Final Report
Near Coastal Waters Program

Guidelines for Water Resources Permitting:
Nutrient Requirements for Maintenance
of Galveston Bay Productivity

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Preface

The work described here was supported by the Texas Water Development Board (TWDB) and in by the USEPA. The guiding purpose was to provide new data and a new synthesis of this new and existing data to aid management of Galveston Bay, Trinity-San Jacinto Estuary, resources. The approaches presented here are also applicable to other Texas bays.

Section I presents nitrogen and phosphorus budgets for Galveston Bay during low inflow and high inflow periods. The section ends with proposals of specific minimum nitrogen and phosphorus requirements to maintain characteristic productivity of Galveston Bay. These are evaluated through budget analysis and reference to historic conditions.

Section II is an analysis of community production and respiration in Galveston Bay from application of the diurnal curve technique to TWDB Datasonde dissolved oxygen data. Derivation of nutrient requirements from the results is explored, through comparison of demands from primary producers with supply from regeneration and from outside sources, .

Section III presents a STELLA model based on quantification of relationships between freshwater inflow and components of the Galveston Bay nitrogen budget. The model is used to test the sensitivity of the system nitrogen dynamics to assumptions and uncertainty in the data. The model also shows system-level outcome of components variously responding to changes, such as reductions in freshwater inflows.

Every effort was made to conduct analyses in an objective fashion and to clearly indicate where situations required professional judgment to substitute for missing information. Where analyses described below involve assumptions and professional judgment, these reflect the opinions of the authors, and may not reflect the official positions of the supporting agencies.

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Acknowledgments

Section I.   Nutrient Budgets for the Galveston Bay System

Introduction

This section presents annual budgets of water, total dissolved solids, nitrogen and phosphorus for the Galveston Bay system of Texas.   A major feature of this report is discussion of assumptions involved in evaluations of contributing processes.   The purpose of the study is to provide information for resource management.

Dissolved nutrients available to estuarine primary producers are determined by processes which remove nitrogen as well as by rates of inputs.   The Trinity-San Jacinto Estuary receives nutrients from stream flows, wastewater, rain falling directly on its waters, and from biological fixation of atmospheric nitrogen.   The system loses nutrients in outflows to the Gulf of Mexico, through the biological process of denitrification, by burial in deep sediments, and through the harvest and migration of fish and other animals which grow (accumulate biomass) in the estuary.   It is useful to bring together sources and sinks of a nutrient in a budget. The balance of the budget, positive or negative, is a positive or negative contribution to resources driving ecological productivity.   Given that the estuary does not undergo qualitative changes, the way in which alterations to inflows affect this balance is a realistic means of judging the impacts of those alterations.

The nutrient budget process provides a basis for assessing potential affects of system alterations in the context of all sources and sinks.   A balance of inputs and losses also tests our knowledge of the system.   If the amount of materials which cannot be accounted for is large, then important sources, sinks, or processes have been overlooked. By quantifying budget components, the relative importance of physical, geochemical, and biological processes are revealed, and the comparative magnitudes of components may be related to sensitivity of the system to changes in those components.

Complications introduced by tidal movements of water and materials present a challenge for development of materials balances for estuaries.   Studies described in Correll et al. (1992) use intensive sampling of concentrations and flows at a major pass within an estuary to determine materials fluxes. Nixon and Pilson (1984) present an innovative approach based on nutrient inputs, nitrogen to phosphorus ratios, and system metabolism to derive nitrogen and phosphorus budgets for Narragansett Bay.

In the procedure described below, results of circulation modeling are used to

describe net and tidal exchanges of the Trinity-San Jacinto Estuary with neighboring waters.

The Trinity-San Jacinto Estuary, referred to here as the Galveston Bay system, includes tidal portions of tributary streams. Basic parameters describing the system are presented in **Table 1**. The estuary is treated as one unit in the budget, without distinguishing important transfers between internal compartments, such as marshes and open water. This whole system view is presented diagrammatically in **Figure 1**.

Budgets were developed for three years, so that results could be related to the differences in annual freshwater inflows. The year 1988 is a year of low freshwater inflows, approximately 86% exceedence combined inflows. That is, the volume of combined inflows from runoff, streams, and wastewater in 1988 is exceeded in 86% of historical annual periods. Flows during 1990 were very high, approximately 12% exceedence. These two years are the primary target years for establishment of a flow relationship. Data for the year 1989 were developed also, as a check on the linearity of the relationship. Inflows during 1989 were moderately high, 30% exceedence.

Methods, Data, Assumptions

Procedures

There is a hierarchy of calculations in the creation of materials budgets, with budgets of materials based on the budget of water, and a budget of total dissolved solids (TDS) used to adjust components of water exchange to bring about a balance of conservative materials. Various aspects which pertain to one or more of these steps are presented in following paragraphs.

Materials Inputs, Loading

Loadings of TDS, total nitrogen (TN) and total phosphorus (TP) from the drainage basin are calculated from measured and simulated stream-flows and concentrations of these materials in those waters. Loadings were compiled monthly, to incorporate seasonal variation in inflows and in concentrations of materials in the tributaries. The source of stream-flow and other hydrologic data is Texas Water Development Board (TWDB) databases compiled as part of the studies reported in

Table 1. Galveston Bay parameters.

Gaged drainage basin area	53973 KM^2
Ungaged drainage basin area	6624 KM^2
Total drainage basin area	60597 KM^2
Area	1391·10^{6}M^2
Volume (V)	2409·10^{6}M^3
Median freshwater inflow (net inflow,Q)	12590·10^{6}M^3
Hydraulic residence time*	0.08 y
Average depth	1.73 M
Average salinity (volume weighted)	16.3
Freshness**	0.43

*$t = f \cdot V/Q$
** $f = (Sg/Sb)/Sg$ where Sg is Gulf salinity and Sb is bay salinity

Figure 1. Estuary Nitrogen Budget Components
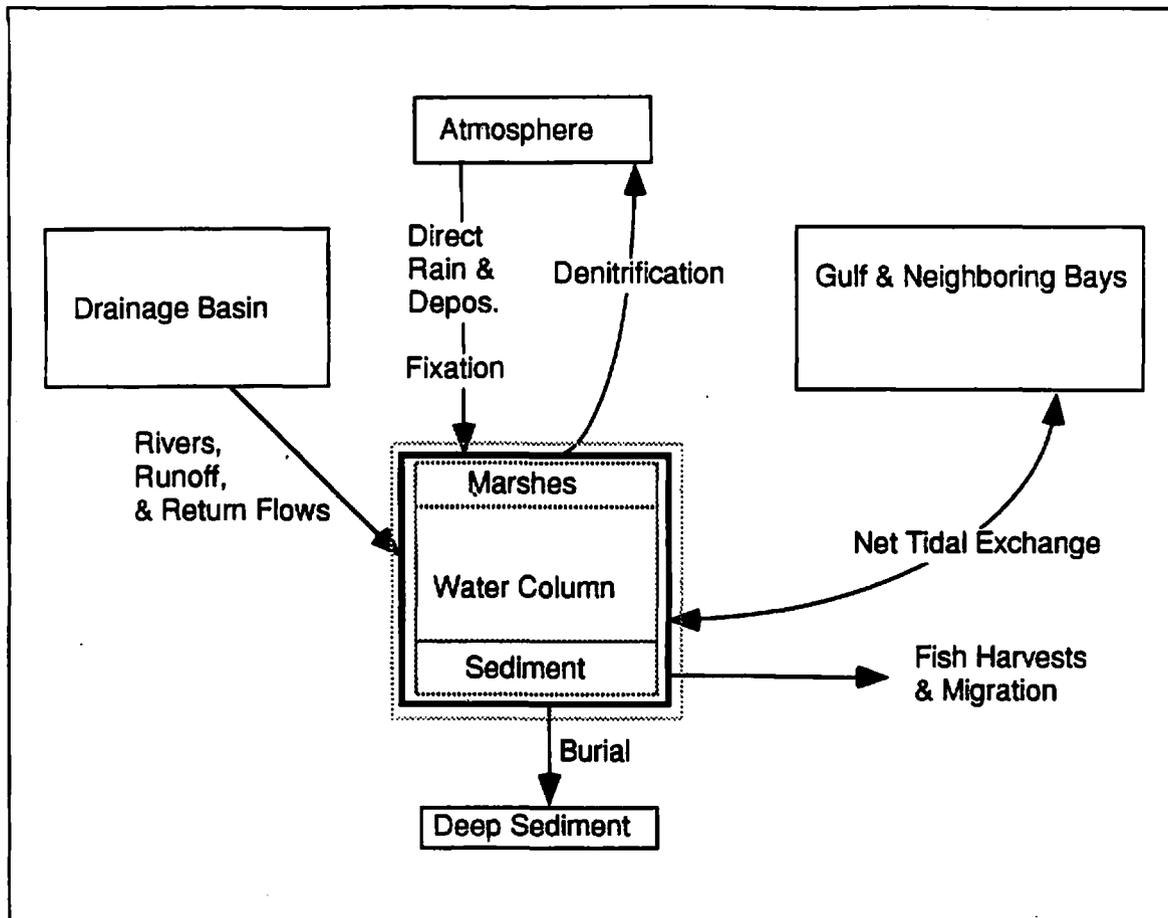
Water Balance:

0 = Fresh Inflows + Direct Rain + Net Flows from Neighbor Bays
    - Evaporation - Net Flows to Gulf - Net Flows to Neighbor Bays

Materials Balance:

0 = Fresh Inflow Dissolved and Suspended + Direct Rain & Deposition
    + Tidal Entrainment + Net Flows from Neighbor Bays + Fixation
    - Net Outflow to Gulf - Net Flow to Neighbor Bays - Tidal Entrainment
    -Denitrification - Fisheries Harvests - Burial
    +/- Storage

Longley (1994).   Gaged stream-flow, ungaged rainfall runoff, diversions, and wastewater return flow volumes were calculated for each coastal watershed (**Figure 2**).

Stream-flows: Inputs of materials from gaged stream-flow and ungaged runoff take into account reductions of flow through diversions.   Both gaged stream-flows and ungaged rainfall runoff loadings were calculated as concentration-flow, with average concentrations for each month and each month's total flow volume. Concentrations were compiled from data in USGS and Texas Natural Resource Conservation Commission (TNRCC) databases and routine monitoring programs.   TDS concentrations were often measured (residue dried at 180 $C°$), but often had to be estimated from specific conductance ($65.1197+0.5009 \cdot C$, $r^2 = .73$).   Total nitrogen is defined as nitrate-N + nitrite-N + total Kjeldahl N (TKN).   Total nitrogen was estimated from dissolved inorganic nitrogen in cases for which TKN was unavailable, based on regression ($TN=1.8446+0.8239 \cdot DIN$, $r^2 = .37$), or from ammonium and/or nitrate as data were available.   Approximately one-half of total nitrogen concentrations were estimated for inflow sources.   Total phosphorus was commonly measured, but was occasionally estimated from ortho-phosphate concentrations.

Values of nitrogen and phosphorus species which were reported as less than measurement detection limits were assigned a value, usually approximately half or less than the detection threshold. These assigned values were the averages of values reported below those thresholds for each threshold.   This is possible as thresholds have changed with changes in methodology.   Threshold-concentration comparisons were based on work done in the Guadalupe estuary involving data found in the Texas Water Commission (now TNRCC) Statewide Monitoring Network (SMN) database for the basin, and involving water samples collected in Nueces and San Antonio Bays and analyzed by Whitledge (1990).

Materials concentrations were averaged by month for each year, for each station.   Missing month concentrations were estimated by interpolation.   For watersheds without concentration measurements , concentrations were used from neighboring watersheds with similar land use and size.   This includes watersheds with un-monitored streams and those without sizable streams.

Return Flows: For this estuary, the percentage of nitrogen entering the bay from wastewater is equivalent to that entering from major tributaries, and slightly more than that entering from non-point sources (Armstrong and Ward, 1993). Wastewater return flows include volumes from many different sources with widely

