through analysis of field data and application of a tidal hydrodynamic model. The hydrographic influences include channel morphology (bathymetry and cross sections), freshwater inflow, and density structure, which is dominated by salinity and forced from the Gulf of Fonseca, notably by tides. Most important is the effect of tides. As noted earlier, the tide apparently is amplified in propagating into the Monypenny Bay area, due to convergence of the cross section. The result is a semidiurnal (12.4 h) tide of amplitude from 2 to more than 3 m.

In the early work on these estuaries (Ward, 1995), there was virtually no information on channel geometry, so these parameters (depth, width, cross section area) had to be estimated from inspection of topographic maps and from limited field excursions on the estuaries. In the subsequent period, much better data on estuary depths and some valuable cross section survey data were provided by Granjas Marinas. Consequently, the physiographic depiction of the channel estuaries is much improved, though data is still needed on the upper (i.e., upstream) reaches.

Because there is no operating tide gauge in this region of the Gulf, we have little quantitative information on tidal behavior. During this project a recording water-level meter was operated for short periods of time to monitor tides. Figure 3 (p. 6) shows the monitored tide in Estero El Pedregal. The resulting movement of water with the flood and ebb in these channel estuaries is substantial.

The channel estuaries addressed in this study are in fact distributaries of a large fluvial swamp/marsh complex in the eastern segment of Gulf of Fonseca (Figure 1, p. 2). It is a network of dendritic channels maintained by tides and seasonal runoff, which incise extensive tidal flats. The tidal channels are fringed by dense growths of mangroves. There are two main tidal channels in the Pedregal: the Estero El Pedregal and the Estero La Jagua, which until recently received the inflow from the Río Choluteca and conflows with the Pedregal 2 km upstream from its mouth (Figure 2, p. 5). Similarly, there are two main tidal channels for the San Bernardo: the Bernardo per se, which receives inflow from the Río Negro and the Estero La Berberia that conflows with the San Bernardo about 5 km up from the mouth. An important geometric feature of the Pedregal and the San Bernardo is their sharply declining cross-sectional area with distance upstream: these are horn-shaped estuaries, whose cross section drops from nearly 25,000 m^2 at the mouth to less than 50 m^2 in about 30 km. Therefore, the channel itself has a quickly diminishing capacity for flow, as well as a quickly increasing resistance to flow.

An equally important feature is the large tidal flats, which communicate with the main tidal channel through small scoured tidal passes through the mangrove fringe. These tidal flats have the capacity to store a great amount of water on the rising tide and release that water back to the tidal channel as the tide stage falls. The extent to which the flats are regularly flooded therefore is an important feature of the tidal functioning of this system.

One concern expressed early in this work was the possibility that installation of shrimp pond levees might reduce the tidal prism and thereby diminish tidal flushing, contributing to degraded water quality in the tidal estuaries (Ward, 1995). In effect, these shrimp farms could eliminate the tidal flats, hydraulically isolating these areas by enclosure within levees to create their shrimp ponds. A concerted effort was made to evaluate this potential by determining the extent of regular inundation of the tidal flats before shrimp pond construction and to locate as precisely as possible the actual extent of the farm levees. The former was accomplished through examination of topographic maps and satellite photographs and through the efforts of Felix Wainwright (1996), who has studied these areas extensively. These areas of normal inundation are mapped in Figure 9. The practice of most of the shrimp farms of placing the levees inside the mangrove fringe follows approximately the extent of normal tidal inundation, so the impact on tidal prism reduction is limited. The farms in the upstream sections of the estuaries are the exception, but in these regions the more important control on water quality is the volume and quality of inflows (see the next section, Water Quality of the Channel Estuaries).

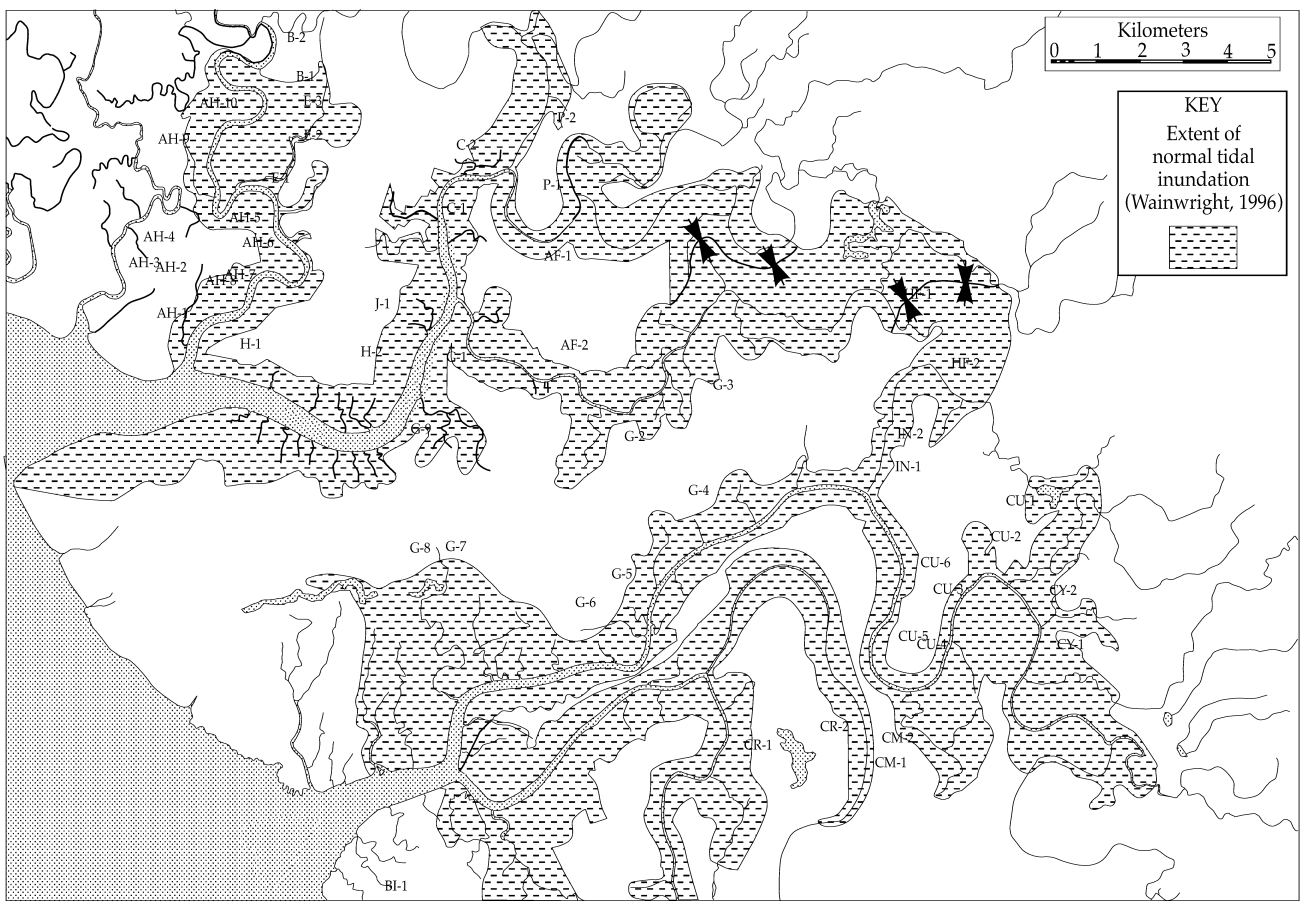

Figure 9. Areas inundated under normal tide variations (Wainwright, 1996).

The salinity structure is an important aspect of the hydrography of these channel estuaries because it is a direct demonstration of the movement of water in response to tides and because it provides an index to density stratification. Salinity is a virtually conservative parameter, its value being determined by the mixing of fresh and saline waters. Because salinity is also a measure of water quality, and consideration of the salinity structure provides insight into the observed dissolved oxygen variation, presentation and analysis of field data on salinity structure is addressed in the next section, Water Quality of the Channel Estuaries. In the present context, we note that the 1995 field data on salinity indicate the tidal excursion in the lower reach of the estuaries to be at least 25 km in the Pedregal and 10 km in the San Bernardo. Moreover, within the salinity intrusion zone`, there is also substantial vertical stratification, especially on the flooding tide. This is due to the longitudinal salinity gradient created by the freshwater inflow and to the tidal influx of more saline water from Monypenny Bay. Such dramatic stratification appears to be a phenomenon only of the combination of transient conditions during the high inflow freshet of the rainy season. With the stabilization of inflows in July, the longitudinal salinity gradient mixes out more and the vertical stratification is much less pronounced. This behavior is addressed in more detail in the following section.

A tidal hydrodynamic model was applied to the combined Pedregal-Jagua system and the combined San Bernardo-Berberia system. This model is a numerical solution to the differential equations of momentum and continuity, which are averaged over the cross section of the estuary so that the system is treated one-dimensionally, i.e., as a function of distance along the main axis (Dronkers, 1964; Hauck and Ward, 1980). These equations are of the following form:

$$\text{momentum:}\quad \frac{\partial Q}{\partial t} + \frac{\partial uQ}{\partial x} = -gA\frac{\partial h}{\partial x} - g\overline{D}\frac{Q^2}{C^2} \qquad (1)$$

$$\text{continuity:}\quad \frac{\partial Q}{\partial x} + B\frac{\partial h}{\partial t} = q$$

where

A = cross-sectional area and

Q = uA = longitudinal flow.

For the present modeling work, the friction coefficient C was converted to an equivalent form involving Manning's n as the roughness parameter.

Because of the importance of the tidal flats to the hydrography of this system, special provision is made in the numerical treatment. Momentum is considered to be confined to the main tidal channel, and the momentum equation 1 is solved for that cross section A. The adjacent tidal flat acts as storage and is activated whenever the water level in the channel exceeds the elevation of the bank. The area of the tidal flat is an input to the model that must be determined from survey data, in this case from the tidal inundation areas provided by Wainwright (1996) shown in Figure 9.

For numerical solution, the estuary channel is discretized into finite segments (not necessarily of the same length) from its mouth to an upstream terminal point taken to be the head of tide. The momentum and continuity equations are then solved by the method of finite differences for each of the nodes defined by the segmentation. A key boundary condition is the tidal stage variation at the mouth of the estuary, which is an external input to the model. This method of solving the hydrodynamic equations in a tidal system was documented by Dronkers (1964). Its adoption for a flooding tidal flat is described by Hauck and Ward (1980), who applied it to several tidal deltas on the Texas coast. Wolanski (1993) recently applied the same basic model to mangrove swamps in Australia. However, we did not assign a high friction to the mangrove flats as he did, because in the Gulf of Fonseca area the mangrove flats are relatively narrow fringes with scoured tidal channels.

Each of the Pedregal and San Bernardo systems was treated as a linked network of computational nodes. The hydrodynamic model nodes are shown in Figure 10. Two different input files were created for each, one with "natural" physiography, i.e., with tidal flat areas as determined from topographic maps prior to shrimp farm development and the tidal flat areas of Figure 9, and the second with 1995 shrimp ponds in place, as depicted in Figure 2 (p. 5). Since no tidal data were available farther up these estuaries, frictional dissipation had to be judged; a Mannings n of 0.030 (British units) was assigned to all tidal channels.