Figure 2. Coastal watersheds of the Trinity-San Jacinto Estuary.
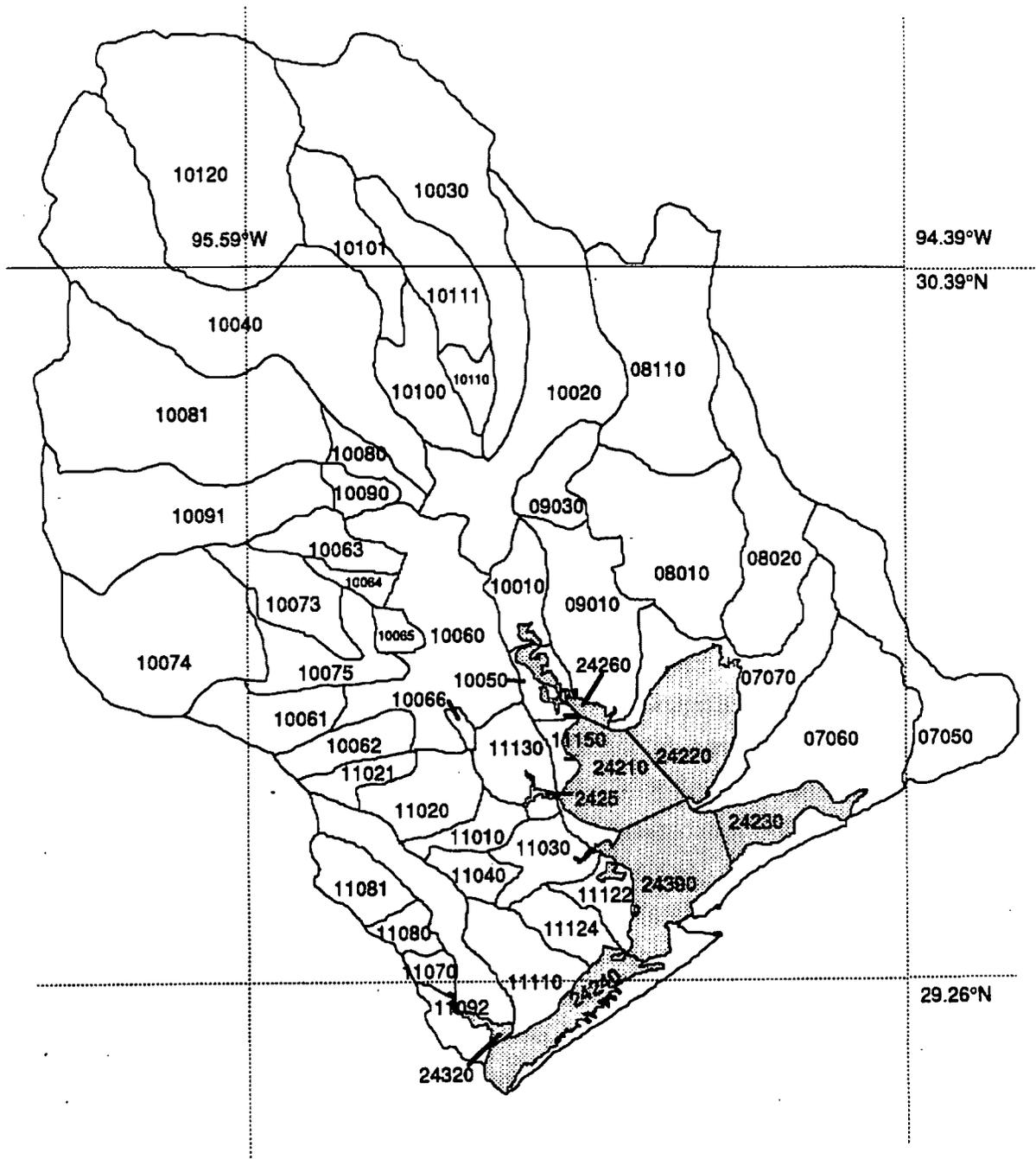
variant concentration profiles. Only those sources are considered which are not already included in gaged inflow volumes. Volumes of discharges are available through self-reporting records on file at the TNRCC. Concentrations reported as part of waste discharge permit compliance often do not, however, cover all species of dissolved nutrients which need to be considered in the budget. The TNRCC and its predecessor agency (TWC) has sampled many wastewater outfalls as part of special studies to determine the capacity of rivers and streams to process these wastes. To represent wastewater concentrations, trimmed averages were computed from data gathered during extensive sampling performed as part of the Houston Ship Channel study (Kirkpatrick, 1986). These average TDS, TN, and TP concentrations were 2989 mg/l, 11.7 mg/l, and 2.58 mg/l, respectively.

Rainfall and Evaporation: Contributions from direct rainfall to the bay are also included. TWDB hydrologic data include volumes of rainfall estimated from coastal NWS stations using a Thiessen network. Concentrations were obtained from the National Atmospheric Deposition Program (NADP, 1993) for the Attwater station, ca. 50 mi.(80 km) from the main bay. Loading was based on the combined concentrations of nitrate and ammonia nitrogen in NADP samples, averaged over 1988-1993 (0.41 mg/l). Monthly concentration data were available for the years studied; the average was used to better represent concentrations applicable over the broad estuary area. Evaporation from the bay surface affects the materials budgets indirectly through the water budget of the estuary.

Hydraulics, Water Budget

Net flows and tidal flows through the Gulf passes constitute the major loss of nitrogen from the system. Net outflows to the Gulf are the sum of freshwater inflows, minus evaporation, distributed by the model among the passes. Tidal volumes were obtained from detailed simulation of Galveston Bay circulation. Volumes flowing in from and out to the Gulf and neighboring bays were developed from results of the TxBLEND model (Matsumoto, 1994). TxBLEND simulates water movement in the two horizontal dimensions based on the finite element method, with triangular elements and linear basis functions. Water circulation calculations are based on conservation of mass and momentum, from input tidal elevations, freshwater inflows, precipitation, evaporation, and wind. Simulations were carried out for each of the three annual periods considered here, with inflow and outflow volumes compiled monthly from much smaller computational increments.

Although simulations included east-west flows in the ICWW and other channels from West Bay and East Bay, there were neither flow data for confirmation, nor concentration data to support complete inclusion of materials transport accompanying these flows in the budgets. "Place-holder" estimated contributions are shown in materials balance tables. These flows and materials fluxes are very small in relation to Gulf exchanges, tidal flows collectively 3% of those at Bolivar Roads.

In this application of TxBLEND to the Galveston Bay system, small net flow volumes could not be accurately accounted for against the overwhelming volume of tidal flow. Net outflows were distributed among the passes in the proportions obtained from results of the model run without tides. In these model runs, net outflows east and west in the ICWW and in Cold Pass could not be distinguished from 0.0.

Tidal Exchange in the Budget

A complete budget for a tidal water requires quantification of materials gained and lost through tidal exchange (eg. Nixon et al. 1994). During flood and ebb tides, mixing occurs along the front of contact between bay and Gulf waters. For material transfer, this is usually considered as a process of diffusion. Here, I treat this mixing as a process of entrainment, for clarity in visualizing and comparing quantities involved. Tidal mixing can be represented as entrainment of Gulf water in the bay and bay water in the Gulf, in addition to movement as net flows. For example, entrainment volume of incoming Gulf tidal water = entrainment rate · gross flood tidal volume.

For Corpus Christi Bay, Smith (1985) used a tidal "mixing rate" of 15%/h, giving 50% mixing of tidal inflows into bay waters on an average flood tide. Lockwood and Carothers (1967) suggested "mixing efficiencies" between 10% and 40% would be appropriate for inlets of the Texas coast. A 15% rate of entrainment is used as a base rate in these analyses.

As a rough check on tidal mixing rates for this estuary, a simple iterative model was used to simulate the recovery of TDS (volume weighted average) in the estuary (daily average salinities off Dollar Point) between January and June, 1988. Recovery was modeled first using an exponential equation:

$$TDS_i = TDS_{begin} + (TDS_{end} - TDS_{begin}) \cdot (1 - \exp(-k \cdot i)),$$

for each time step i, from beginning and ending values of TDS, and with a constant k. TDS recovery was then modeled through iterative additions of salt into the estuary volume through entrainment at several rates. The exercise shows (**Figure 3**) that 12% mixing reproduces the initial slope of recovery better than a higher rate, but 20% mixing achieves the ultimate recovery better than a lower rate. The 15% mixing rate was applied to tidal outflow at Bolivar Roads, a relatively high energy environment, promoting mixing. Another rate, for application to interior entrainment and mixing at smaller passes, was derived through iterative adjustment within the TDS budget calculations to bring the TDS balance close to 0. This procedure means that instead of evaluating the results of the TDS budget to determine water budget bias which would influence the nitrogen budget, the TDS data is assumed competent to correct water budget bias, and the correction is applied. The derived interior mixing rate was 13% for 1988, and 13% for 1989, and 12% in 1990.

Gulf Influence

Some of the water coming into the estuary on flood tides is water previously exhalent from the bay. Some proportion of the incoming water can be considered new water from the Gulf outside the estuary plume. Here, entrainment calculations are based on movement into the bay from offshore, beyond the bay outflow plume. Concentrations are chosen to be consistent with this. If there is a transport into the bay of nutrients derived from neighboring estuaries traveling in the immediate shoreline zone, this part of the Gulf contribution would be underestimated.

Bay and Gulf Concentration Data

Bay nitrogen and phosphorus concentrations were taken from the TWDB and TNRCC databases. TDS for bay areas was computed from salinities measured by TPWD during trawl sampling.

There are limited monitoring data for nutrients in Texas near-coastal shelf waters. TWDB and TWC have collected data near entrances of Gulf Passes, within the plume of estuary outwelling. To represent the Gulf proper, data were used from near-shore sites sampled in studies of the Bryan Mound ocean brine disposal site (Hann and Randall, 1981), from TPWD Gulf trawl salinity data (Dailey et al. 1991), from near-shore sites sampled in cruises of the Tx A&M RS Gyre (Biggs, 1988-1991) and from near-shore sites sampled during the Texas-Louisiana Shelf Circulation Program (LATEX) cruises (Jochens, 1994) above 60 meters depths. Gulf TDS levels were estimated from average monthly TPWD offshore trawl salinities from regression

Figure. 3 Salinity recovery, fit line vs recovery calculated
using trial entrainment rates, Dollar Point.
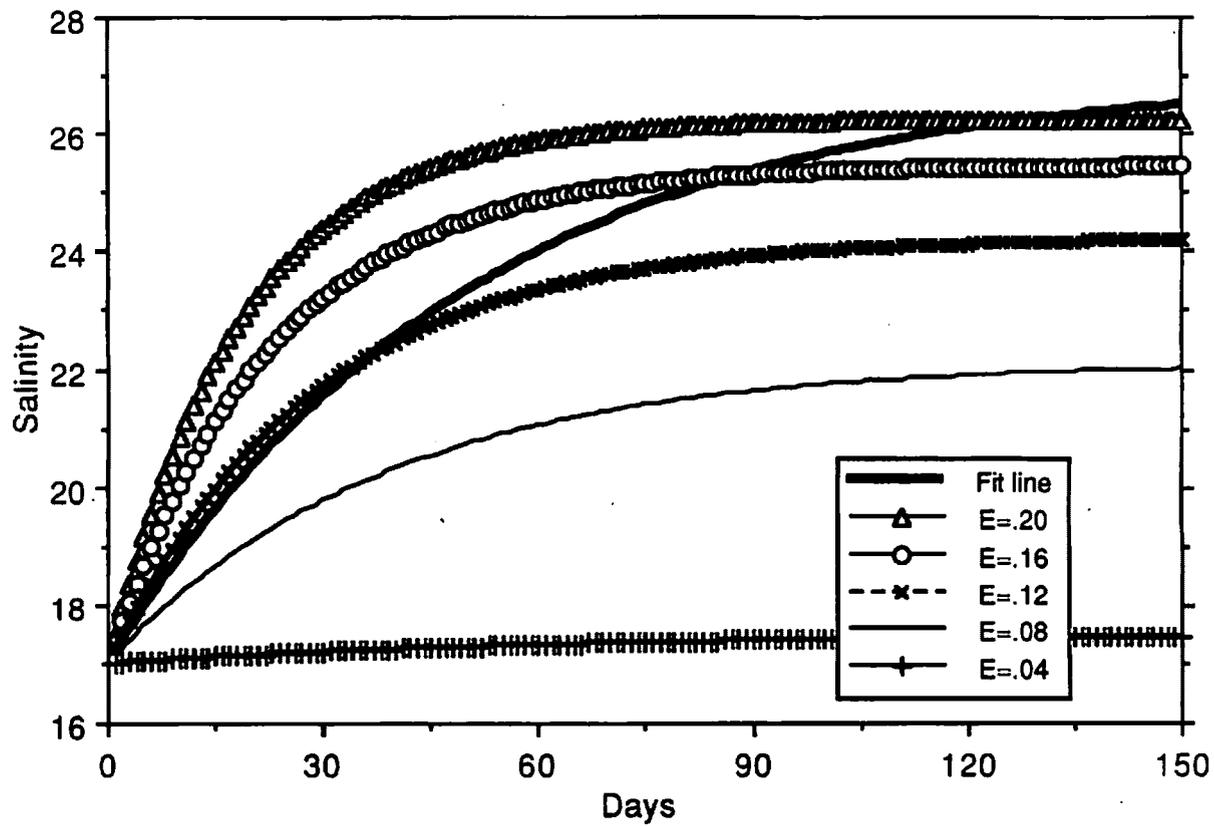
(TDS=1598 + 940.14·S, R2=.87, based on Galveston Bay TWDB data). Gulf nitrogen measurements were predominantly of dissolved inorganic species (DIN). Total nitrogen was estimated from DIN based on the average ratio of TN/DIN=6.94 in samples at the Galveston entrance jetties. Average nitrogen concentrations from the three Gulf studies were very similar, TN=.06-.07 mg/l. Average total phosphorus measurements varied from .004-.01 mg/l among the studies. Because the cruise data did not cover monthly variation and few dates were available to represent the years of interest, I used grand average concentrations for Gulf N and P. In the budgets, N and P values were based on LATEX data.

Denitrification

Denitrification results in loss of biologically available nitrogen from the estuary as nitrogen gas, N_2. This process typically takes place in anoxic environments, such as are found in the sediments. Denitrification rates were measured in Galveston Bay during 1993 (Zimmerman and Benner, 1993). From these data, monthly denitrification rates were estimated from long-term average monthly water temperatures using a regression relationship based on the seasonally collected samples (**Table 2**). Denitrification measurements were made from cores collected at four upper and mid-bay sites. The sediments at these sites did not encompass all major sediment types in the bay, particularly not including sandy lagunal sediments. Therefore, a step was taken to adjust the sediment area which the denitrification measurements could be assumed to represent. Rates were prorated based on sediment total organic carbon (TOC) concentrations, with the upper bay measured rate applied directly to sediments with TOC > 1.0 mg/kg (**Table 3**). The aerial distribution of sediment organic carbon was obtained from samples collected on a mile-square grid by White et al.(1985). Denitrification rates would also be expected to vary with rate of delivery of nitrogen to the sediment--and by inference, to TN loading-- at some time scale, but data on that relationship is not available.

Burial

Burial in bay sediments removes some nutrients from cycling within the system. This process was included in nitrogen and phosphorus, but not TDS budgets. Most of the sediment activity which can result in a flux of dissolved nutrients from the sediment to the water column or which results in assimilation of material into the benthic biota takes place in the top few centimeters of bay mud. We assume a generous 10 cm active depth, to include bioturbative expansion of the active layer.

Table 2. Monthly denitrification rates used in the Galveston nitrogen buget from relationship N_2 ($\mu M/m^2/h$) = 1.5476 · 10 (0.04827 · T), r2=.99, based on rates at upper bay sites. Temperatures are average 1977 - 1989 TWDB monitoring data, all Galveston Bay sites.

Month	Average Water Temperature, °C	Denitrification μg N/m^2/h
Jan	9.4	123
Feb	9.6	126
Mar	14.7	222
Apr	22.2	511
May	25.2	713
Jun	28.1	985
Jul	29.8	1190
Aug	29.6	1163
Sep	28.7	1053
Oct	20.2	409
Nov	17.7	310
Dec	11.9	163

Table 3. Calculation of Galveston Bay denitrification from apportioning base rate according to relative total organic carbon (TOC) content of sediments. Base rate, rate assuming all sediments equally active is $6978 \cdot 10^6$ g N/y. Proportional rate assumes direct proportionality with sediment % nitrogen, and %sediment organic N = $0.004 + 0.0917 \cdot$ %sediment organic C, r2=.96, from data in Zimmerman and Benner (1994) Table 2.

Sediment %TOC	Proportion Galveston Bay	Proportional rate	Denitrification 10^6 g N/y
1.0 <	0.401	1.0	2798
0.9 - 1.0	0.047	0.86	281
0.8 - 0.89	0.043	0.716	215
0.7 - 0.79	0.052	0.574	208
0.6 - 0.69	0.062	0.433	187
0.5 - 0.59	0.073	0.291	148
0.4 - 0.49	0.043	0.149	45
< 0.3	0.280	0.0	0
Total for bay			3883

Material which moves below this depth, as new sediment accumulates, is lost to the system.

The nitrogen content of the active sediment layer at five sites in Galveston Bay, from data in Montagna (1993) is approximately 1.22% dry weight. From P/N ratios in limited sediment data collected by TWC, TP averages approximately 0.44% sediment dry weight. Percentage composition was translated into amount of N and P per volume wet sediment based on 60% sediment water content (Zimmerman and Benner, 1994) and the specific gravity of sediment mineral constituents (2.65, Hakanson and Jansson, 1983, pg. 82).

From Shepard (1960), the average long-term rate of sedimentation in Galveston Bay is 0.44 cm/yr. Actual sedimentation rates should reflect differences in sediment delivery to the bay from the wide range of inflows during the period. We assumed a linear relationship between Trinity River inflows and sediment load, and a linear relationship between sediment load and sedimentation rate. Assuming Shepard's long-term sedimentation rate can be associated with a median inflow rate, the ratio of inflows : median inflow can be applied to provide an estimate of sedimentation rate for each year from the long-term average. Baseline burial rate is sedimentation rate times sediment nutrient content of a meter square, cm-thick slice of active sediment layer times the area of the bay bottom. The baseline rate was modified on the assumption that sediment nitrogen is not equal throughout the bay, but varies in a way associated with variation in sediment TOC content. The aerial distribution of sediment TOC was obtained from samples collected on a mile-square grid by White (1985). A sediment TOC to nitrogen conversion was taken from data in Montagna (1993). Burial rate was prorated from baseline by the relative frequency of categories of sediment content.

There are a number of assumptions critical in the estimation of nutrient burial. This reflects a lack of sediment data collection in this estuary and the difficulty of making process measurements applicable to the question. The effort may seem out of proportion to the importance of the contribution to nutrient budgets. However, the area of the system is so great in relation to its volume that all sediment processes are probably significant. Nixon (1987) argues that basing an estimate of short term sediment deposition rate on long term sediment accumulation may bias results. Long term sediment history incorporates episodic large inputs of sediment from major floods and storms, and so may not truly indicate routine deposition rates.

Nitrogen Fixation

Biological nitrogen fixation has been measured in estuarine habitats, chiefly at sediment surfaces. Nitrogen fixation rates are based on figures presented in Howarth et al. (1989). Planktonic fixation was assumed negligible. A uniform rate of 0.37 g N/M2/y was applied to the non-vegetated bay sediment area. A relatively conservative estimate of fixation occurring in areas of submergent and emergent vegetation was used, 1 g N/m2/y, based on the cautions stated by Howarth et al concerning errors which may arise with various techniques. Areas of vegetated and non-vegetated bay-bottom, to which these rates were applied were taken from Diener (1975). This may overestimate contribution from submergent vegetation according to analysis of trends reported by White et al. (1993).

Loss to Fisheries

Fisheries harvest is an obvious, if sometimes discounted, loss of nitrogen to the estuary. In addition, many of the fisheries species undergo movements through their life history which also result in a net loss of material to the estuary. Many species grow from millimeter-long larvae in the estuary to maturity and then leave to spawn. Their mortality in the Gulf (including Gulf fisheries) is then great enough so that return of adults to the estuary can be discounted. Both in-bay fisheries harvest and escapement of fish and shellfish to the Gulf are included in the budget.

Escapement rates for dominant fish species are based on analyses of the population structure determined through systematic sampling of fisheries populations in the bay by TPWD Coastal Fisheries biologists, and using other information. Shrimp escapement rates are calculated from Gulf harvest, weighted by estuary contribution ratios (Robinson, 1993). **Table 4** lists major species and biomass escapement loss with other fisheries losses. Bay commercial harvest data and estimates of bay recreational harvest were taken from reports of the Texas Parks and Wildlife Department (Campbell, 1992; Weixelman et al., 1992). Values of tissue content and conversion factors from Zison et al (1978) were used to convert biomass loss to nitrogen and phosphorus loss.

Water Column Storage

For budgets covering a discrete time period, as produced here, changes to the pool of material residing within the system can also be large enough to require inclusion in the budget. Data are insufficient to calculate changes to the sediment nitrogen and phosphorus pools. Changes to the pools of nitrogen and phosphorus

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Table 4. Fisheries harvest and escapement data for the Galveston Bay nutrient budgets. 10^3 kg .

Category of Loss	1988	1989	1990
Bay Commercial Harvest			
White Shrimp	1400	819	1050
Brown Shrimp	1566	1020	895
Blue Crab	1414	975	865
Oyster	659	320	529
Subtotal	5039	3134	3340
Bay Recreational Harvest			
Finfish	512	398	807
Escapement			
Atlantic Croaker	10562	7613*	4665
Red Drum	14	14*	15
Gulf Menhaden	271	157*	43
Striped Mullet	2928	4900	4149
Spot	95	76*	58
White Shrimp	1310	1429	786
Brown and Pink Shrimp	3256	1965	3713
Blue Crab	6621	23175*	39728
Subtotal	25058	39329	53158
Total Biomass Loss	30608	42681	57306
Total N Loss	765	1072	1433
Total P Loss	199	270	361

* Estimated from 1988,1990 trend

held in the bay waters can be calculated from monitoring data of the TWDB and TNRCC.   For this analysis, changes in water column nutrient mass were based on bay-wide volume-weighted average concentrations and total bay volume.   Simulations showed negligible difference between beginning and ending volume of the estuary.

Results   and   Discussion

Nitrogen   and   Phosphorus   Loading

Table 5 presents annual masses of nitrogen and phosphorus supplied to the estuary from gaged stream inflows, ungaged rainfall runoff, returned waste water, and direct rainfall on the estuary surface.   From these data, what would be termed point source loadings can be seen to be very important.

In Table 6 these loadings are compared to those compiled by others.   The comparisons are not exactly on the same basis: NOAA (1989) calculated generalized annual TP and TKN loadings to the bay in point, nonpoint, and upstream categories; Ward and Armstrong (1993) contrasted effluent loads with major tributaries; here, our "gaged" category includes major tributaries and minor streams within the coastal drainage basin.   The table shows NOAA estimates to be significantly higher than those compiled by this project.   Part of this is due to their tabulation of point sources upstream of gages and so included in our gaged category.   Other differences may reflect their use of generalized land use-to-loading relationships.

A detailed GIS-based approach to compilation of nonpoint source loadings to Galveston Bay is presented by Newell et al (1992) for the Galveston Bay National Estuary Program (GBNEP).   Table 7 compares TN and TP loadings compiled by that report to those compiled for this project, based on watersheds which were equivalent in the two studies.   There are several watersheds for which the two studies give divergent results.   For this study, loadings for Clear Creek are higher and for Cedar Bayou they are much lower than reported by Newell.   However, there is general agreement between the two studies for many watersheds.   Total ungaged loadings compiled for 1988-1990 fall in the range of Newell's totals for average year and wet year nonpoint source loadings.

Point source loadings compiled for this project are compared with those of Armstrong and Ward (1993) in Tables 8 and 9.   Our estimates were compiled by watershed while Armstrong and Ward's are compiled by stream segment, so again comparisons are limited to situations in which there is a

Table 5. Sources of drainage basin TN and TP loadings (10^6 g/y) to the Trinity-San Jacinto Estuary.

Source	1988	1989	1990
Total Nitrogen			
Gaged Stream Flow	8360	25900	35090
Ungaged Rainfall Runoff	3320	9590	5890
Wastewater Returns	7250	7290	7570
Direct Rainfall	570	760	700
Total	18930	42780	48550
Total Phosphorus			
Gaged Stream Flow	1730	3870	3810
Ungaged Rainfall Runoff	1140	2750	1760
Wastewater Returns	1720	1730	1800
Direct Rainfall	0	0	0
Total	4590	8350	7370

Table 6. Nitrogen and Phosphorus Loadings compiled by NOAA (1989) and projects of the Galveston Bay National Estuary Program (Armstrong and Ward, 1993, Table 5.2).

Nitrogen

NOAA, TKN	10^6 Kg/y	GBNEP, TN	10^6 kg/y	This Study	10^6 kg/y
Point	36.7	Point	8.4	Point	6.7
Non-Point	19.2	Non-point	6.4	Ungaged	8.4
Upstream[a]	4.5	Tributaries[a]	9.7	Gaged	22.5
Total	60.5	Total	24.5	Total	37.6

Phosphorus

Point	9.3	Point	4.0	Point	3.1
Non-point	1.7	Non-point	1.1	Ungaged	1.8
Upstream[a]	1.0	Tributaries	1.5	Gaged	3.6
Total	12.0	Total	6.6	Total	8.5

a   NOAA upstream are tributaries gaged outside the estuarine drainage area while Armstrong and Ward used gaged Trinity River plus Lake Houston spills, and Gaged flows in this study included small gaged streams inside coastal drainage area.

Table 7. Comparison between total nitrogen annual loading rates compiled for this project and those presented in the Galveston Bay Non-point Sources report (GBNEP-15, Newell et al. 1992), for corresponding watersheds.

Total Nitrogen 10^3 kg

Watershed	This Report			GBNEP-15	
	1988	1989	1990	Ave Year	Wet Year
White Oak Bayou	292	621	396	365	518
Sims Bayou	432	856	598	235	408
Dickinson Bayou	153	311	192	130	201
Clear Creek	609	1895	722	301	520
Cedar Bayou	24	110	57	321	500
San Jacinto below Lake Houston	68	125	160	126	189
Trinity Bay	372	1183	831	356	572
East Bay	215	728	500	388	615
Total all watersheds	3320	9590	5890	6422	10073

Total Phosphorus 10^3 kg

Watershed	This Report			GBNEP-15	
	1988	1989	1990	Ave Year	Wet Year
White Oak Bayou	180	382	244	69	98
Sims Bayou	124	232	196	41	71
Dickinson Bayou	68	61	36	21	32
Clear Creek	246	911	303	51	87
Cedar Bayou	6	23	12	58	90
San Jacinto below Lake Houston	43	63	96	22	33
Trinity Bay	25	81	57	59	93
East Bay	9	32	22	68	107
Total all watersheds	1140	2750	1760	1113	1727

Table 8. Wastewater Total Nitrogen loadings into Galveston Bay: Comparsion between annual loadings compiled for this report (TWDB) and those compiled by Armstrong and Ward, 1993, for the Galveston Bay National Estuary Project (GBNEP-36). Comparison limited to areas showing best correspondence between TWC segments and TWDB watersheds.

Area	TWDB Watersheds	GBNEP Stream Segments	Loading,10^6 g/yr TWDB	GBNEP-36
Trinity Tidal	08010	0801	85	42
Cedar Bayou Tidal	09010	0901, 0902	260	113
San Jacinto R. Tidal	10010	1001	127	65
Lower San Jacinto & HSC	10050	1005	406	437
Clear Creek	11010, 11020 11130	1101, 1102, 1113	371	431
Dickinson Bayou, Tidal	11030	1103	45	52
Chocolate Bayou, Tidal	11070,11080	1107	196	1
Upper Galveston Bay	11150	2421, 2438, 2425	268	128
Trinity Bay	07070	2422	76	10
East Bay	07060	2423	102	0
West Bay	24240, 11110, 11092, 11124	2424	162	108
Moses Lake	11122	2431	568	124
Lower Galveston Bay	24390	2437, 2439	271	314
All segments 1990 Wastewater Loading			7570	8425

Table 9. Wastewater Total Phosphorus loadings into Galveston Bay: Comparsion between annual loadings compiled for this report (TWDB) and those compiled by Armstrong and Ward, 1993 (GBNEP-36). Comparison limited to areas showing best correspondence between TWC segments and TWDB watersheds.

Area	TWDB Watersheds	GBNEP Stream Segments	Loading, 10^6 g/yr TWDB	Loading, 10^6 g/yr GBNEP-36
Trinity Tidal	08010	0801	20	21
Cedar Bayou Tidal	09010	0901, 0902	62	37
San Jacinto R. Tidal	10010	1001	30	82
Lower San Jacinto & HSC	10050	1005	96	185
Clear Creek	11010, 11020 11130	1101, 1102, 1113	88	215
Dickinson Bayou, Tidal	11030	1103	11	26
Chocolate Bayou, Tidal	11070,11080	1107	47	0
Upper Galveston Bay	11150	2421, 2438, 2425	64	65
Trinity Bay	07070	2422	18	5
East Bay	07060	2423	24	0
West Bay	24240, 11110, 11092, 11124	2424	39	54
Moses Lake	11122	2431	135	62
Lower Galveston Bay	24390	2437, 2439	64	220
All watersheds 1990 Wastewater Loading			1800	4002

good correspondence between the coverage of the two systems. There are again cases in which the two studies clearly diverge. For Chocolate Bayou, tidal, East Bay, and Moses Lake, our loadings are high. Yet, our total is lower, so it may be a case of differing assignments of sources in watersheds.