The value of this model is that it gives us a means to compute the tidal currents in the estuary based upon the boundary condition of tidal stage in Monypenny Bay. The currents, rather than tide stage, are really the important hydrographic feature, since it is the currents that are responsible for transport and

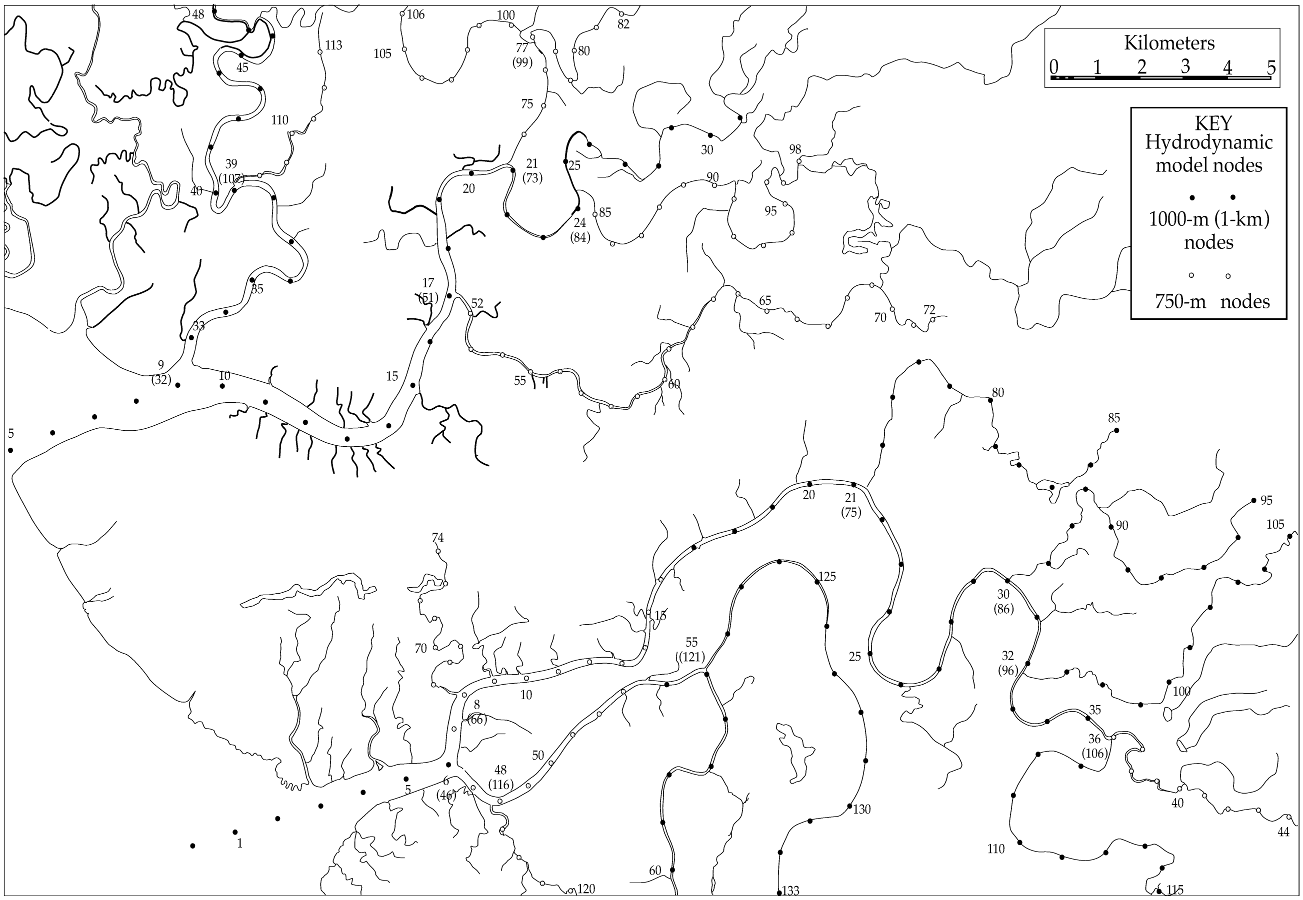

Figure 10. Computational nodes for hydrodynamic model.

tidal dispersion. The tide record of Figure 3 (p. 6) was obtained from a gauge located at the Pedregal intake to the Granjas Marinas farm and used as an estimate of the tide in Monypenny Bay at the mouths of these estuaries. Time integrations of several tidal cycles were carried out, solving for tidal current and water level throughout the estuary, to determine the following three key hydrodynamic indicators:

- tidal excursion—the distance that a parcel of water moves on the flooding tide;
- mean tidal-current speed; and
- tidal prism—the volume of water carried past a fixed point on the flooding tide.

Modeled tidal excursions appear to be on the correct order, compared to observations of Currie (1994) who reported that several buoys released in the lower Estero Real traveled about 24 km upstream and compared to the field observations of tidal excursion in salinity structure discussed in the next section. The most important uses of the tidal calculations are to provide a basis for estimating longitudinal dispersion in the transport modeling (discussed in next chapter) based upon tidal excursion and to compute oxygen reaeration, which is a strong function of current speed.

Water Quality of the Channel Estuaries

From the standpoint of the scientific value of the work in Honduras reported here, the most significant contribution is the collection of field data from the channel estuaries. This was a combined effort of many contributors, stimulated and directed by the PD/A CRSP Auburn University researchers. An inventory of the field data collected in El Pedregal and San Bernardo estuaries is given in Table 3. Only a portion of the data collected has been analyzed and employed in the present study. The data will certainly yield much more information than is presented here.

The measurements consisted of vertical profiles at 0.5-m intervals from surface to bottom of temperature, dissolved oxygen, and conductivity, using an electrometric probe and deck readout. Salinity is determined from conductivity and temperature. Data were

Table 3. Inventory of field data sets for Monypenny Bay estuaries. Each data set consists of vertical profiles at each station at both high and low tide unless otherwise noted.

Date(s)	Parameters		Number of Stations	Sampled Reach (km)	Comment
	profiled	*sample**			
ESTERO PEDREGAL					
21 Nov 94	S/T/DO		11	22	Low tide only
10 Mar 95	S/T/DO		10	22	
22 Jun 95†	S/T/DO	X	7	11	Water samples Stations 1–4
29 Jun 95†	S/T/DO	X	7	11	Water samples Stations 1–4
6 Jul 95†	S/T/DO	X	7	11	Water samples Stations 1–4; DO low tide only
13 Jul 95†	S/T/DO	X	7	11	Water samples Stations 1–4
20 Jul 95†	S/T/DO	X	7	11	
27 Jul 95†	S/T/DO	X	7	11	
4 Mar 96	S/T/DO		7	11	
11 Mar 96	S/T/DO		7	11	
19 Mar 96	S/T/DO		7	11	
26 Mar 96	S/T/DO		7	11	
3 Apr 96	S/T/DO		7	11	
ESTERO SAN BERNARDO					
20 Jan 95	S/T/DO		8	26	
11 Mar 95	S/T/DO		9	31	Station 9 at high tide only
10–11 Aug 95†	S/T	X	7	14	Water samples Stations 1–4
17 Aug 95†	S/T	X	7	14	Water samples Stations 1–4
24 Aug 95†	S/T/DO	X	7	14	Water samples Stations 1–4
1 Sep 95†	S/T	X	7	14	Water samples Stations 1–4
8 Sep 95†	S/T/DO		7	14	
17 Mar 96	S/T/DO		7	15	
25 Mar 96	S/T/DO		7	15	
1 Apr 96	S/T/DO		7	15	

* water samples for chemical analyses

† rainy season

obtained in both the dry and rainy seasons. These data were used for two purposes. First, analysis of these data provided a characterization of the hydrography and water quality of the estuaries. Second, the data were used to validate a numerical model of constituent transport, which then became the basis for evaluating alternative levels of shrimp farm development to assess the carrying capacity of the system.

Station locations (for the rainy-season data) are shown in Figure 11. The initial influx of precipitation in May and June 1995 extruded the salinities from the Pedregal and left the water nearly fresh except for the lower 10 km. The diminished precipitation and runoff in July (see the section Climatology and Hydrology) then allowed re-intrusion of salinity. The time histories of precipitation at Marcovia and vertical-mean tidal-mean salinity at the CRSP stations display the dynamics of salinity extrusion and intrusion (Figure 12). It is noteworthy that after the response to the June pulse of inflow, the re-intruded salinity structure acquired an approximate steady state in July that was in equilibrium with the freshwater throughflow. The same sort of response and acquisition of equilibrium is evidenced in the DO data, again averaged in the vertical dimension and from low to high tide (Figure 13).

By averaging the data over the tidal cycle and in the vertical dimension, the longer-term variation in

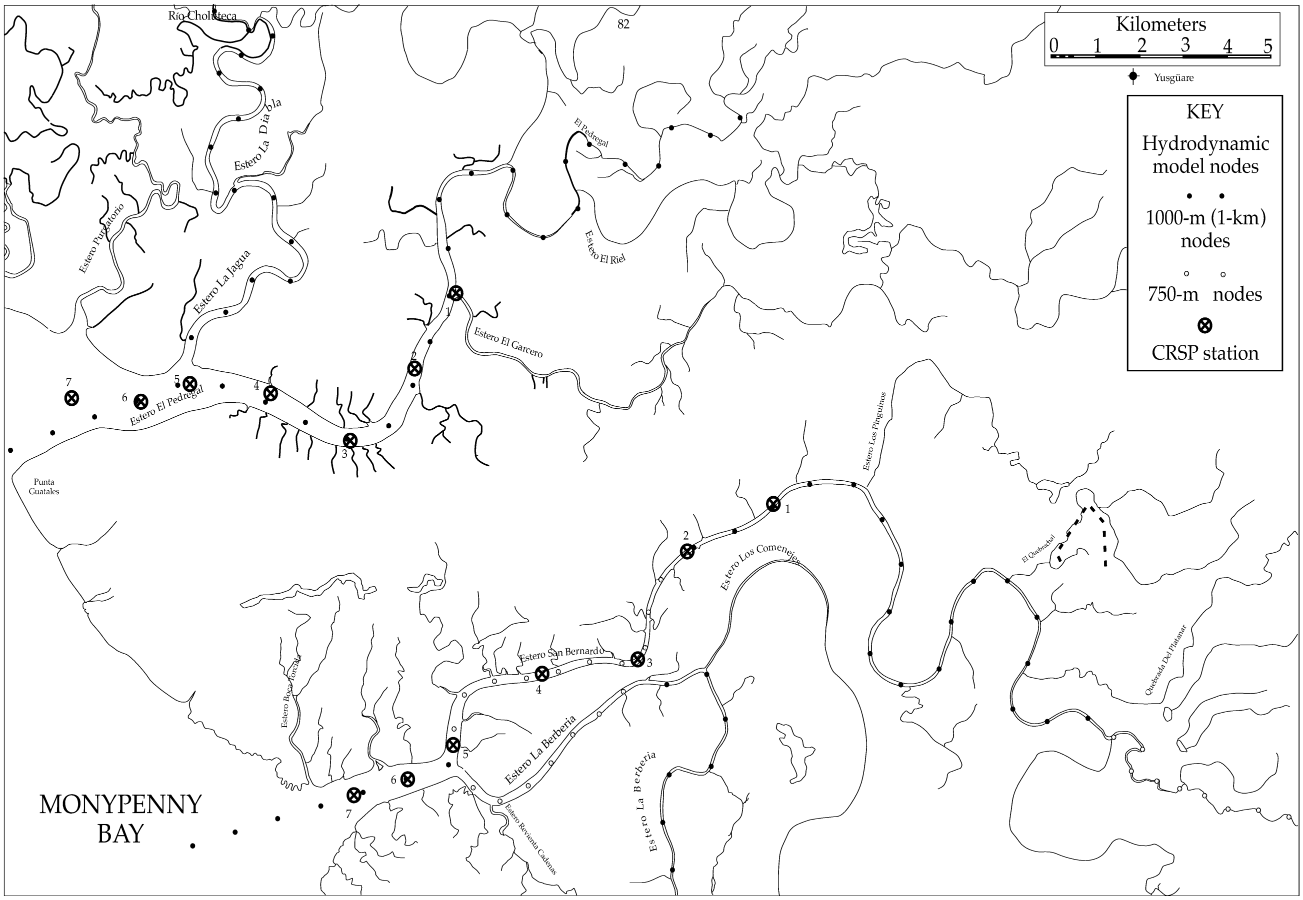

Figure 11. Sampling stations (rainy-season profile) and transport model network.

estuary structure is made evident (Figures 12 and 13). But there is considerable variability in estuary structure both in the vertical dimension and over the tidal cycle. This is illustrated by the data profiles during the salinity re-intrusion, within the salinity gradient reach (Figure 14). The range of salinity from high to low tide is especially noteworthy.