Water Balance Results

The summary of annual flow components is presented in **Table 10**. Notice that freshwater inflow in 1988 is about one-forth the inflow during 1989 and 1990. Ungaged inflow in 1989 is greater than during the nominal wet-year, 1990. This result of local basin storm events will have ramifications visible through the materials balances, because nutrient concentrations are relatively high in local basin streams. **Figure 4** demonstrates the relative magnitude of freshwater inflow volumes vs. tidal volumes moving in and out of Bolivar Roads and San Luis Pass. With tidal volumes one or two orders of magnitude greater in volume than freshwater inflow, tides have a large impact on materials transport if tidal mixing parameters are on the order of magnitude of 10% and even though Gulf concentrations are lower than bay concentrations by an order of magnitude. The differences between total tidal volumes among the years is on the order of 1%, less than probable error in simulation results. Therefore, average tidal volumes could probably have been used in the materials balances for all years.

TDS Balance

The TDS balance was used to calibrate tidal mixing rate (entrainment rate). Therefore, the results of the TDS budgets can not be analysed as a strictly independent conservative materials check to validate materials transport mechanisms, as is desirable in nutrient budget exercises. The TDS budget can be used to show the major pathways of materials transport in the system. In **Table 11**, values for ICWW East, ICWW West, and Cold Pass represent magnitude only; there were inadequate concentration data to satisfactorily represent inputs. The major differences among years in TDS mass exchange are the products of tidal movement and different bay and Gulf concentrations. Freshwater contribution is in the form of reduced bay concentrations with high inflows.

Nitrogen Budget Results

Nitrogen budget annual summaries for 1988, 1989, and 1990 are presented in **Table 12**. The signs of the remainders for high and low inflow years

Table 10. Galveston Bay system annual water budget during project years.
Units are 10^6 M^3/Yr.

	1988	1989	1990
Freshwater Inflow			
Gaged Streamflow	3439	11376	15155
Ungaged Inflow	1339	3970	2509
Diversions	300	261	261
Wastewater Returns	620	623	647
Rainfall on Bay Surface	1395	1862	1700
Evaporation	-2182	-2021	-2121
Subtotal	4309	15552	17629
Tidal Volumes (+/-)*			
Bolivar Roads	163920	164392	164709
San Luis Pass	31784	32319	33145
Rollover Pass	6070	6189	6453
ICWW East	169	170	171
ICWW West	2150	2191	2302
Cold Pass	3010	3101	3320
Net Outflow **			
Bolivar Roads	-3554	-12753	-14455
San Luis Pass	-433	-1555	-1762
Rollover Pass	-347	-1244	-1410
Subtotal Outflow	-4334	-15552	-17627
Net Flows Balance	-25	0	2

*   Symmetric flood and ebb tidal volumes
**   Net outflows to Cold Pass and ICWW East and West were negligible

Figure 4.   Relative magnitude of tidal exchange and freshwater inflows
to Galveston Bay.
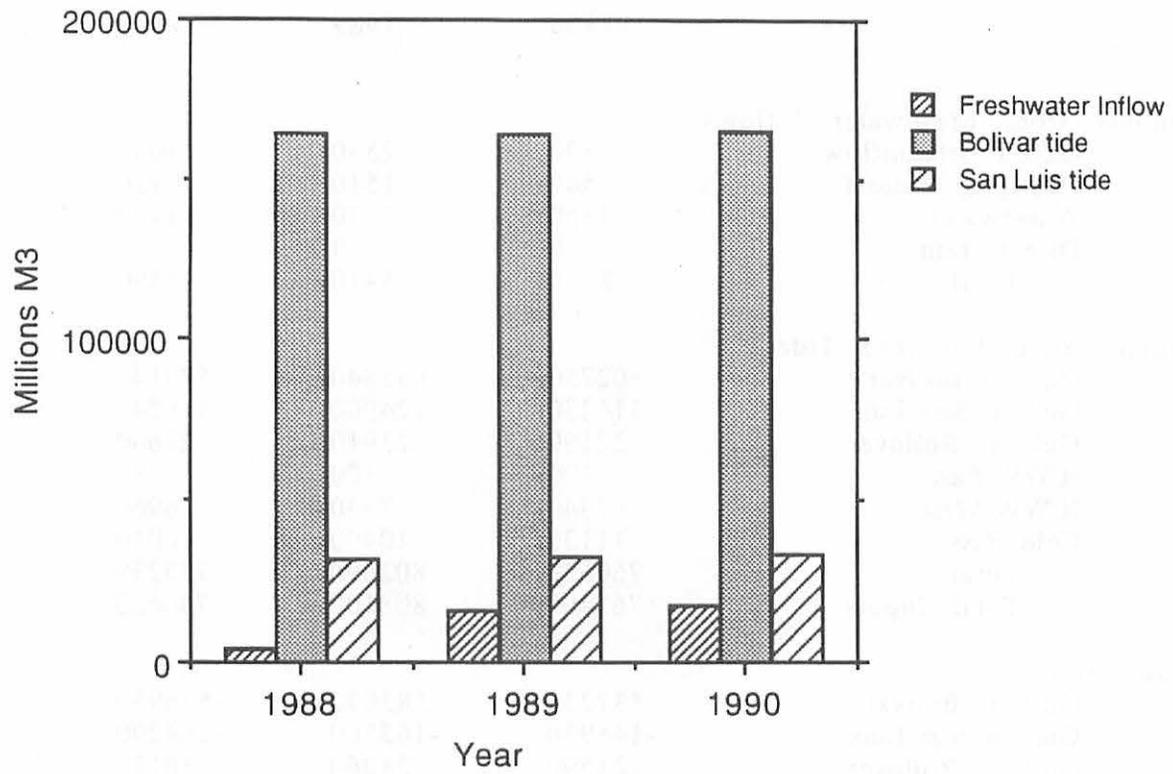
Table 11. TDS budgets for the Galveston Bay system for three annual periods. 10^6 kg/yr TDS .

	1988	1989	1990
Inputs from Freshwater Inflows			
Gaged streamflow	870	2530	3050
Ungaged runoff	540	1510	920
Wastewater	1360	1370	1420
Direct rain	0	0	0
Total	2770	5410	5390
Inputs from Entrained Tides			
Gulf at Bolivar	602750	635840	579140
Gulf at San Luis	116230	124900	116240
Gulf at Rollover	22190	23910	22600
ICWW East	390	300	280
ICWW West	7940	7340	6960
Cold Pass	11130	10400	10010
Total	760630	802690	735230
Total Inputs	763400	808100	740620
Outflows			
Gulf at Bolivar	-537230	-583070	-536930
Gulf at San Luis	-146930	-165500	-164200
Gulf at Rollover	-21590	-28260	-30110
ICWW East	-390	-300	-280
ICWW West	-7940	-7340	-6960
Cold Pass	-11130	-10400	-10010
Total Outflows	-725210	-794870	-748490
Transport Balance	38190	13230	-7870
Water Column Storage	16550	-1320	-4790
Remainder	21640	14550	-3080

Table 12.   Annual total nitrogen budget for the Galveston Bay system, 106 g /y TN.

	1988	1989	1990
Inputs from Freshwater Inflows			
Gaged streamflow	8360	25900	35090
Ungaged runoff	3320	9590	5890
Wastewater	7250	7290	7570
Direct rain	570	760	700
Total	19500	43550	49250
Nitrogen Fixation	560	560	560
Inputs from Entrained Tides			
Gulf at Bolivar	1490	1500	1380
Gulf at San Luis	290	290	280
Gulf at Rollover	60	60	50
ICWW East	20	20	20
ICWW West	200	240	210
Cold Pass	270	330	300
Total	2330	2440	2240
Total Inputs	22390	46550	52050
Outflows			
Gulf at Bolivar	-20000	-33830	-29950
Gulf at San Luis	-3650	-5500	-5210
Gulf at Rollover	-1280	-2460	-2900
ICWW East	-20	-20	-20
ICWW West	-200	-240	-210
Cold Pass	-270	-330	-300
Total Outflows	-25420	-42380	-38590
Transport Balance	-3590	3610	12900
Denitrification	-3680	-3680	-3680
Sediment Burial	-690	-2280	-2620
Fisheries, Fish migration	-770	-1070	-1430
Total Losses	-30560	-49410	-46320
Water Column Storage	170	270	-360
Remainder	-8000	-2590	5370

are consistent with the idea that an estuary might be accumulating nitrogen when loading is high and may be depleting nitrogen internal storage compartments during a year of low inputs. This is also a result expected when many important loss rates are fixed, but inputs vary.

Consider the main losses covered in the budget. Loss in net outflows are proportional to inflows when the concentrations of TN in outflows are similar in years of varying inflows. In the lower Galveston Bay, TN concentrations are weakly correlated with inflows (**Figure 5**). Concentrations during the study years are distinguished by only one high reading in 1989 (**Figure 6**). Entrainment of tidal waters contribute and withdraw similar amounts of nitrogen over the years of study. (Using one set of tides for all years does not materially change results.) For the years in question, or a typical series of years, loss rates which are similar based on similarity of water column and sediment TN may be entirely expected. In sediment and biological storage, the system develops some carry-over between years of high loading. The reason lower bay concentrations remain relatively high may have to do with sediment flux rates maintaining a sort of equilibrium. The reason may also be limited sedimentation or filter feeding rates in this estuary, which would remove particulates and phytoplankton (both contribute to TN used here), or sufficient flushing to prevent a strong dissolved nitrogen gradient. Denitrification has also been assumed to be constant during study years. This is consistent with the idea that the sediment nitrogen provides a system nitrogen buffer, as loadings fluctuate over a relatively short time span. In addition, over most of the estuarine salinity range, denitrification rates may not be influenced by at least short term differences in salinity (Nowicki, 1994). The preceding considerations support the trend in results of budgets as observed for inflows as they presently vary. However, it may be unwise to use the budget results in a straightforward way to predict the situation where the loadings vary around a new average. This is discussed further below.

Figure 7 illustrates proportional contributions to inputs and outflows. Loss of nitrogen to the Gulf is the dominant sink; denitrification is second in importance. Burial loss is greater than fisheries loss in years of moderate or high inflows. Although net movement of nitrogen through tidal action is out of the bay, entrainment inward is shown--a result of the entrainment model of tidal mixing used here. This is a small contribution, but it is significant, especially for the low flow year. The amount of nitrogen unaccounted for in the budgets is 10% or greater of total inputs. We do expect some differences among the years in aspects of the system

Figure 5. Variation of lower Galveston Bay TN with inflow. CDS points are TN=TKN+NO3+NO2, others from regression on NO3+NO2+NH3. Line is fit through CDS points.

$$y = 0.59784 + 2.0130e\text{-}5x \quad R^2 = 0.157$$
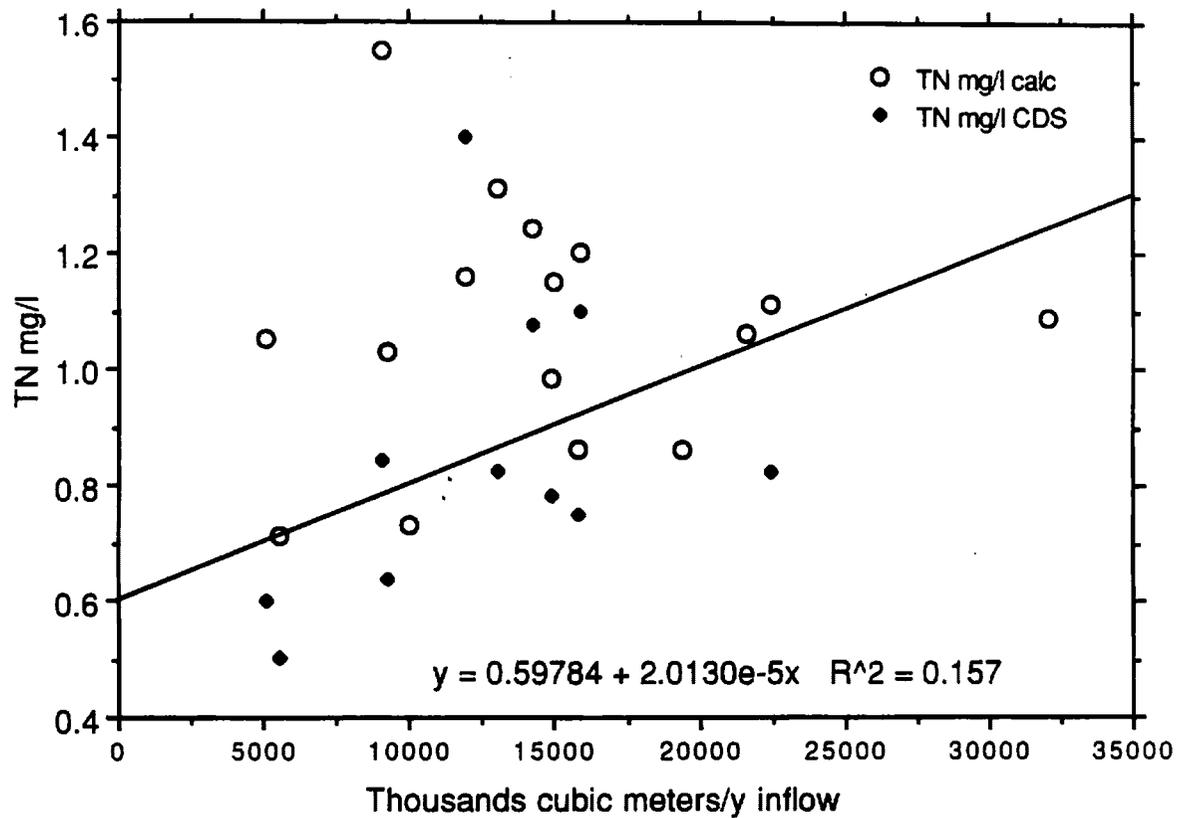
Figure 6. Average TN concentrations in lower Galveston Bay.
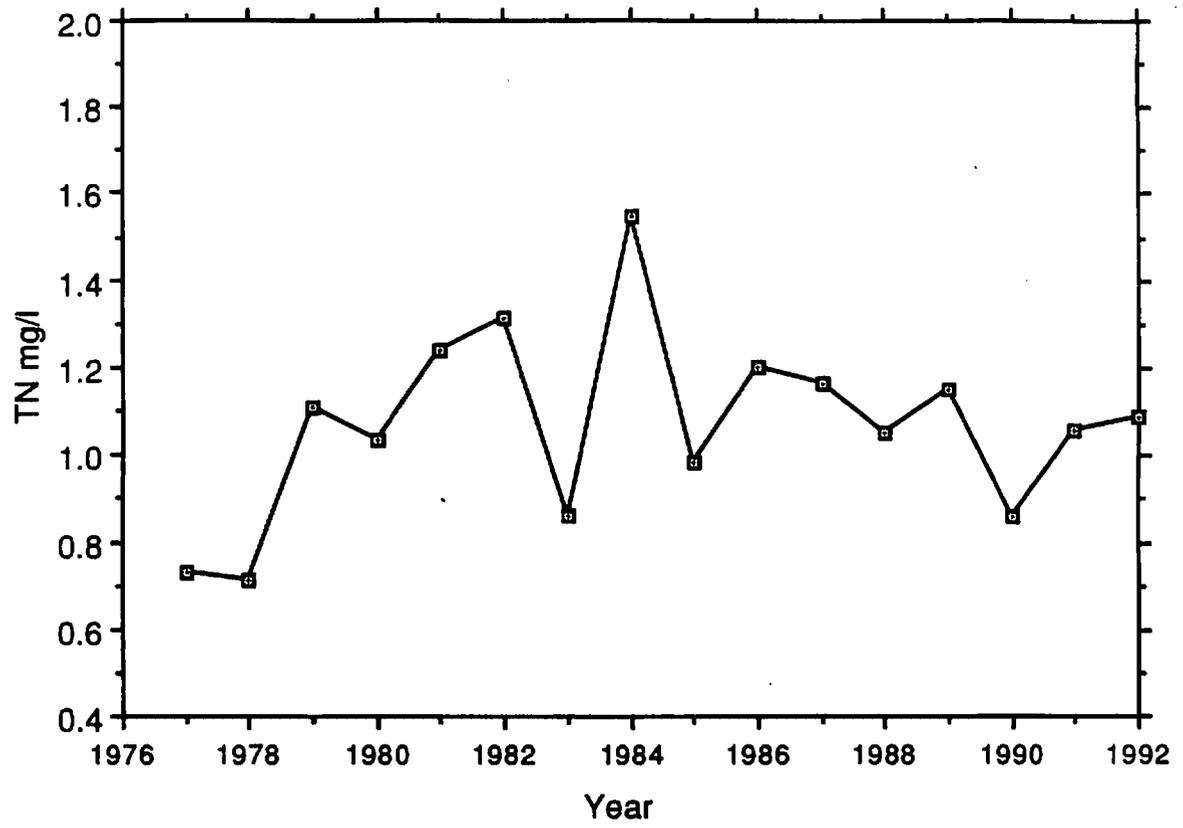
Figure 7. Comparison of TN inputs and losses, Galveston Bay. Note log scale.
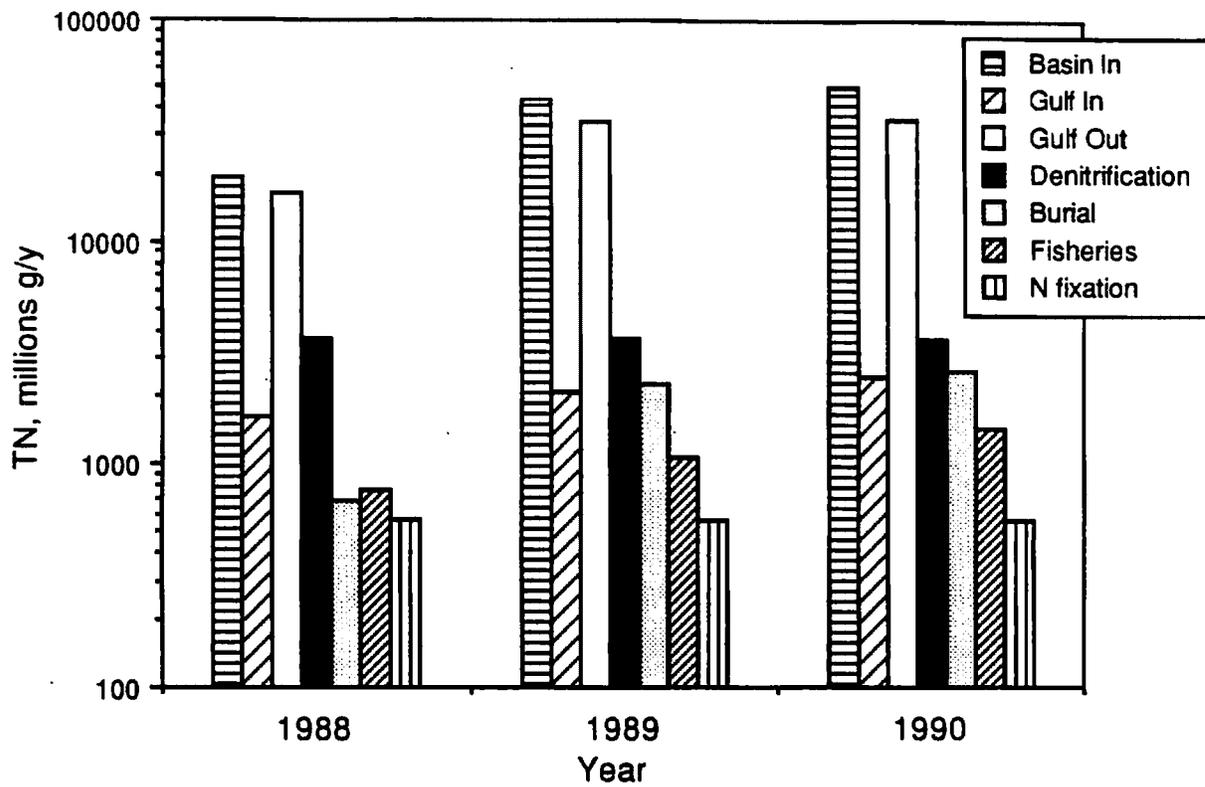
which are not well represented here, namely storage in sediment and biological compartments.

The negative remainder in the year of moderately high inflows, 1989, may indicate a bias in underlying assumptions, underestimating inputs and/or overestimating losses. Entrainment rate is a factor which could produce such a bias in general results, because the volumes involved are so large. **Figure 8** shows how the budget remainders. change with entrainment. Although we attempted to derive entrainment rates which were not arbitrary, it is clear that more certainty regarding this term would be needed to assure confidence in the budgets' bottom line.

Phosphorus Budget Results

Table 13 shows the phosphorus budgets compiled for each year. These results do not seem consistent with those of the nitrogen budget, as the remainders are positive in a low flow year and negative in the high inflow year. The difference between the high and moderately high inflow years also raises questions. Advective transport loss removes for 91% of inputs in 1990 vs. 82% and 64% of inputs in 1988 and 1989, respectively. Proportionally higher TP loss in 1990 is a result in part of higher TP concentrations in the lower bay during 1990 than during 1988 (**Figure 9**). Historical data do not indicate a clear or statistically significant increasing trend of TP concentrations in the lower bay with inflow (**Figure 10**). So it is unclear whether we can use 1990 results to infer TP dynamics at high-inflow. Conceptually, however, the high rate of loss in a period of high inflows may be believable: Much of the phosphorus which enters the estuary is absorbed/adsorbed to clay minerals (Froelich, 1988), and the rate of deposition of these particulates in the upper bay is likely to be higher during low flow periods than during high flow periods. Therefore, there is more TP in the water column to be transported during high flow periods.

High burial rates applied to high inflow conditions are also responsible for the tilt of the phosphorus balance. Uncertainties concerning the sediment pool and burial rates are thus a major concern for this TP balance. Other considerations mean that sediment TP dynamics is important for understanding the TP dynamics in the estuary as well.

The P/N ratio in Galveston Bay sediments, averaging 4/10, is more than twice as high as we would expect from the ratio in living organic matter. This indicates that Galveston Bay sediments either gain more phosphorus through geochemical mechanisms than through sedimentation of organic detritus and planktonic

Figure 8. Impact of value of entrainment rate on nitrogen budget remainder.
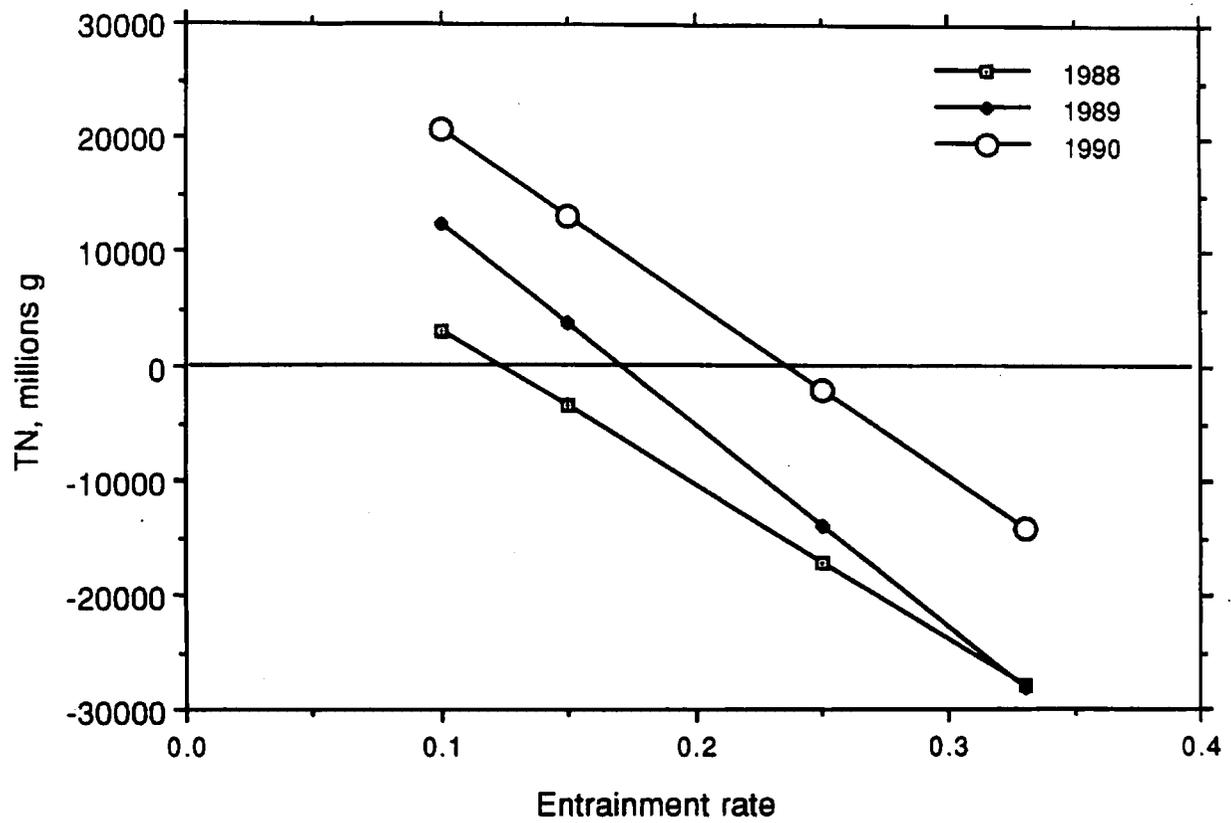
Table 13. Annual total phosphorus budget for the Galveston Bay system, 10^6 g/yr TP.