Another useful display of the data is as a longitudinal axis versus depth section for both high and low tide for a specific sampling run. Figures 15 and 16 are examples, depicting salinity and DO structure, respectively. (In these and later graphs, longitudinal position along the estuary axis is measured in kilometers from a point far upstream from the region of shrimp farm development.) The salinity structure

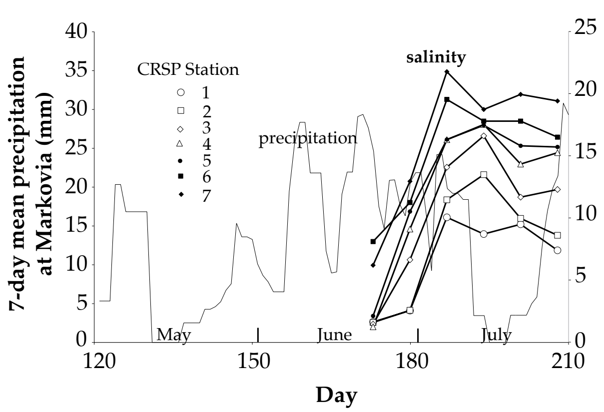

Figure 12. Time history of precipitation and salinity in Estero El Pedregal, rainy season 1995.

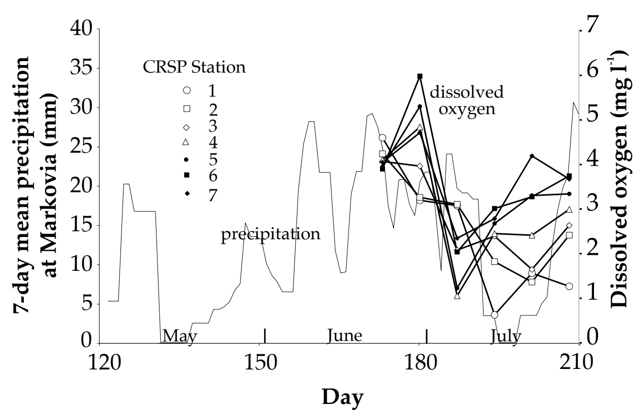

Figure 13. Time history of precipitation and dissolved oxygen in Estero El Pedregal, rainy season 1995.

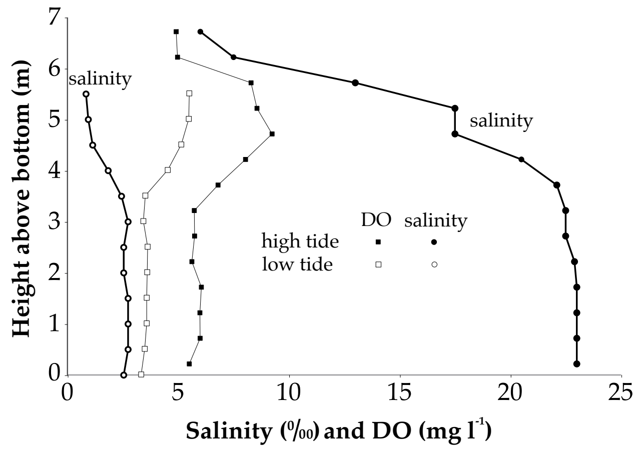

Figure 14. Vertical profiles in El Pedregal at CRSP Station 5, 29 June 1995.

is particularly important, because it is a direct demonstration of the movement of water in response to tides. It is virtually conservative, its value being determined only by the mixing of fresh and saline waters. On the rising tide, a volume of water is driven up the estuary large enough to completely replace the salinities less than 3‰ with salinities approaching 25‰ from Monypenny Bay. Then on the falling tide this entire volume drains back down the estuary to return salinities to the low values of 2 to 3‰. The tidal intrusion of water from Monypenny Bay also brings higher concentration DOs into the lower reach of the estuary. Some of these DOs are highly supersaturated.

Within the salinity intrusion zone there is also substantial vertical stratification, especially on the flooding tide. This is due to the longitudinal salinity gradient created by the freshwater inflow and to the tidal influx of more saline water from Monypenny Bay. Such dramatic stratification appears to be a phenomenon only of this combination of conditions. With the stabilization of inflows in July, the longitudinal salinity gradient mixes out more and the vertical stratification is much less pronounced (Figure 17). The rainy-season data collection did not begin on the San Bernardo until mid-August, by which time the profiles were responding to the

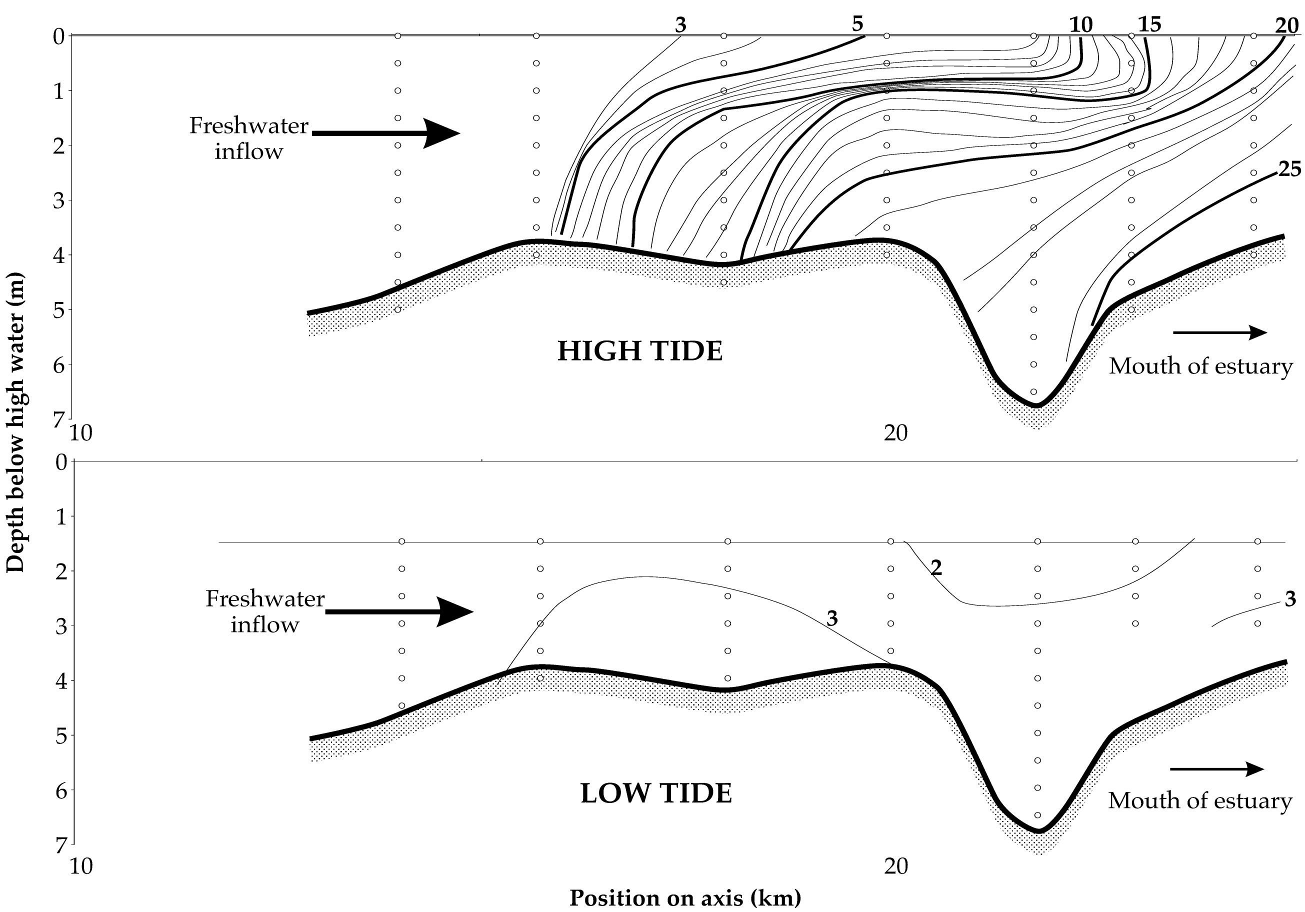

Figure 15. Longitudinal-depth section of salinity, Estero El Pedregal, 29 June 1995.

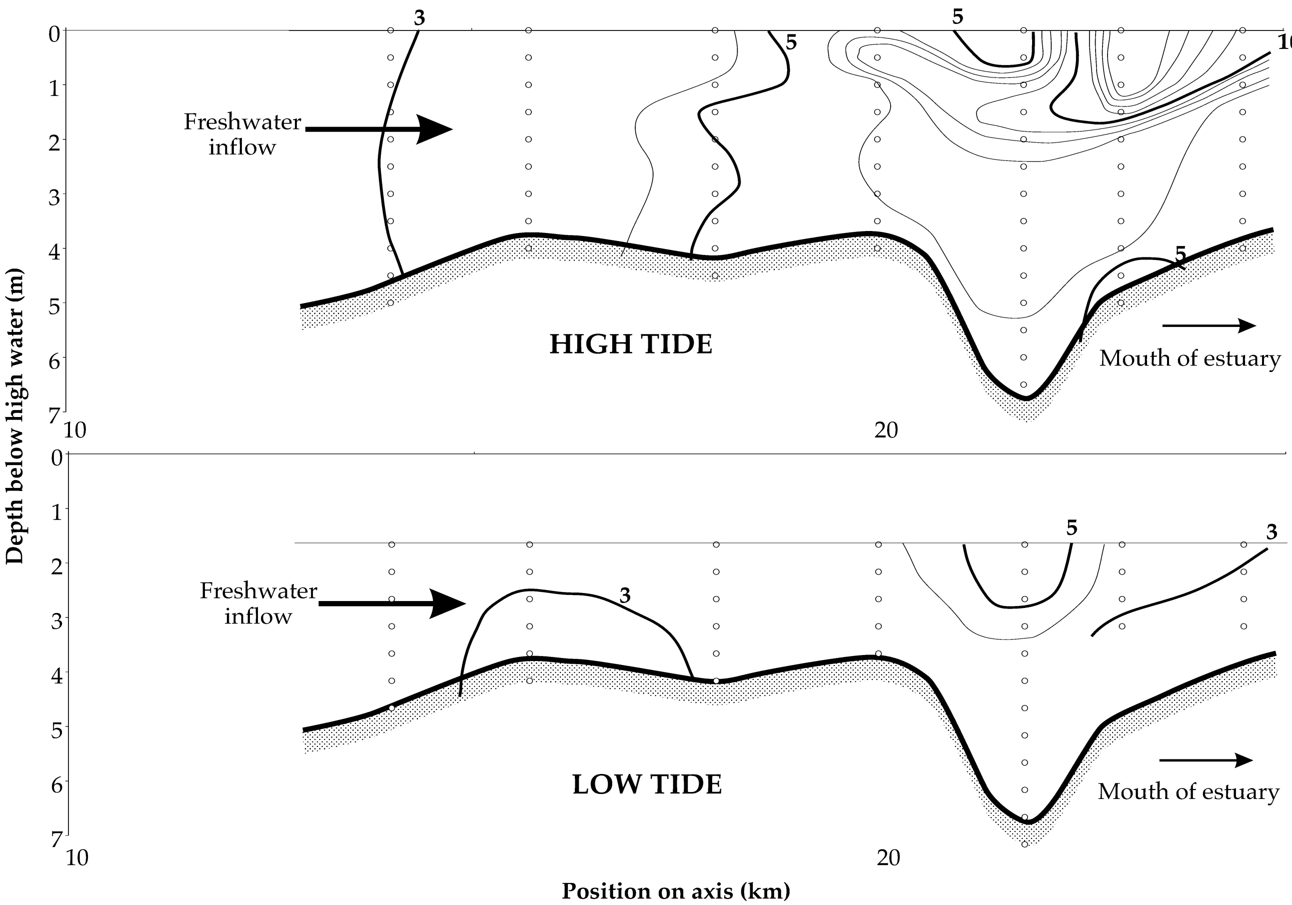

Figure 16. Longitudinal-depth cross section of DO, Estero El Pedregal, 29 June 1995.