	1988	1989	1990
Inputs from Freshwater Inflows			
Gaged streamflow	1730	3870	3810
Ungaged runoff	1140	2750	1760
Wastewater	1720	1730	1800
Direct rain	0	0	0
Total	4590	8350	7370
Inputs from Entrained Tides			
Gulf at Bolivar	90	90	80
Gulf at San Luis	20	20	20
Gulf at Rollover	3	3	3
ICWW East	0	0	0
ICWW West	20	20	20
Cold Pass	20	30	30
Total	150	160	150
Total Inputs	4730	8510	7520
Outflows			
Gulf at Bolivar	-3420	-4780	-5940
Gulf at San Luis	-330	-410	-550
Gulf at Rollover	-90	-200	-270
ICWW East	-0	-0	-0
ICWW West	-20	-20	-20
Cold Pass	-20	-30	-30
Total Outflows	-3880	-5440	-6810
Transport Balance	850	3070	710
Sediment Burial	-360	-1190	-1370
Fisheries, Fish migration	-200	-270	-360
Total Losses	-4430	-6900	-8540
Water Column Storage	100	240	-240
Remainder	400	1850	-1260

Figure 9. Total phosphorus in lower Galveston Bay, averaged monthly over all depths.
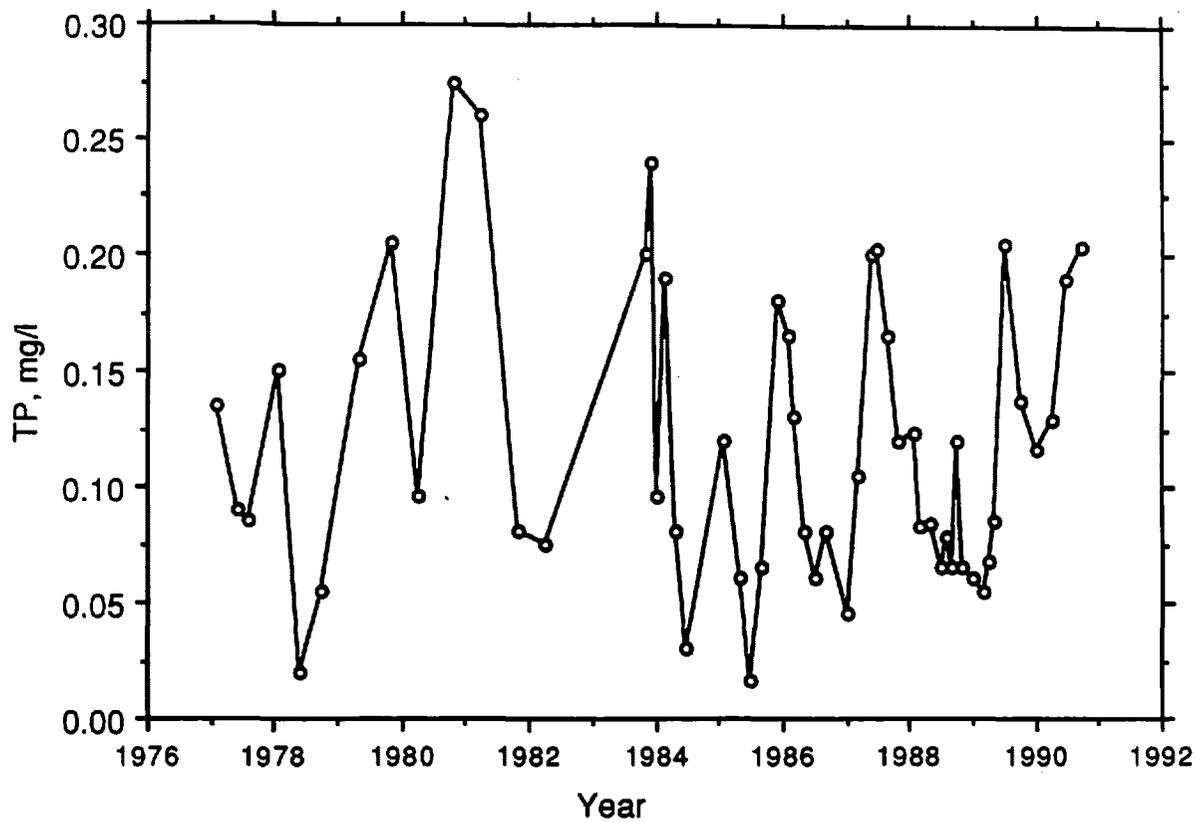
Figure 10. Total Phosphorus concentrations in lower Galveston Bay vs inflows.
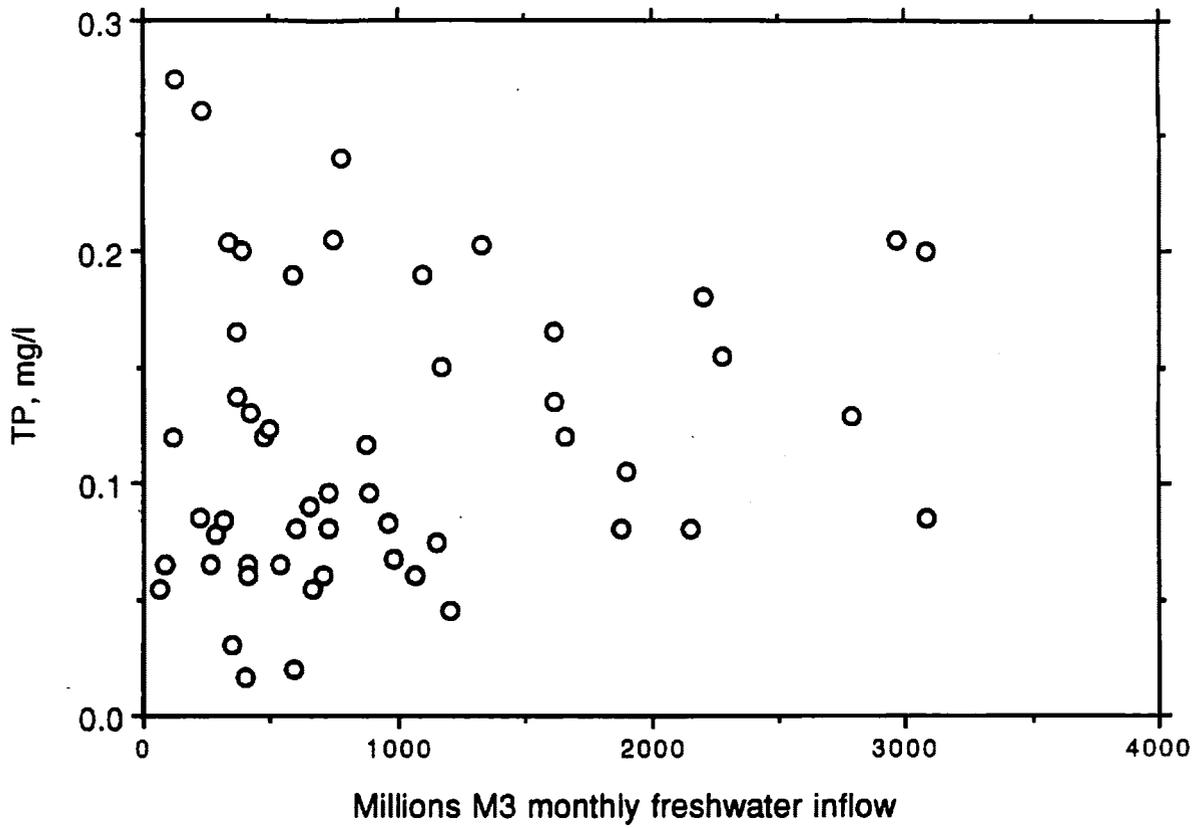
production, or that processes remove proportionally more nitrogen from deposited material. With the predominance of clays in Texas bays, we might expect a strong geochemical retention effect based on other studies (Froelich, 1988). The size of the sediment pool would be a major consideration if phosphorus were suspected of influencing system production. Studies show that kinetics of phosphate mineralization/demineralization and absorption/desorption can maintain dissolved phosphorus in the estuarine water column in biologically important levels. These kinetics are influenced by salinity, the difference between phosphorus concentrations in solid and liquid phases, and pH. Typically, phosphorus entering an estuary precipitates out of solution with iron and aluminum complexes as it enters brackish waters. As salinities increase down the estuary, or if higher salinities move into the upper estuary, some of the bound phosphorus is released to the aqueous medium. An increase in pH may also aid phosphorus release. So, movement of P into the aqueous phase of the estuary where it may be lost to tidal exchange is dependent on an interaction of factors.

The difference between 1989 and 1990 TP budgets shows principally the strong impact of loadings from local drainage flooding in 1989. These urbanized coastal drainages have higher concentrations of materials in comparison with those of the Trinity River, which contributed the largest volume to flood flows in 1990.

Long-term Budgets for Nitrogen and Phosphorus

The nutrient budgets presented for years 1988-1990 demonstrate what processes are most important during those years and provide a basis for comparative analysis. However, for some purposes, the budgets for specific years are limited. One data gap became an important limitation for these annual budgets: the difficulty of measuring change in system storage. Aside from the lack of aerial and temporal coverage, sediment storage in particular is difficult to measure. Change in sediment storage is also difficult to accurately determine over a year, which may be a relatively short span of time in the context of sediment change. Over a longer span of time, changes in system storage are either measurable or become negligible as short term changes hover about an equilibrium. As this problem demonstrates, the compilation of a budget applicable to a longer time span may be more reliable in some aspects than budgets for annual periods. So a long-term perspective may be most appropriate for some management concerns. Here, the nitrogen and phosphorus budgets developed for years of various inflows are used to propose long-term budgets for the Galveston Bay system.

Long-term budgets can be based on two general assumptions: (1) some of the sources and sinks are relatively constant (or vary moderately about an average) over the years; (2) other terms can be stated as an indirect or direct function of drainage basin freshwater inflows. Processes of rainfall inputs, nitrogen fixation, denitrification, and wastewater returns can be considered relatively independent of drainage basin inflows. Wastewater inputs have changed dramatically during this century (Armstrong and Ward, 1993, Figure 2.2), increasing to a peak in the 60's, then decreasing. However, wastewater loadings have been relatively stable since the 70's. For non-inflow related inputs, we can use average values from **Tables 12 and 13** for these constant processes in a long-term budget.

The drainage basin inputs of nitrogen and phosphorus can be estimated as the product of the median rate of gaged+runoff inflows and median inflow concentrations. Net outflows of the bay to the Gulf are estimated as the product of median water-balance inflow (including rain and evaporation) and median lower-bay concentrations. Magnitudes of other sources and sinks which are inflow dependent can be estimated for a long-term budget based on a rate-inflow regression developed from the data in the present study. Their expected value is then determined for median inflows. With respect to inflow volumes, the average volume may be substantially larger than the median, showing an influence of flood volumes. While flood flows probably contribute large loadings, and perhaps loadings which more likely increase system nutrient storage, as may be inferred from the 1990 budget, it is probably not the case that using a average inflow represents the system impact of flood flows. Floods have a qualitative as well as quantitative impact, that is hard to quantify. For some planning concerns, we have to assume that there will be floods of an order which will do for the system what floods have done. The assumptions and constants used in the long-term budgets are presented in **Table 14**.

A long-term nitrogen budget is presented in **Table 15**. The first column is the prototype from application of the equations and relationships discussed above. It has a negative nitrogen balance. Over the long-term, if the system is stable, this balance should be zero, even if it is not during some years. The discrepancy signals the need for more data or re-evaluation of assumptions used to create the original budgets or the long-term budget (such as linear relationships with inflow). For the purposes of the present discussion, however, the approach taken is to modify source or sink masses for the components of the budget to produce a zero-balance budget. If we can modify those in an objective manner and in a way which preserves their realism then we can use the budget in a qualitative way. The second column shows

Table 14. Data and relationships used in constructing long-term nitrogen and phosphorus balances for the Galveston Bay system.

GUD= Median inflows, gaged+ungaged-diversions $10.1 \cdot 10^6$ acre-feet/y
GMD=Median inflows, gaged+ungaged-diversions $12453 \cdot 10^6$ m^3
WB= Median water balance inflows $12590 \cdot 10^6$ m^3

Nitrogen
GN=Gulf TN concentration   0.07 mg/l N
BN=Average Lower Galveston Bay TN concentration   0.91 mg/l N
Volume-weighted average inflow TN conc.   2.44 mg/l N
Average Rain inputs $700 \cdot 10^6$ g N/y
Average wastewater reurns $7300 \cdot 10^6$ g N/y
Average nitrogen fixation $560 \cdot 10^6$ g N/y
Average denitrification $3680 \cdot 10^6$ g N/y
Inward entrained Gulf tide   =0.12·(205880+0.000234·GUD)*GN
Net export to Gulf   =GMD*BN
Outward entrained Gulf tide   =0.15·(205880+0.000234·GUD)*BN
Loss to fish harvest, migration   =510.34+0.0000628·GUD
Loss to burial   =-100.77+0.2699·GMD

Phosphorus
GP=Gulf TP concentration   0.004 mg/l P
BP=Average Lower Galveston Bay TP concentration   0.12 mg/l P
Volume-weighted average inflow TP conc.   0.41 mg/l P
No Rain inputs
Average wastewater reurns $1750 \cdot 10^6$ g P/y
Inward entrained Gulf tide   =0.12·(205880+0.000234·GUD)*GP
Net export to Gulf   =GMD·BP
Outward entrained Gulf tide   =0.15·(205880+0.000234·GUD)*BP
Loss to fish harvest, migration   =139.57+0.0000628·GUD
Loss to burial   =-37.467+0.0000962·GMD

Table 15. Long-term nitrogen budget. First column is as calculated, the second column contains adjustments to * items in proportion to their relative magnitude to obtain a zero balance. Units are 10^6 g N/y.

	Calculated	Adjusted
Inputs		
GMD Streamflow	30386	30386
Wastewater	7300	7300
Total Drainage Basin	37686	37686
Inward Gulf Tide Entrainment	1749	1749
Direct Rain	700	700
Nitrogen Fixation	560	560
Loses		
Net Export to Gulf*	11333	9752
Loss to Gulf Tide Entrainment*	28424	24460
Loss to Fisheries	1065	1065
Loss to burial*	2616	2251
Loss to Denitrification*	3680	3167
Total In	40695	40695
Total out	47118	40695
Remaining	-6423	0

modifications. Here, the budget error was split between net outflows, entrained outflows, burial, and denitrification, in proportion to their magnitude. We know that there is some uncertainty in our estimates of some budget components. For denitrification and burial, as examples, measurements and estimates of rates were applied to the estuary as a whole based on a hypothetical relationship between sediment TOC content and rates.

The resulting adjusted long-term budget is still similar to the prototype. Regarding the modification of outflow losses, calculations show that the adjustments to outflows to achieve a balance are equivalent to what we would see with bay concentrations reduced from an annual median 0.91 mg/l to 0.78 mg/l, or from slightly lower tidal entrainment. The lower concentration lies between the median and 25% quartile of historical concentrations.

A long-term phosphorus budget is presented in **Table 16**. The difference between inputs and losses in the prototype is much greater than in the nitrogen budget. In general, we are not accounting for losses which must be occurring. Possibly concentration data do not adequately represent conditions in the lower bay during windy periods, when phosphorus bound to clay minerals would be in the water column and subject to transport loss. Lower bay concentrations which would be required to support transport losses meeting inputs would be near .17 mg/l, around the 75% quartile of observed data. It is also likely that burial rates of TP near the Trinity River delta are larger than the data from other stations in the bay suggest.

The long-term nitrogen budget is a description of the status quo, showing persistent relationships underlying yearly variation. We can use it to predict the impacts of perturbations to the system nitrogen-status over the short-term. However, although it describes a long-term state, it does not alone provide a guide to system status following prolonged change in input or loss rates. For example, if wastewater inputs suddenly ceased, the budget would indicate the system would lose more nitrogen than it received, implying that some internal nitrogen storage would be reduced. This would be the case for one or several months. However, if this condition persisted for a number of years, we would not expect to see continued increasing system nitrogen deficits. Various mechanisms would force bay water concentrations down slightly, which would reduce transport losses and bring the net balance back to zero. This capacity for the system to respond to persistent change needs to be considered in estimating effects on the system of environmental management. System responsiveness is addressed further in the last section of the

Table 16. Long-term phosphorus budget. First column is as calculated. the second column contains adjustments to * items in proportion to their relative magnitude to obtain a zero balance. Units are 10^6 g P/y.

	Calculated	Adjusted
Inputs		
GMD Streamflow	6974	6974
Wastewater	1750	1750
Total Drainage Basin	8724	8724
Inward Gulf Tide Entrainment	100	100
Loses		
Net Export to Gulf*	1494	2069
Loss to Gulf Tide Entrainment*	3748	5190
Loss to Fisheries	271	271
Loss to burial*	934	1293
Total In	8824	8824
Total out	6448	8824
Remaining	2376	0

report and is used in the following section in arguments related to a nitrogen requirement for Galveston Bay.

Nutrient Requirements for Galveston Bay

This section presents data, assumptions, and discussion of ideas concerning establishment of a nutrient requirement for the Galveston Bay system. The topic is addressed again as part of the section on primary production, but here information is discussed in the context of nutrient budgets and system processes.

A nutrient requirement for an estuary should be consistent with set goals of maintaining "an ecologically sound environment...that is necessary for the maintenance of productivity of economically important and ecologically characteristic sport or commercial fish and shellfish species and estuarine life upon which such fish and shellfish are dependent" (Texas Water Code 11.147(a)). The bounds of a requirement for nutrients could be stated as an amount great enough to promote production supporting commercial and recreational harvest, great enough to maintain the community characteristic of the estuary, but an amount not so large as would bring the environment to unsound conditions, nor promote an uncharacteristic ecological community.

At a management level, definition of a target requirement involves reconciliation of demands for high inputs to fuel maximal production with desires for a lower input rate consistent with maintaining the quality/diversity of the ecological community. Here, we assume that the goal is to maintain a balanced system, not necessarily one with the greatest productivity of total biomass, but one with high production of desirable species.

Nitrogen Focus

Nitrogen is the nutrient usually assumed to be limiting to production in Texas estuaries. Here, a nutrient requirement is stated in terms of nitrogen only. Armstrong and Hinson (1973) report results of nutrient limitation experiments which suggest light is the principle limitation on primary production in Galveston Bay waters, although additions of nitrogen and phosphorus can at times increase production. Dortch and Whitledge (1992) show that silicate and phosphorus limitation as well as nitrogen limitation occurs at times in the plume of the Mississippi River. They conclude that silica limitation of desirable species growth

may be more important than nitrogen limitation of overall production. Smayda (1990) argues that anthropogenic loading may be shifting the balance between silicate and nitrogen in marine waters to levels more favorable to noxious phytoplankton species than species which are desirable for secondary productivity. These aspects of the control of production in Galveston Bay should be given further attention.

The nitrogen status of Galveston Bay

The nitrogen status for Galveston Bay can be generally addressed by comparing its nitrogen income to that of other Texas estuaries. **Figure 11**, from data in Tables 4.3.3 and 6.8.1 of Longley, 1994, compares nitrogen loadings and fisheries harvests in five Texas estuaries; the Sabine-Neches Estuary is omitted for lack of complete data. The estuaries are loosely ordered in the figure by loadings. Nitrogen loadings are weighted by residence time, in effect multiplying the input times the portion of a year the estuary has to process that input. This is a surrogate for placing inputs in the context of major loss rates. The comparison among the estuaries shows that Galveston Bay production is high, but not the highest. Mission-Aransas Estuary has higher harvests per area. The general agreement in the trends of harvest and nitrogen loading is an indication that increasing loadings do increase production in these bays. The Mission-Aransas data shows other factors are involved as well, and perhaps that an estuary can be very productive with lower loadings.

If phytoplankton production in Galveston Bay is limited by nitrogen availability, then we should see concentrations of dissolved inorganic nitrogen (DIN) much reduced in the bay as phytoplankton cells assimilate as much as is available to the limit of their capacity. Dortch and Whitledge (1992) suggest concentrations of DIN below 1.0 μm, 0.014 mg/l, indicate nitrogen limitation. We looked at Galveston Bay DIN concentrations in TWDB monitoring data, 1975-1989. DIN was calculated as the sum of nitrogen as ammonium, nitrate, and nitrite. Values reported below detection limits were assigned a value of 0.0. Galveston Bay grand average DIN is 0.14 mg/l. In the lower bay--samples below a Smith Point- Eagle Point line--DIN averages 0.077 mg/l. In 31% of samples taken in the lower bay, DIN fell below the limitation threshold. However, only in 1988 was the lower-bay average DIN at or below threshold. Therefore, while local nitrogen limitation occasionally occurs, DIN concentration data do not suggest Galveston Bay phytoplankton are commonly limited by nitrogen availability.

Figure 11. Nitrogen loading vs bay harvest in five Texas estuaries. Nitrogen loading per volume is weighted by hydraulic residence time.
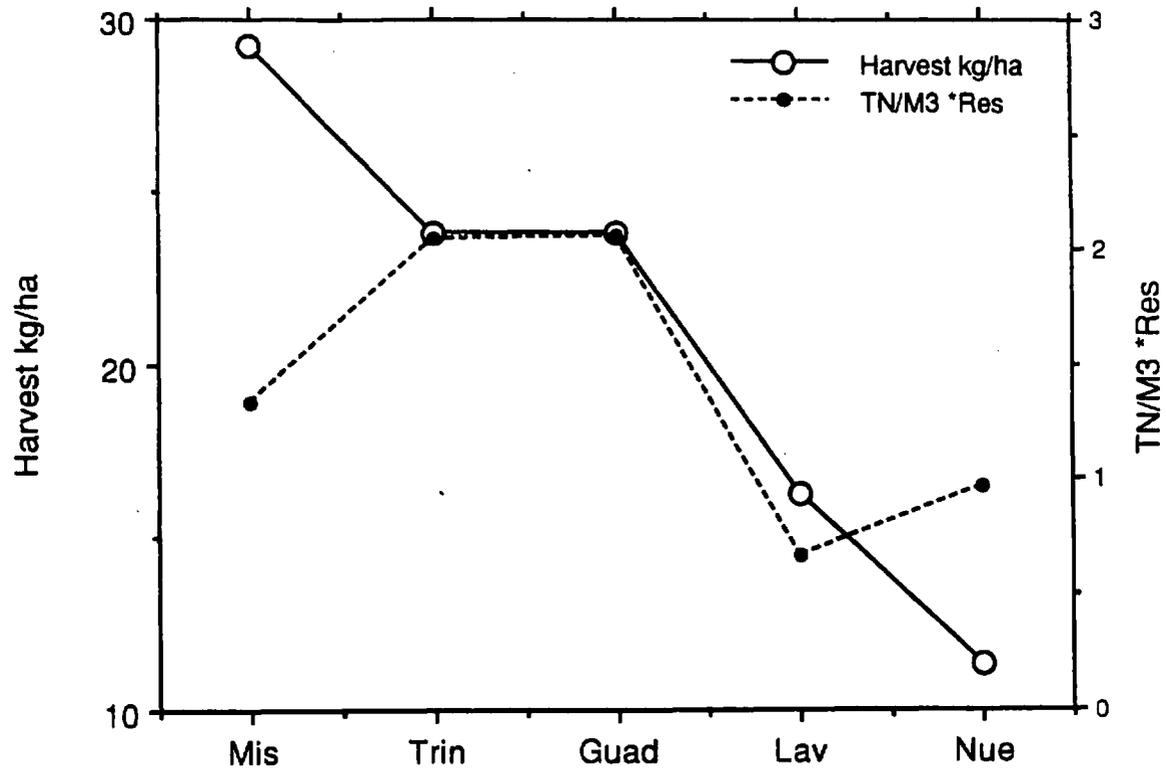
Deductions from the long-term nitrogen budget

The long-term budget presented above is a description of the bay responding to present conditions. We cannot use it alone to deduce requirements for the bay. However, by linking the way some of the components of the budget change in response to the nature of the bay ecosystem, as it responds to altered inputs, we may assess what level of changes to nitrogen inputs are consistent with a desirable system state. This is pursued quantitatively in the final section of this report, but some aspects of the question are investigated here.

Nitrogen inputs and desired bay conditions

A basis for a minimal nutrient requirement for the Galveston Bay system can be developed from knowledge that the system has been a productive system at least as long as chroniclers have been on its shores, although early nutrient loadings to the system were undoubtedly lower than they are at present. Jensen et al. (1991) suggest pre-modern stream nitrogen concentrations were on the order of 1.2 mg/l N, about half of present volume-weighted average levels. Therefore, the ecological system in pre-modern Galveston Bay revolved around lower average nitrogen inputs than exist at present. The biological system in the bay has changed since that time, partially in response to changes in nutrient inputs, partially in response to the many other changes which have occurred since the 1800's. However, the early productivity and biological community of the system can be considered as inherently characteristic of Galveston Bay as the present productivity and biological community, and we can assume these were associated with lower nitrogen loading.

From a nutrient budget perspective, we can ask how the system could be productive with lower nutrient loading given the high nutrient loss rates we see today. The answer is that internal biological and geochemical processes adjust to changing nutrient inputs, affecting internal storage compartments and recycling rates of nutrients. For example, in an estuary with a highly developed benthic community, including that associated with submerged aquatic macrophytes, more of the incoming nitrogen would be captured from the water column, leaving less vulnerable to be lost to transport. With lower incoming nutrient concentrations, the basis of primary production would shift away from being primarily planktonic to including more seagrass and other macrophytes (see Kemp et al (1983) for a discussion of the factors contributing to macrophyte decline in Chesapeake Bay). Positive feedback occurs in a system with increasing macrophyte production

through reduction in turbidity, which increases the depth macrophytes can colonize, further increasing nutrients contained in the macrophytes, their epiphytes, and associated community. From the nutrient budget perspective, lower levels of dissolved and particulate (including phytoplankton) nutrients in the water column mean lower export rates. Thus, over time, the estuarine system in essence adjusts some rates of loss to compensate for some reductions in inputs and other system changes. There are, then, two important points here for consideration in establishing a nutrient requirement: (1) the system has the capacity to adjust to lower nutrient inputs, and (2) some aspects of the adjustment (such as lowered turbidity which increases depths available to macrophyte colonization) are themselves desirable in terms of maintenance of an ecologically sound community.

Among the qualitative differences between present day Galveston Bay and the bay at the time of pre-modern lower stream concentrations is the possibility that present amounts of materials entering the estuary with floods are lower than they once were, because reservoirs serve as catchements. Therefore, we don't know if the true nitrogen economy of the bay in the past can be linked simply to estimated lower average stream concentrations. This is hard to gage. We could argue that other, new sources of nutrients to the system, such as wastewater, compensate for loss of flood-borne material. However, such new sources of materials are likely different in quality and timing of delivery from earlier inputs.

Other determinants of nutrient processes then and now have changed. Perhaps the greatest factor affecting the nutrient balance was the construction of major navigation channels that have enhanced the exchange of water between the bay and Gulf.

A minimal nitrogen requirement for Galveston Bay

A minimal nitrogen requirement is proposed here, based on the applicability of pre-modern nitrogen loading. We postulate that the reduction of historical nutrient input to the system due to detention of floods by upstream impoundments and other changes is of the same order of magnitude as increased nutrient inputs from wastewater. We also postulate that losses to fisheries harvest and escapement will be of the same order of magnitude past and present, assuming the biomass productivity of the system at historical nutrient loading rates is equivalent to today's productivity, though the species distribution of that production may have changed. Loading of nitrogen to Galveston Bay to characterize pre-modern conditions is based on stream concentrations of 1.2 mg/l N, after Jensen et al (1991) and median inflows,

assuming the present median flow can represent pre-modern flows. This
concentration times median inflow rates gives a loading of $14,944 \cdot 10^6$ g N/y. This
past median stream-flow load is proposed as a present minimal nitrogen load.

A minimal nitrogen loading is evaluated by considering what changes would
be required in the long-term nitrogen budget to accommodate lower inputs. To
achieve a balanced budget in **Table 17**, losses to outflows, burial, and denitrification
had to be reduced. Although fisheries might also be reduced, this exercise aims to test
whether other losses can be reduced and still within realistic limits while not
altering fisheries loss. To achieve a long-term balance with pre-modern nitrogen
inputs, lower-bay TN concentrations would have to average 0.54 mg/l, and burial and
denitrification rates would have to be much reduced. The lower bay concentration is
arguably within realistic bounds and in line with conditions in bays of the lower
coast. At present there are insufficient data for checking burial and denitrification.
This budget can be considered a statement of an hypothesis on the functioning of an
estuary with pre-modern nitrogen loading. Invoking a median flow basis is an
important safeguard here. The estuary will be expected to vary around the average
condition, and the average condition reflects buffering effects of storage during
high inflow as well as reductions during low inflow periods.