August freshet. Most of the sampled reach exhibits vertical near-homogeneity by dint of the fact that the saline intrusion reach is much farther down the estuary toward the mouth. An example from 24 August is given in Figures 18 and 19. From these cross sections the tidal excursion in the lower reach of the estuary is seen to be at least 25 km in the Pedregal and 10 km in the San Bernardo.

A numerical transport model was applied to each of these estuaries to model salinity and DO. The model assumes an equilibrium profile, i.e., steady state, and treats the longitudinal variation of properties, i.e., the tidal-mean and vertical-mean concentrations, as a balance between transports and kinetics. The vertical-mean approximation is certainly valid for salinity except for the very dynamic responses during a freshet event, as exemplified by the longitudinal-depth section of Figure 15. The steady-state approximation is appropriate for the stable profiles exhibited in July and August, as demonstrated by Figures 12 and 13. Moreover, there is residual uncertainty in many of the parameters controlling water quality in these systems, which in our view argues against a more complex model with even more poorly defined parameters.

Briefly, the model is a numerical solution to the finite-difference approximation to the steady-state, section-mean mass balance equation, in which internal kinetics, loads, and boundary conditions (at the ends of the modeled reach of the estuary) must be specified:

$$\frac{\partial c}{\partial t} = \frac{1}{A}\left[-\frac{\partial Qc}{\partial x} + \frac{\partial}{\partial x} EA \frac{\partial c}{\partial x}\right] + \sum S_i \qquad (2)$$

where

c = section-mean mass concentration of substance,

E = longitudinal dispersion coefficient,

S_i = source or sink of substance (including kinetics and loads),

and the other symbols are as before.

The steady-state approximation, whose solution in a one-dimensional system is detailed in many standard references (Ward and Espey, 1971; Thomann and Mueller, 1987), requires $\partial c / \partial t = 0$. This approximation eliminates time as an independent variable, and the finite-difference equation becomes a tridiagonal array, which is inverted by Gaussian elimination. This means the computational demands of the model are much more modest than those of the time-advancing hydrodynamic model presented in the Hydrography of the Channel Estuaries section, but the mathematical setup of the model is more complex. The same numerical node system was employed as for the hydrodynamic model, depicted in Figure 10 (p. 16), to ensure that the physiographic inputs (channel depths and cross-sectional areas) are consistent and to simplify transport of variables from one model to the other (such as mean tidal current speed). Two minor changes were made. First, the computational overhead of solving the equations for a diffluent segment (where the Jagua connects to the Pedregal or the Berberia to the San Bernardo) did not justify directly coupling the two channels. Instead, the primary channel model (Pedregal or San Bernardo) was operated first, with flow from the secondary channel (Jagua or Berberia) into the primary channel included. Then the salinity, BOD, and DO values at the confluence were used as boundary conditions to drive the model for the secondary channel. The second minor change is that the indices are numbered in the opposite direction for the mass balance model.

Three versions of equation 2 are used in the model, in which c depicts salinity, DO, and BOD. Salinity is a conservative parameter (if evaporation is not large), which means that the profile of salinity is governed by the single source—the salinity at the mouth of the estuary and the mixing and dispersion processes that transport salinity up the estuary. Therefore $\sum S_i = 0$ in equation 2. One of the principal values of modeling salinity is that it provides a direct test of how accurately the transport terms are quantified, since sources and sinks are eliminated from the problem. The other two parameters are DO and BOD. The level of DO is one of the most fundamental

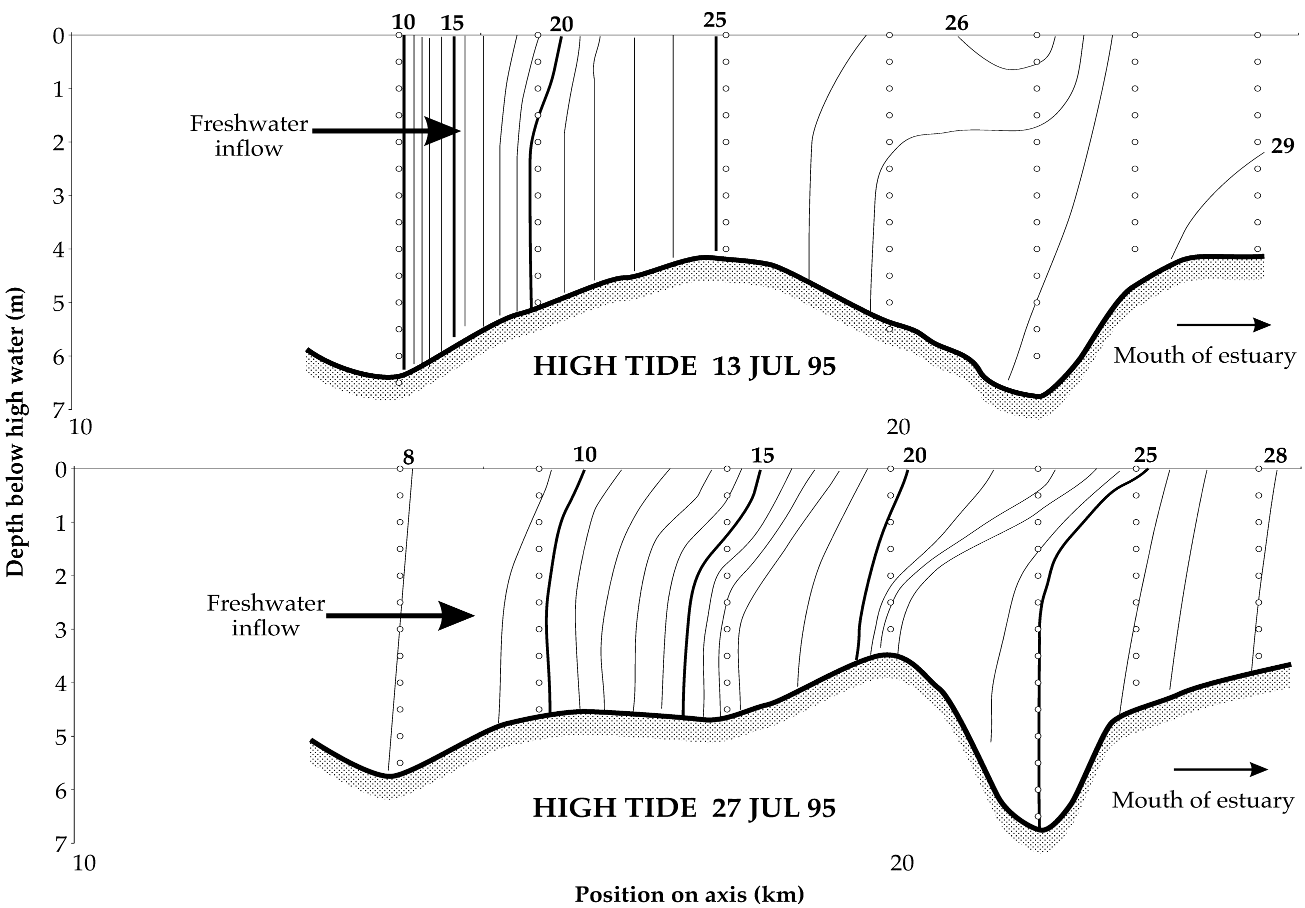

Figure 17. Longitudinal-depth cross sections of salinity, Estero El Pedregal, July 1995.

measures of estuary quality and is certainly an important constraint on suitability of estuary water as shrimp pond influent. In order to model DO, BOD must be modeled first and fed forward into the dissolved oxygen calculation, which represents a coupled calculation; BOD is driven by the sources of BOD in the form of background organics and shrimp pond effluents and then fed forward into the DO equation as a sink term.

Each estuary was depicted by a network of computational nodes running up the channel to a point well above the region of shrimp farm development. These computational networks are shown in Figures 7 (p. 9) and 11 (p. 18). The Pedregal main channel and its La Jagua tributary and the San Bernardo and its Berberia tributary are explicitly modeled. The other tributaries, which are less important, were not modeled explicitly in the transport model, but instead are depicted simply as an inflow and associated concentration. For salinity, as noted above, only the inflow, the boundary conditions, and the longitudinal dispersive, i.e., nonadvective, transport need to be specified. Inflow was estimated from the 1995 precipitation data averaged for Marcovia and Yusgüare, with a runoff coefficient of 0.4 as described in the section Climatology and Hydrology. Dispersion coefficients were estimated based upon the location in the salinity intrusion reach, using the estuary data compiled in Ward and Montague (1996). This is, however, a spongy parameter which is difficult to estimate with any accuracy a priori.

The model application to the Pedregal salinities for July inflows is shown in Figure 20. For comparison, the tidal-mean vertical-mean salinities are also shown. The vertical bars on the profile data represent the tidal range in salinity observed at that station for that sample run. Figure 21 shows a similar plot for the San Bernardo. (There are no profile measurements from either the La Jagua or the Berberia, so these model results are not displayed.) Generally, the agreement between model and observed salinities is considered satisfactory. As before, longitudinal position is measured in kilometers from a point far

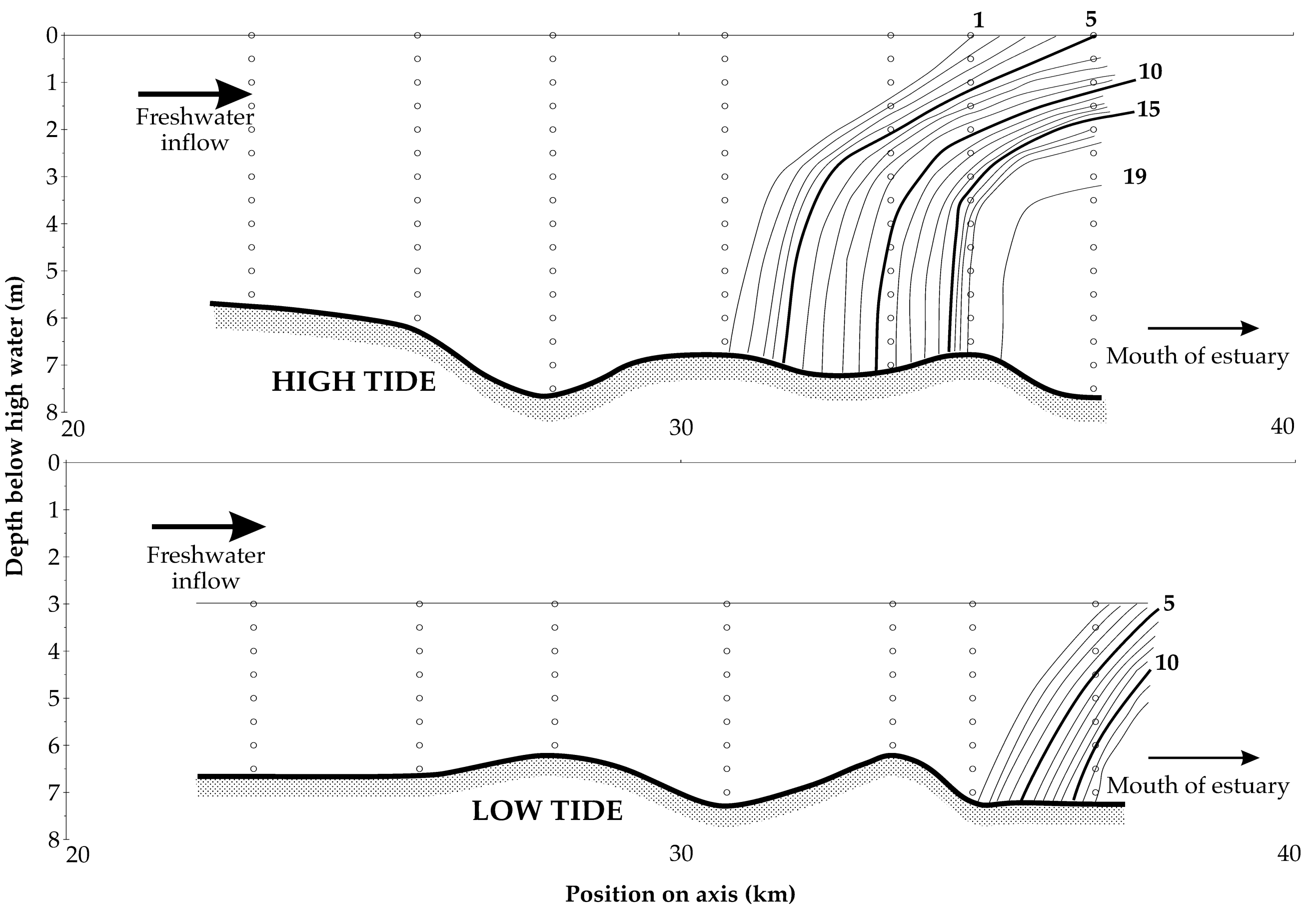

Figure 18. Longitudinal-depth section of salinity in Estero San Bernardo, 24 August 1995.

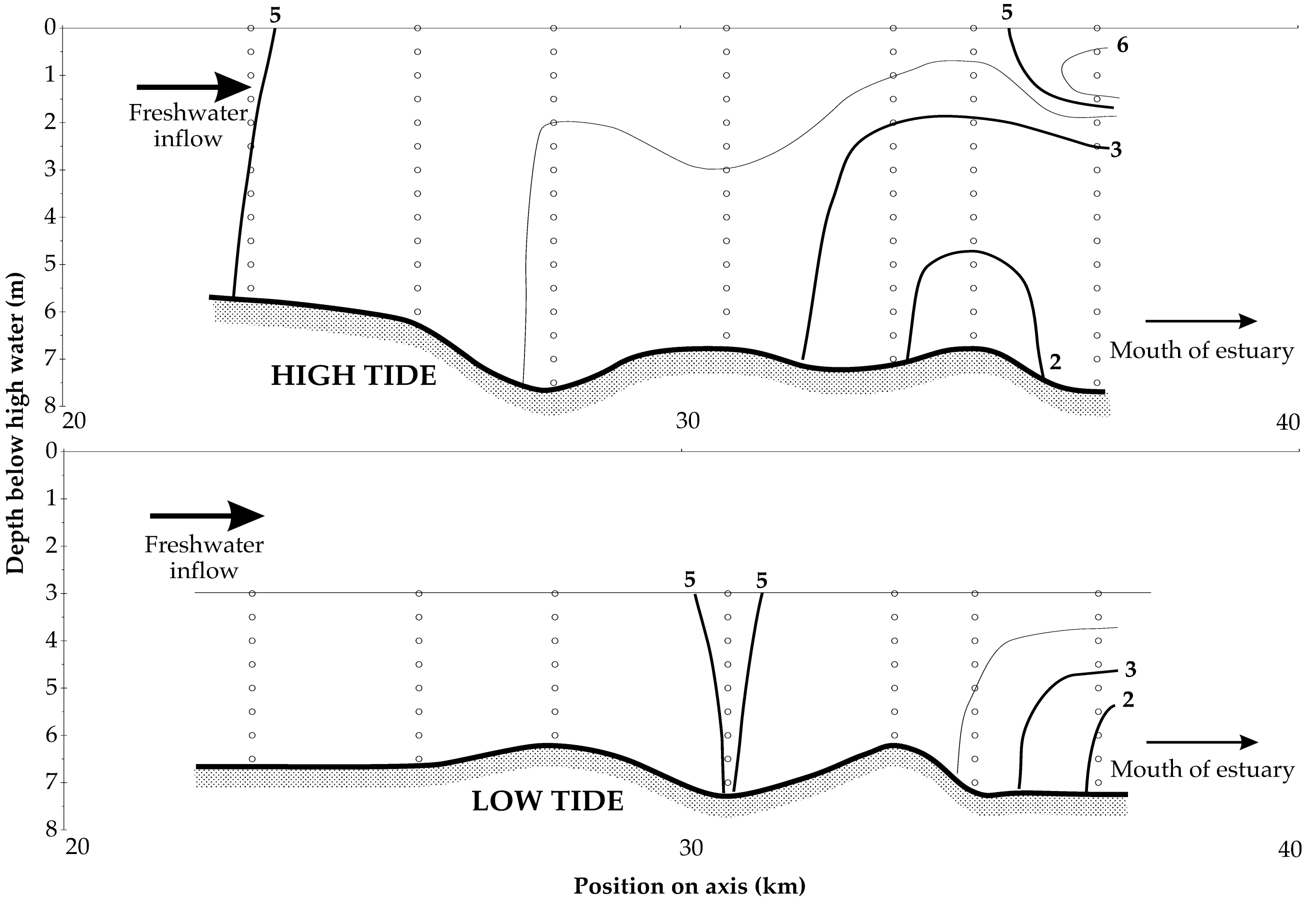

Figure 19. Longitudinal-depth section of DO in Estero San Bernardo, 24 August 1995.

upstream on both Figures 20 and 21. Several geographical features are identified on these figures to help orient the reader.

The objective of the modeling is to determine the DO response to shrimp farm loading. The main purpose of modeling salinity is to provide some assurance that the transport terms are approximately correct in the model. Modeling DO not only requires the same transport terms as in the salinity model, but also specification of the various kinetic processes that affect DO in the watercourse. For present purposes, these are restricted to surface reaeration and water-column degradation of BOD. Other processes, such as zero-order sinks of DO (e.g., sediment oxygen

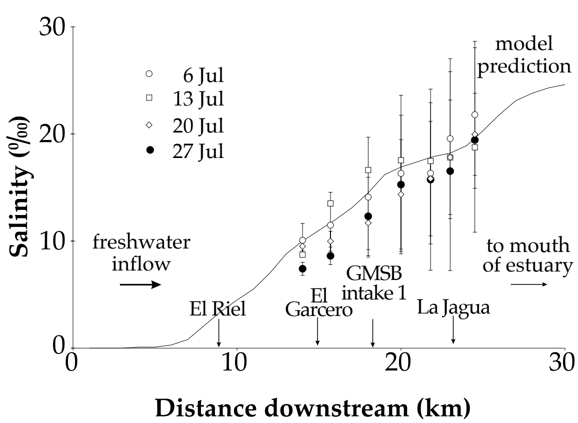

Figure 20. Field observations and model predictions of salinity, El Pedregal, July 1995.

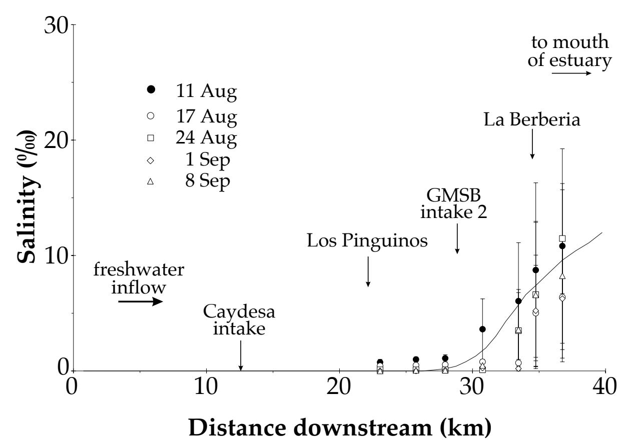

Figure 21. Field observations and model predictions of salinity, San Bernardo, August–September 1995.

demand and algal photosynthesis / respiration) can be accommodated in the model, but we have no direct measurements of these in the Monypenny Bay estuaries. We preferred to omit their consideration for the initial model setup, so that the comparison of model prediction to observation could serve as an appraisal of the importance of these other processes. Reaeration was computed using the O'Connor-Dobbins (1958) equation with tidal current velocities that were determined from the companion hydrodynamic model (although the above tidal excursion estimates would have been sufficiently accurate for this purpose). BOD degradation was assumed to proceed with the rate coefficient inferred from the laboratory BOD determinations (see the Shrimp Farm section). The inflow from the adjoining salt flats and tidal marsh watershed is considered to be high in labile organics (Flores-Verdugo et al., 1990), and the normal DO content of this water is low. To depict these effects, a BODu of 10 mg l^{-1} was assumed for the inflows.

Figure 22 shows El Pedregal DO profiles measured in July during the rainy season and the corresponding model prediction of DO. The total shrimp farm BOD load to El Pedregal for 1995 conditions is about 16,000 kg d^{-1}. The data show decreasing DOs with distance upstream from the mouth. The model predicts a substantial zone of low oxygen, the combined result of the BOD loads from tributary runoff from the lowlands and the load from the shrimp farms. The relation between the locations of the shrimp farm loads (shown as arrows on Figure 22) and the region of low DO should especially be noted. Unfortunately, the data collection stations did

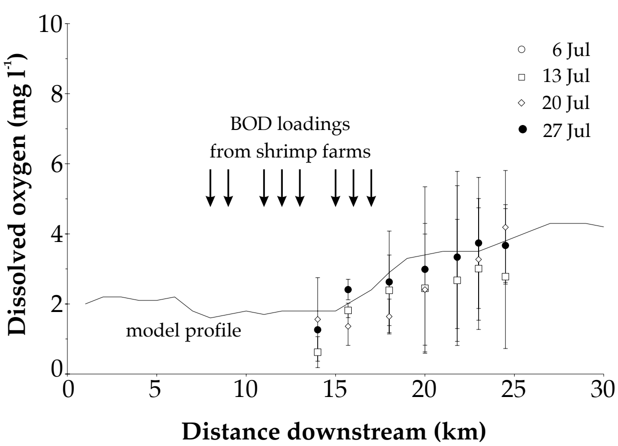

Figure 22. Model prediction and observations of DO in El Pedregal, July 1995.

not extend far enough up the estuary to separate the effects of the shrimp farms from the intrinsic poor water quality, but the model prediction of DO shows general agreement within the reach in which measurements are available. What is important is that all of the measured (vertical-mean) DOs upstream from about 13 km from the mouth are below 3 mg l^{-1} and some are less than 1 mg l^{-1}. The model results substantiate that these low DOs result in part from the BOD loads from the shrimp farms. These two facts indicate that the 1995 shrimp farms are already approaching the carrying capacity of El Pedregal.

The San Bernardo is a deeper system than the Pedregal in its lower reach and clearly received a greater volume of freshwater throughflow in 1995, because the salinities are lower and the salinity intrusion reach is much shorter (Figure 21). While the salinity model shows satisfactory agreement with the data, this is somewhat misleading. Most of the modeled reach has near-zero values, and the saline-intrusion reach near the mouth is also dominated by the boundary condition. There could still be substantial error in the transport terms that would not degrade the comparison of model and data. In particular, there is much uncertainty in the specified inflow. The method of estimating runoff (hence tributary flow) from the precipitation record at two stations 100 km distant is clearly an optimistic procedure. But an additional problem is the inflow of the Río Negro, which enters the head of the San Bernardo from its watershed in Nicaragua. This appears to be a major source of inflow for San Bernardo; moreover, it is evidently water of high oxygen content, originating in a mountainous basin in Nicaragua. Within the scope of this work, we were unable to determine its watershed area or to locate suitable precipitation data to use in estimating flow. For preliminary model runs, we assigned a value of 25 $m^3 s^{-1}$ with an oxygen concentration of 5 mg l^{-1} to this inflow, which is (at best) a guess. As matters developed, this flow is probably a substantial underestimate.

That this inflow is likely to be a major source of error is indicated by the comparison of model-predicted DOs to measurements (Figure 23). Although the model does a fair job of predicting a DO depression like that depicted in the data, the location of the low point of the sag is some 5 to 10 km too far up the estuary. This suggests that the throughflow is greater than that assumed in the model runs. As an example of how this inflow can control the location of the sag, a second model run was made with Río Negro flow increased by 60 $m^3 s^{-1}$, shown as the upper curve in Figure 23.

While we could re-adjust our estimated inflow in this manner to force the model to replicate the observations, we feel that it is more useful to operate the model with the best independent estimates possible, then display the shortcomings of the model prediction as evidence of the need for additional refinements. From this model run, however, we can conclude that the San Bernardo is not as seriously stressed as the Pedregal under these inflow conditions. However, this estuary received a total BOD load in 1995 of approximately 24,036 kg d^{-1}, nearly twice that of the Pedregal. It seems certain that high inflows during August 1995 prevented the occurrence of depressed DOs.

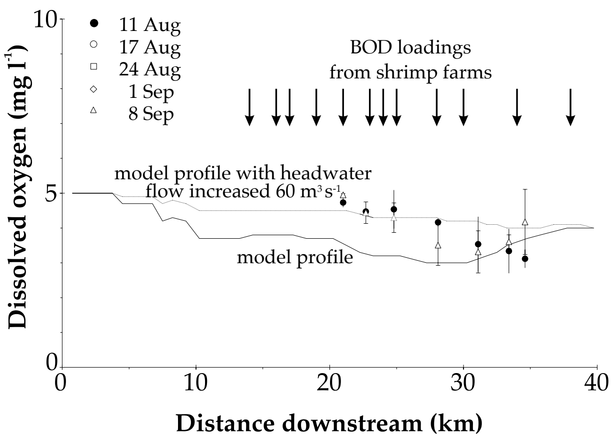

Figure 23. Model prediction and observations of DO in San Bernardo, August–September 1995.

From a salinity standpoint, modeling the dry-season regime is not very interesting. With the diminishment of freshwater inflows, salinity intrudes up the estuaries, attaining values equal to those in Monypenny Bay. The dry-season data for the reaches sampled show vertically and horizontally uniform salinities in both El Pedregal and San Bernardo (Figure 24). This is no challenge to a model because the salinities are equal to the boundary condition, so setting the boundary condition also establishes the salinities in the estuary.

For DOs it is a different matter. The measured tidal-mean vertical-mean DOs and the corresponding model prediction are shown in Figure 25 for El Pedregal. (As before, the vertical bars about the data

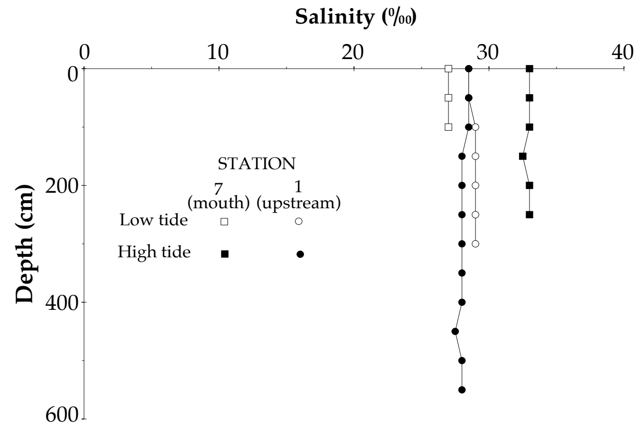

Figure 24. Salinity profiles in El Pedregal, 11 March 1996.

points are the excursion in DO at that station from low tide to high tide.) The DOs are somewhat lower than those measured (and predicted) in the rainy season, with DOs upstream from the first 10 km from the mouth much too low to ensure healthy aerobic communities. The immediate conclusion is that the 1995 level of shrimp farm development had already exceeded the carrying capacity for this estuary under these inflow conditions.

For the San Bernardo, again the uncertainty in the headwater flow of the Río Negro is problematic (Figure 26). With a zero headwater flow value, the model prediction is for an anoxic reach of 15-km length, which does not agree with the measurements. On the other hand, an assumed headwater flow of 15 $m^3 s^{-1}$ produces a DO profile closer to that observed (see the broken curve of Figure 26). (A higher headwater flow would improve even more the model comparison to data.) In any event, the observed DOs—regardless of the model prediction—demon-

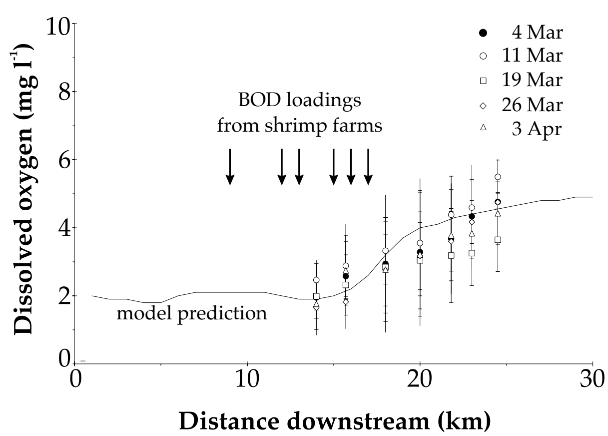

Figure 25. Model prediction and observations of DO in El Pedregal, March 1996.

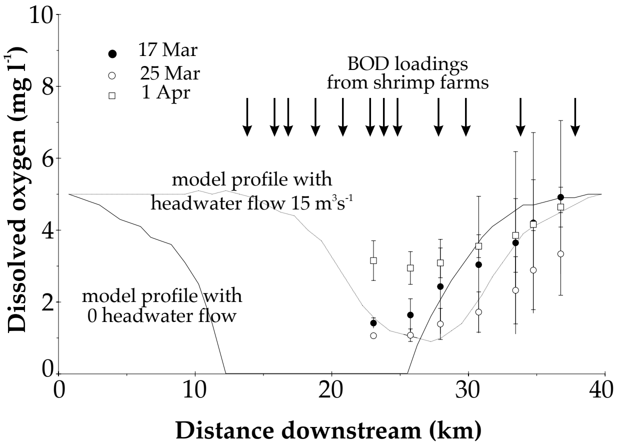

Figure 26. Model prediction and observations of DO in San Bernardo, March–April 1996.

strate that under these low-flow conditions, the 1995 farm loads (or at least farm management at the 1995 level of development) had already reached the carrying capacity of the San Bernardo during the dry season. Analyses of estuarine water quality variables over time demonstrated similar trends to those predicted by the DO models. Water quality variables increase in concentration upstream from the mouth of estuary and are higher during the dry season than the rainy season (Teichert-Coddington, 1995; Green et al., 1999).

Shrimp Farm Impacts and Carrying Capacity Analysis

A model allows insight into important features of an estuary by hypothetical "what if" scenarios. In an earlier study, Ward (1995) used an early version of the present water quality model and considered the problem of a single shrimp farm operating under dry-season conditions on the Estero la Jagua, selecting as an example Aquacultivos de Honduras. This farm drains into the Jagua about 1.5 km from its confluence with the Pedregal (or about 24 km from the head of the model estuary). Here the estuary channel is wide and deep; there is a large tidal prism and free exchange with the waters of the Gulf of Fonseca. The model showed the effect of the effluent on BOD or DO in the receiving water to be negligible. Using the model, the shrimp farm was then moved to points farther upstream, namely 4, 10, and 13 km. With distance upstream, the estuary cross section, the tidal prism, and the exchange (dispersion) all diminish, so the same BOD mass load was found to result in a higher BOD concentration. The effect is a greater impact on DO. In fact, the model showed that for any location farther upstream than 12 km, the DO would be driven to zero by this one shrimp farm. This experiment illustrated that the impact of a specific shrimp farm depends not only upon the effluent load of that farm but also where it is located within the estuary. This experiment also illustrated that a mass load in such a highly dispersive system as these river-channel estuaries affects water quality a great distance both downstream and upstream from the point of discharge.

The present model is a considerable improvement over that used by Ward (1995), not the least because its parameterizations are considered more accurate due to the existence of field data against which to validate the model. Moreover, the work carried out in Honduras by CRSP personnel over the past five years has greatly improved the quantification of the shrimp farm loads. However, the general conclusion from the hypothetical experiment, that moving a shrimp farm farther upstream will amplify its impact on estuary water quality, is still valid because the physical conditions contributing to that result are qualitatively the same. The present model development now permits a much more reliable estimation of the combined impacts of the regional shrimp farms under various scenarios of increased development. This section describes the model runs and the results for this problem configuration. In effect, this is a determination of the carrying capacity of the Monypenny Bay channel estuaries.

Information was compiled on shrimp farm enlargements and new operations as of 1998 (in contradistinction to the 1995 data of Table 1, pp. 10–11)—the additional pond area that could be developed on operating farms and the concessions capable of future development. This future "full development" scenario is summarized in Table 4. These drain locations and corresponding loads were implemented in the BOD-DO model using the same 1995 rainy-season conditions, so that the model runs (and associated field data) provided in the section Water Quality of the Channel Estuaries could be used as a baseline for comparison.

Several of these new or expanded loads do not directly affect the Pedregal and San Bernardo mainstem channels: BIOMAR drains directly to Monypenny Bay; INEXA, CRIMASA, and part of CAMARSUR discharge to the Estero de la Berberia; and BIMAR and EXMAR to Estero la Jagua. Also, the 400-ha projected expansion of

Table 4. Projected future development of shrimp farms.

Farm	Pond Area (ha)	Exchange (% d^{-1})	Throughflow (Mm^3 d^{-1})	BODu (mg l^{-1})	BOD Load (kg d^{-1})	BOD Load (lb d^{-1})
Acuacultura Fonseca	792	2	0.158	4	634	1,394
Aquacultivos de Honduras	600	8	0.48	16	7,680	16,896
BIMAR	160	5	0.08	10	800	1,760
BIOMAR	440	5	0.22	10	2,200	4,840
CADELPA	352	3	0.106	6	634	1,394
CAMARSUR*	750	10	0.75	20	15,000	33,000
CAYDESA	110	10	0.11	20	2,200	4,840
CRIMASA	1,568	6	0.941	12	11,290	24,837
CUMAR	600	3	0.18	6	1,080	2,376
EXMAR	149	8	0.119	16	1,907	4,196
GMSB	3,132	4	1.252	8	10,022	22,049
HONDUESPECIES	473	2	0.095	4	378	832
HONDUFARM	350	10	0.35	20	7,000	15,400
INEXA*	800	10	0.8	20	16,000	35,200
INMAR	238	12	0.286	24	6,854	15,080
La Jagua	320	2	0.064	4	256	563
PIONEROS*	100	10	0.1	20	2,000	4,400
PROMASUR	578	6	0.347	12	4,162	9,156
SURMA*	800	10	0.8	20	16,000	35,200

* Assumed area and exchange

GMSB toward Punta Guatales is assumed to drain directly to Monypenny Bay.

The resulting DO profiles under rainy-season conditions predicted by the model with these loads are shown in Figures 27 and 28 for El Pedregal and San Bernardo, respectively. Since the 1995 model results and field data indicate that the San Bernardo is near its carrying capacity and the carrying capacity of the Pedregal has been reached, the effect of further increased farm loads could be anticipated: even lower DOs in the main sag zones. For El Pedregal this looks particularly crucial, as a reach of some 10 km results with average DOs less than 1 mg l^{-1}. For the San Bernardo there is a reach with DO below 2 mg l^{-1} (considered unacceptable), but with a higher assumed flow in the Río Negro this reach would not occur (see the upper curve on Figure 28). Under low-flow, dry-weather conditions (not shown here), the model indicates that both estuaries exhibit anoxic reaches.

The data and the model suggest that under these rainy-season conditions, well-aerated waters are brought into the channel estuaries from the adjacent Gulf of Fonseca on the flooding tide and poorly oxygenated waters enter the estuary channels from the tidal flats and tributaries draining the tidal lowlands. Even if shrimp farms did not exist, some degraded water quality would result as a result of the latter intermixing with the former. This raises the

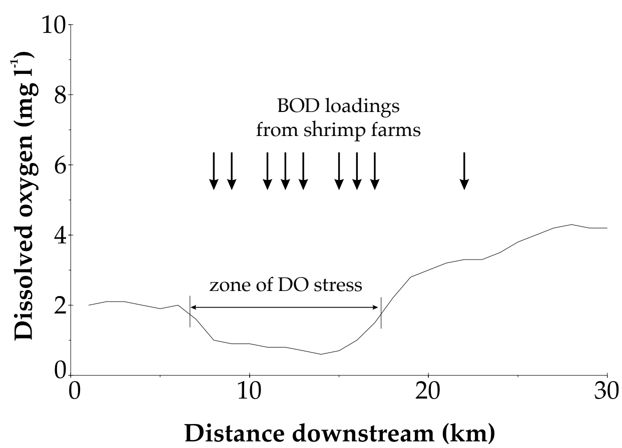

Figure 27. Model prediction of DO in El Pedregal, 1995 rainy-season conditions, full shrimp farm development.

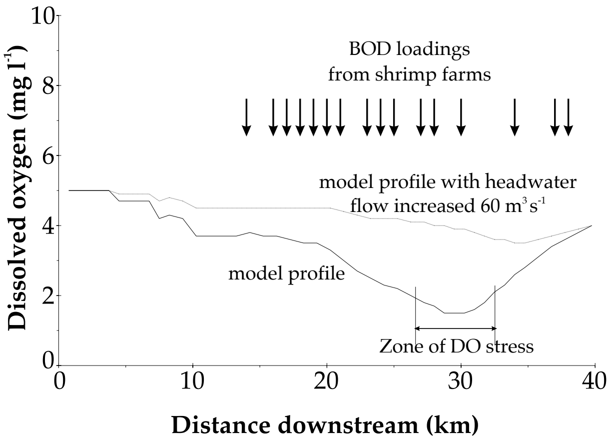

Figure 28. Model prediction of DO in San Bernardo, 1995 rainy-season conditions, full shrimp farm development.

question of how much additional degradation is contributed by the shrimp farms or, equivalently, what the DO profile would look like if the shrimp farms were removed. Model results to evaluate this "what if" scenario are shown in Figure 29, which compares the DO profile from the 1995 shrimp loadings to that resulting from complete removal of these loads. The fact that the DO concentration is shown to be higher in the reach from km 13 to km 22 than in the adjacent Gulf of Fonseca indicates that probably either reaeration is too high or background BOD is too low within this reach. (This suggests where additional model validation work is warranted.) Apart from this anomaly, a comparison of these two curves implies that the DO in the Pedregal would not be much over 4 mg l^{-1} under "natural" conditions. The net effect of shrimp farms at their present level is to reduce DO by about another 2 mg l^{-1}.

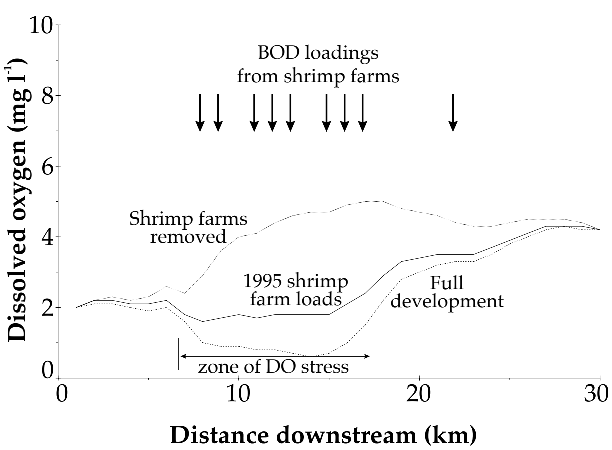

Figure 29. Model prediction of DO in El Pedregal, 1995 rainy-season conditions.

Conclusions and Recommendations

Two principal conclusions follow from this study. The first is that the hydrographic conditions in the regions in which shrimp farming operates are at least as important as chemical quality in determining the suitability of the water for influent supply. These hydrographic conditions include:

- tidal range and period;
- freshwater throughflow;
- physiography and morphology;
- tidal currents and parameters determined from the currents (such as excursion and prism);
- mixing and dispersion; and
- salinity—especially gradients of salinity.

Many parameters crucial to the economical operation of shrimp farms are affected or controlled by hydrographic parameters, including turbidity and suspended sediments, dissolved oxygen, nutrients, toxicants, and pathogens.

The second conclusion is that for systems such as Estero El Pedregal and San Bernardo—which are typical of many of the river-channel estuaries within the larger Gulf of Fonseca along which shrimp farming is being developed—hydrographic features dictate the effect of shrimp pond operations on their water quality, and a level of development exists at which the estuary can become so degraded as to prohibit economical aquaculture, i.e., shrimp farming can become self-limiting.

It could be presumed these conclusions were possible even before the present study was initiated, based upon the hydrographic nature of the system and knowledge of the physical limitations on assimilative capacity. What the present study has sought to accomplish is the quantification of these conclusions.

Ward (1995) reported a preliminary analysis of carrying capacity for this region, including the development of hydrodynamic and water quality models. At that time there were virtually no data available other than occasional water samples from the influent of shrimp farm intake pumps. The present work employs much better information on the morphology and water depths in the estuaries, on hydroclimatology, and on the configurations and BOD loads of shrimp farms. It also uses a base of field data to better characterize water quality in these estuaries and validate the model applications. While there remain significant uncertainties in the information base, the level of confidence that can be placed in our knowledge of these estuaries is much greater, thanks primarily to the data collection efforts in Honduras of the past several years.

This study has taken a rather narrow view of the problem of estimating carrying capacity by confining its attention to a single water quality parameter, DO, in only two of the channel estuaries of Monypenny Bay. There are other water quality parameters which in high concentrations could limit the use of the water as influent to ponds or negatively impact the biota in the estuaries, including nutrients (nitrogen and phosphorus especially) and toxics.

The probable impacts of such low DOs as indicated by this work are reduced oxygen in the influent water—potentially increasing low-DO stress in the shrimp ponds themselves—and elimination or diminished occurrence of aerobic organisms in the estuary. Impacts will of course be greater for the less mobile species in the estuaries, especially quasi-planktonic fingerlings and post-larvae. Low-DO sags, such as those shown in Figures 27 and 28, also represent a barrier to movement of aerobic species past the sag zone.

This report has not sought to identify DO impacts specific to individual farms nor to rate the relative impacts of the various farms, though these could certainly be accomplished using the mathematical model. The importance of effluent location on the impact of a specific farm is, however, a potentially major factor. In general, the smaller the physical dimensions of an estuary channel, the smaller the tidal excursion, and the more remote an area is from the mouth of the estuary, the greater the impact on DO that will result from an organic load placed in that area. Again, this qualitative statement could be made a priori about any channel estuary and any type of wasteload. The work reported here has quantified this statement for the Pedregal and San Bernardo, and it can now be asserted that farms placed farther up-stream than about 15 km from the mouth will most likely have excessive impact on the DO in the estuary. The impact is exacerbated under dry-season conditions. Similar conclusions were stated by Teichert-Coddington (1995) after evaluating water chemistry along the lengths of these same estuaries during wet and dry seasons. Negative impacts of a specific farm can be ameliorated by reducing or eliminating pond water exchange during the dry season.

The estuaries are flushed a couple of times a year during periods of heavy rainfall when runoff enters the headwaters of estuaries in high quantities. During these periods, the salinity drops to zero throughout the estuary indicating that the entire body of water has been displaced. This annual cleansing ameliorates the impact of farm discharge on the estuary. As long as rainfall is normal, the results of the work reported here are dependable, but effects of farm loading on the estuaries could become worse if rainfall were abnormally low. It has been the experience of CRSP researchers that every year since the CRSP water quality monitoring study began, nutrient concentrations increased during the dry season and decreased during the rainy season as the estuaries are flushed out. There is a high probability for water quality problems to develop during the dry season (which may have been controlled by ANDAHs interest in protecting these estuaries through regulated, managed development of the industry). It may be necessary for farms placed farther upstream on the estuaries to cease operation during the dry season—a practice already observed by several farms—and farms lower down on the estuary to reduce exchange and nutrient use. But additional field data, especially from the dry season, are needed to better quantify these suggestions.

In this context, annual precipitation data in Figure 4 (p. 7) shows that 1995 had the highest rainfall since establishment of the shrimp farming industry in the region (and the highest in the data record available to this study, extending back to 1972). The fact that the field sampling (and the above modeling) took place under such exceptional conditions means that the impacts of shrimp farms were tempered in at least two respects. First, the higher throughflows more effectively diluted the effluent and flushed the estuaries. Second, the relative importance of ambient loads (in runoff from the watershed) versus shrimp farm loads was maximized. The effect of the high 1995 rainfall may even have extended into the subsequent dry season, at least in the flow of the Río Negro in the San Bernardo.

The issue of carrying capacity determination in this region is governed by temporal and spatial scales. With respect to the former, emphasis in this study was placed on long-term transports integrated over many tidal cycles. This was implicit in the use of a steady-state model to address the vertical-mean tidal-mean data. With respect to the tidal-mean data, there are (at least) three operative scales of spatial reference, indicated schematically in Figure 30. The first is internal to the channel estuaries. This is concerned with the immediate impact of farm development on the receiving estuaries. This is the space scale addressed in the work reported here. The basic assumption is that the boundary at the mouth of the estuary is fixed and unaffected by shrimp farm operations. That is, the impacts on each estuary, e.g., El Pedregal, are independent of the other channel estuaries, e.g., San Bernardo.

The second scale of reference includes the larger area of Monypenny Bay, in which the open waters of the Gulf of Fonseca now become the fixed boundary condition. At this scale, the tidal exchange between each channel estuary and Monypenny Bay is explicitly considered, and the degree to which the effluents from all of the channel estuaries intermix to affect the adjacent estuaries must be determined. It is at this scale, also, that the potential impact of shrimp farm development on the Estero Real would exert an impact on the carrying capacity of the Honduran estuaries.

It is not clear how the tidal prisms of El Pedregal and San Bernardo are diluted and mixed at low tide before re-entering the estuaries. The morphology of Monypenny Bay suggests that most of the tidal exchange occurs up the Estero Real, in which case the waters out from the Honduran estuaries may be a poorly exchanging tidal-spun vortex, as indicated in Figure 30. The presence of supersaturated DOs in the Monypenny Bay waters out from the mouth of El Pedregal is particularly significant (Figure 16, p. 20). There are two mechanisms by which such high levels of supersaturation could occur. One is mechanical, due to wave overtopping and entrainment, driven by winds over open water. The other is photosynthesis, stimulated by high concentrations of nutrients. The former requires sustained winds into the study area over a long over-water fetch. The latter requires the accumulation of nutrients in the water column with little dilution. The former would imply vigorous circulation and dilution with waters of the open Gulf of Fonseca. The latter would imply isolation of the waters out from the Pedregal mouth and poor exchange and dilution with the open waters of the Gulf (which

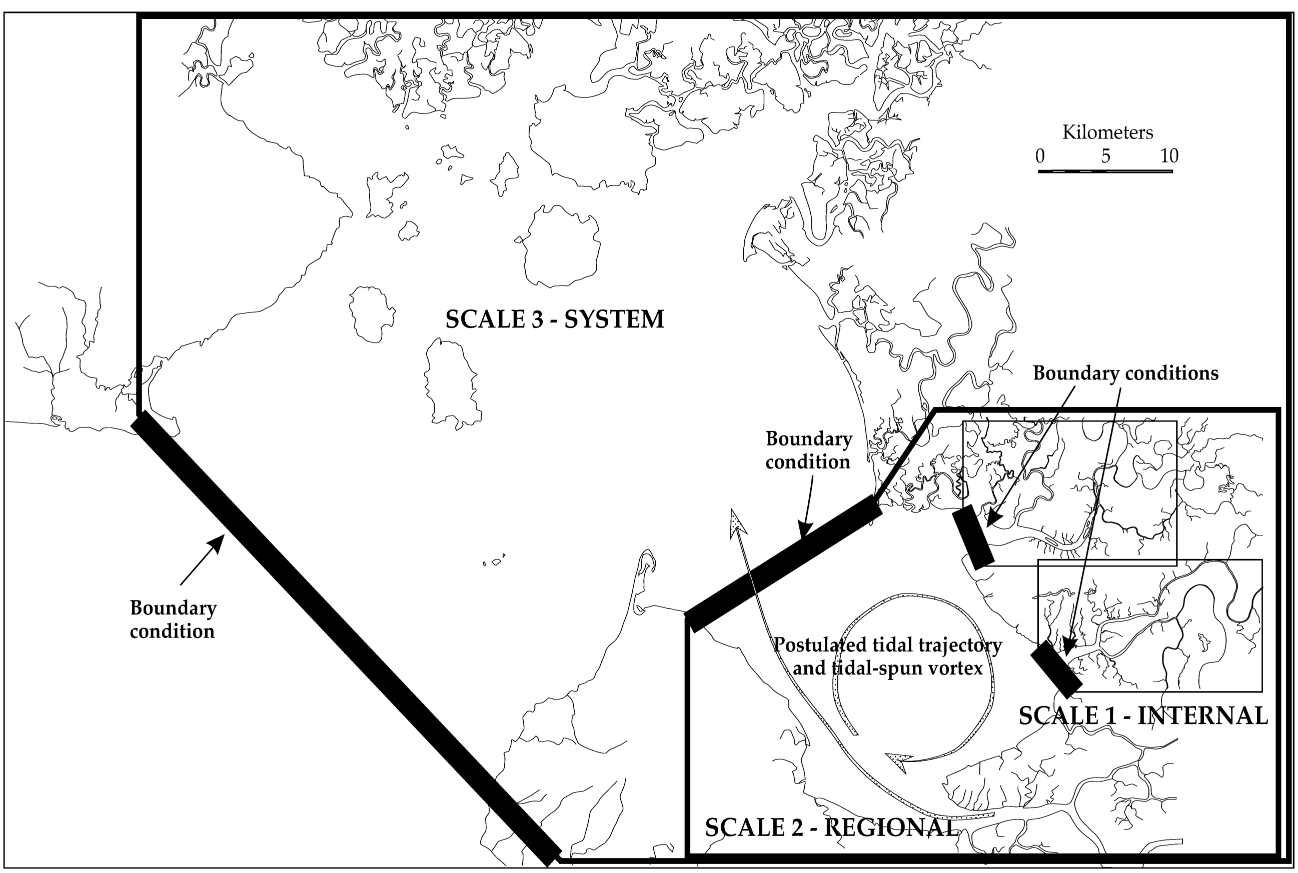

Figure 30. Spatial scales of analysis and relation to Fonseca system.

would be the result of a tidal-spun vortex such as that suggested in Figure 30).

The third and largest scale is the entire Gulf of Fonseca, for which the Pacific entrance becomes the boundary condition. At this scale, the tidal exchange between Monypenny Bay and the open waters of the Gulf determines the level of concentrations within Monypenny Bay, thence within the channel estuaries.

In order to address these larger-scale concerns, data are needed on the physical and chemical properties of waters within the open waters of the Gulf of Fonseca, and modeling will have to incorporate two (and perhaps three) dimensions. We consider the second scale of reference to be particularly crucial in evaluating the carrying capacity for shrimp farming and urge that future research be directed toward this end.

In the previous work, reported in Ward (1995), it was commented that:

> The great weakness remains our data base on the main body of the Golfo de Fonseca, which is practically zero… Salinity / temperature / DO profiles need to be carried out on some regular basis at a network of stations in the Gulf, supplemented by occasional water sampling… The potential return on this investment, in ensuring the continued economic viability of shrimp farming in this area, is, however, huge. The important point is that a resource of historical water quality data is indispensable to evaluate operational problems and devise appropriate management strategies. No mathematical can compensate for a lack of data. Once a problem develops and immediate action is required, it is too late to begin collecting data.

Unfortunately, almost half a decade has passed, and the situation of lacking these basic data on the Gulf of Fonseca is practically unchanged.

Acknowledgments

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Literature Cited

Admiralty Hydrographic Office, 1951. West Coasts of Central America and United States Pilot, Sixth Edition. Her Majesty's Stationary Office, London, 539 pp.

Chang, J. and L. Lau, 1983. Definition of the humid tropics. In: M. Bonell, M. Hufschmidt, and J. Gladwell (Editors), Hydrology and Water Management in the Humid Tropics. Cambridge University Press, Cambridge, pp. 571–575.

Currie, D.J., 1994. Ordenamiento de camaronicultura, Estero Real, Nicaragua. Reporte de Estudio. GOPA Consultants, Alemania.

Dronkers, J.J., 1964. Tidal Computations in Rivers and Coastal Waters. North-Holland Publishing Co., Amsterdam, 518 pp.

Flores-Verdugo, F., F. González-Farías, O. Ramírez-Flores, F. Amezcua-Linares, A. Yáñez-Arancibia, M. Alvarea-Rubio, and J. Day, 1990. Mangrove ecology, aquatic primary productivity, and fish community dynamics in the Teacapán-Agua Brava lagoon-estuarine system (Mexican Pacific). Estuaries, 13(2):219–230.

Green, B.W., D.R. Teichert-Coddington, M.P. Micheletti, and C.A. Lara, 1997. A collaborative project to monitor water quality of estuaries in the shrimp producing regions of Honduras. Proceedings of the IV Ecuadorian Aquaculture Symposium, 22–27 October 1997, Escuela Superior Politécnica del Litoral / Centro Nacional de Investigaciones Marinas, Camera Nacional de Acuacultura, Guayaquil, Ecuador, CD-ROM.

Green, B.W., D.R. Teichert-Coddington, C.E. Boyd, D. Martinez, and E. Ramírez, 1998. Estuarine water quality monitoring and estuarine carrying capacity. In: D. Burke, J. Baker, B. Goetze, D. Clair, and H. Egna (Editors), Fifteenth Annual Technical Report. Pond Dynamics / Aquaculture CRSP, Oregon State University, Corvallis, Oregon, pp. 87–98.

Green, B.W., D.R. Teichert-Coddington, C.E. Boyd, D. Martinez, and E. Ramírez, 1999. Estuarine water quality monitoring and estuarine carrying capacity. In: K. McElwee, D. Burke, M. Niles, and H. Egna (Editors), Sixteenth Annual Technical Report. Pond Dynamics / Aquaculture CRSP, Oregon State University, Corvallis, Oregon, pp. 103–113.

Hauck, L. and G. Ward, 1980. Hydrodynamic-Mass Transfer Model of Deltaic Systems. In: P. Hamilton and K. Macdonald (Editors), Estuarine and Wetland Processes. Plenum Press, New York, pp. 247–248.

Instituto Geográfico Nacional, 1986. Guía para Investigadores de Honduras, 2a Ed. Secretaría de Comunicaciones, Obras Públicas y Transporte, Tegucigalpa, D.C., Honduras, 104 pp.

Larsen, M. and I. Concepción, 1998. Water budgets of forested and agriculturally developed watersheds in Puerto Rico. In: R. Segarra-García (Editor), Tropical Hydrology and Caribbean Water Resources. American Water Resources Association, Herndon, Virginia, pp. 199–204.

O'Connor, D. and W. Dobbins, 1958. Mechanism of reaeration in natural streams. Trans. Am. Soc. Civ. Eng., 123:641–684.

Officer, C.B., 1976. Physical Oceanography of Estuaries (and Associated Coastal Waters). John Wiley & Sons, New York, 465 pp.

Teichert-Coddington, D.R., 1995. Estuarine water quality and sustainable shrimp culture in Honduras. In: C.L. Browdy and J.S. Hopkins (Editors), Swimming through Troubled Water. Proceedings of the Special Session on Shrimp Farming, Aquaculture '95. World Aquaculture Society, Baton Rouge, Louisiana, pp. 144–156.

Teichert-Coddington, D., L. Milla, D. Martinez, and G. Ward. Relationships between biochemical oxygen demand and selected water quality variables in shrimp farming water, in preparation.

Teichert-Coddington, D.R., B. Green, C.E. Boyd, D. Martinez, and E. Ramírez, 1997. Estuarine water quality. In: D. Burke, B. Goetze, D. Clair, and H. Egna (Editors), Fourteenth Annual Technical Report. Pond Dynamics/Aquaculture CRSP, Oregon State University, Corvallis, Oregon, pp. 87–89.

Thomann, R. and J. Mueller, 1987. Principles of Surface Water Quality Modeling and Control. Harper and Row, New York, 644 pp.

Wainwright, F., 1996. Cuantificación de areas de esteros que drenan al Golfo y que pertenecen a la cuenca del Río Negro. Departamento de Choluteca. Manuscript with maps (unpubl.).

Ward, G., 1995. Hydrographic limits to shrimp aquaculture in el Golfo de Fonseca. III Simposio Centroamericano sobre Camarón Cultivado, Tegucigalpa, Honduras.

Ward, G. Relationships between biochemical oxygen demand and selected water quality variables in shrimp farming water, in preparation.

Ward, G. and W. Espey, 1971. Estuarine modeling: An assessment. Report 16070 DZV 02/71, Environmental Protection Agency, Washington, D.C., 497 pp.

Ward, G. and C. Montague, 1996. Estuaries. In: L. Mays (Editor), Water Resources Handbook. McGraw-Hill Book Company, New York, pp. 12.1–12.114.

Wolanski, E., 1993. Hydrodynamics of mangrove swamps and their coastal waters. Hydrobiol., 247:141–161.

Wyrtki, K., 1966. Oceanography of the eastern equatorial Pacific Ocean. In: H. Barnes (Editor), Oceanogr. Mar. Biol. Ann. Rev., 4:33–68.

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