A phosphorus requirement could be derived from an estimate of pre-modern
loading in the same way. Without benefit of an estimate of pre-modern phosphorus
concentrations, an alternative is to propose a requirement based on the nitrogen
requirement and the nitrogen:phosphorus ratio typical of phytoplankton, 16:1 gram
atoms (Redfield et al. 1963). With this reasoning, $2092 \cdot 10^6$ g P/y would be a required
phosphorus input. However, given interactions of phosphorous and clay minerals
which appear so important in controlling dissolved phosphorus in an estuary
(Froelich, 1983), phosphorus bounds should be investigated for viability in the
context of expected adsorption/desorption equilibria.

A nitrogen-based lower bound on required freshwater inflows can be based on
pre-modern nitrogen loading, as a quantity referable to an historical bay condition.
This is a requirement for an inflow volume which provides a median pre-modern
nitrogen loading. Referencing the requirement to a median basis provides
assurance that normal variation in inflows, including times of greater and lesser
inputs, will result in a nitrogen sufficiency averaged over a number of years. From
Table 17, drainage basin inflows excluding wastewater inputs deliver $14,944 \cdot 10^6$ g
N/y to the estuary. To deliver this amount at present average stream concentrations
(median 2.49 mg/l), a target median 4,870,000 ac-ft annual inflow (gaged+ungaged-

Table 17. Galvéston Bay nitrogen budget based on streamflow volume providing premodern nitrogen loading, $6000 \cdot 10^6$ M^3 (4870000 acre-ft). First column is as calculated, the second column contains adjustments to * items in proportion to their relative magnitude to obtain a zero balance. Calculated transport losses assume lower bay TN 0.69 mg/l. Units are 10^6 g N/y.

	Calculated	Adjusted
Inputs		
GMD Streamflow	14944	14944
Wastewater	7300	7300
Total Drainage Basin	22244	22244
Inward Gulf Tide Entrainment	1749	1749
Direct Rain	700	700
Nitrogen Fixation	560	560
Loses		
Net Export to Gulf*	4226	3330
Loss to Gulf Tide Entrainment*	21553	16984
Loss to Fisheries	1065	1065
Loss to burial*	1235	973
Loss to Denitrification*	3680	2900
Total In	25253	25253
Total out	31759	25253
Remaining	-6506	0

diversions) would be needed. This annual inflow target is near a 75% exceedence inflow, or near the 25% quartile of the distribution of annual drainage basin inflows. This inflow quantity could be used as an inflow constraint in the TWDB TxEMP optimization model used to reconcile various aspects of estuary inflow requirements and water resource demands.

Establishment of an inflow constraint based on nutrient delivery assumes tributary nitrogen concentrations will remain substantially as at present. Any change should signal a need for re-evaluation.

Discussion of requirements

The proposed minimal nitrogen requirement does not necessarily deliver loading to maintain the status quo of the latter 20th century, although it may maintain production of desirable species. However, the status quo may not be the best definer of a healthy bay environment. During the latter 20th century there have been declines in submergent vegetation (Adair, et al. 1994) and occurrences of anoxic and hypoxic bottom water conditions (TWDB CDS data, Ward and Armstrong, 1992). Therefore, there are aspects of the present environmental state which can be identified as problems, problems possibly linked or exacerbated by a high nitrogen loading rate.

A nitrogen loading constraint has been suggested here based on conditions thought to have previously existed in the estuary. The budget style analysis of the degree to which the estuary would have to change shows the change is not inconceivable, though it depends on system accommodation which requires time, to develop. Further evaluation of the proposal could therefore include evaluation of the process of system change and the desirability of system change in response to reduced loading conditions.

Section II. Assessment of Galveston Bay Nutrient Requirements to Support Biological Production

This section covers the development of nutrient requirements for the Trinity-San Jacinto Estuary based on the needs to support biological production. The focus is on the base of the food chain, phytoplankton and submersed macrophytes in the waters of the estuary. This analysis makes use of an extended database on community production and respiration in the estuary developed from the application of free-water diurnal curve methods. Estimates of the nutrient requirement for an estuary to sustain observed primary production are most likely to be realistic if based on a set of productivity data long enough to span nutrient loading variation over a significant span of time. The application of the dissolved oxygen diurnal curve technique to the TWDB Datasonde long-term, high-frequency data set currently offers the only data meeting that criterion for Galveston Bay.

Introduction, Application of Diurnal Curve Method to Galveston Bay

The dissolved oxygen data available for Galveston Bay includes a series of month-long deployments of Hydrolab Datasondes over the 1987 - 1994 period. These instruments recorded temperature, pH, dissolved oxygen, conductance, and salinity every 90 minutes routinely, with some data at 60 or 120 minute intervals. Our strategy was to use as much of the Datasonde database as possible, including records more recent than the end of the project period, to develop productivity data. This provided a greater opportunity to develop a meaningful relationship between production and nutrient loading. Production measurements made after 1990 were paired with estuary inflows and nutrient loading estimated from available gaged inflow and other data.

Choice of sites

From 1987 through September, 1989, there were only two Datasonde sites in Galveston Bay: Dollar Point and Trinity Bay (**Figure 12**). After May, 1990, three additional sites were added: Redbluff, East Bay, and Bolivar Roads at

Figure 12. Datasonde locations in the Trinity-San Jacinto Estuary
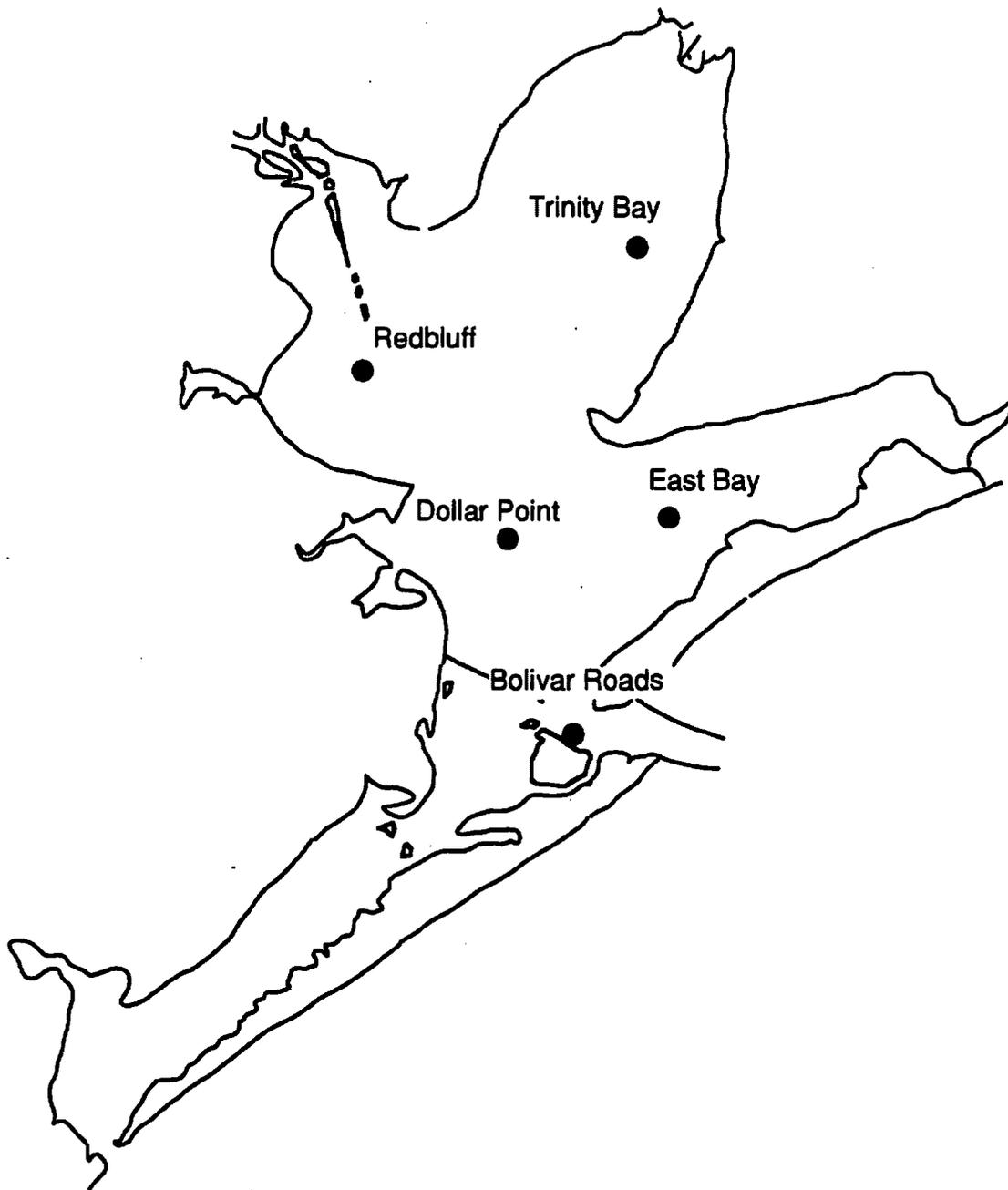
Pelican Island. Because instrument sites had been chosen originally to record
salinity variation in open areas of the bays, we had concerns that the
application of the diurnal curve method might not be considered valid for all
these sites. The method depends on the assumption that all changes in the DO
concentrations at a site are dependent on local processes. At some sites, tidal
movement of water is great enough to potentially violate that assumption: a
Datasonde could be measuring one water mass in the morning and another
water mass in the evening. Bolivar Roads is an obvious site at which the
method should not be applied. Salinity variation at Dollar point show that
there is usually major water movement past that site too. East Bay and
Redbluff sites have periods during which plots of salinity variation indicate
movement of different water masses past the site. The Trinity Bay site seems
best situated to represent a uniform water mass. Yet, at all interior bay sites,
there were days for which application of the technique seemed to be
appropriate, and so only the Bolivar Roads site was excluded. Data selection is
discussed further below.

Methods

Dissolved Oxygen Data Handling

The TWDB Datasonde database consists of a series of month-long
instrument deployments, referred to here as records. Considering the
relatively long duration of deployment and the physical type of probe used,
concern for possible probe drift or degradation of probe accuracy is
justifiable. All data records were plotted and inspected for obvious problems,
such as apparent electronic data transfer glitches, probe spikes, and probe
failure. Independent field measurements of dissolved oxygen (DO) taken at
installation and retrieval of the instrument provided another reference for
data validity. Judging from patterns of dissolved oxygen variation recorded,
the quality of dissolved oxygen data seems to hold up well in some records,
while other records show a marked deterioration, reflected in reduced
amplitude and baseline (probe reporting lower concentrations than observed).
Two means of minimizing effects of probe fouling were implemented. First,
only the first ten days of each Datasonde record were used. Secondly, an
algorithm correcting DO data for drift was routinely applied. Using only the

first ten days reduced possibilities that artifacts of the correction routine would interfere with results, since corrections were typically very small during the early part of the record.

A program was written in FORTRAN to correct Datasonde DO records based on the algorithm illustrated in **Figure 13**. Corrections are dependent on comparisons between beginning and ending Datasonde and independent measurements. The correction algorithm was applied to all Datasonde records for which supporting independent field checks at installation and retrieval were available and applicable. Considerations governing application and acceptance of the correction are as follows:

(a) There were cases in which, because of instrument problems, independent field data were not available. Datasonde records in that category were evaluated graphically for consistency with expected seasonal DO levels and then used or omitted from analyses.

(b) There were cases in which the dissolved oxygen probe drifted out of range or to zero some days before the Datasonde was retrieved, negating the final field check for lack of correspondence. In these cases a correction could not be applied, and the record was graphically evaluated for inclusion or exclusion.

(c) In some cases, the initial field reading was taken more than an hour or so before the Datasonde was programmed to begin recording. In those cases the initial field check could be judged not applicable, especially when the record of DO or salinity variation indicated rapidly changing conditions. In many cases, corrections could still be implemented, when calibration record or other information indicated the initial Datasonde reading was valid.

Correcting the DO data does affect the final results of diurnal curve calculations to a moderate degree. Wilcoxon signed-rank tests (Dixon, 1988) were run for all Trinity Bay records for which corrected and uncorrected data could be compared. Comparing ten-day average results, these tests show corrected records produce slightly higher diffusion and production rates than uncorrected records (**Table 18**). However, the differences are not so large as to suggest further conclusions will hinge on the use of the correction procedure.

Figure 13. Dissolved oxygen data correction algorithm

Input field installation DO, OBS1
Input field retreival DO, OBS2
Input constant correction DO, CONST
Read in datasonde records

First Datasonde DO reading = D1   Last DO reading = D2

Is CONST > 0.0 ?

YES

Add constant to
all DO records

BEGIN = OBS1 - D1   END = OBS2 - D2   DIFF = END - BEGIN

Are (absolute values) BEGIN & END
both < 0.50?

YES

Do not apply other
corrections.
Exit

NO

Is abs(BEGIN) > abs(END)
or is DIFF < 2.0 ?

YES

Apply linear Correction

xnum= total number of records

Di is datasonde DO
change=DIFFf/xnum

DOerr=BEGIN

Do i=1,xnum
Di=Di+DOerr
DOerr=DOerr+change
end do

NO

Apply exponential correction

find r from
END = BEGIN * exp(r*xnum)

Di is datasonde DO
error starts as abs(BEGIN)
change starts at 0.0

Do   i=1,xnum
Di=Di+BEGIN+change
change=error*exp(r*i)-error
end do

Table 18. Comparison of parameters of community metabolism from corrected and uncorrected dissolved oxygen records, Trinity Bay.

	Uncorrected	Corrected	P *
Diffusion constant	0.934	1.301	.041
Respiration	2.732	5.211	.253
Production	1.303	1.724	.052

* Wilcoxon signed ranks test

Diurnal Curve Considerations

Calculations involved in applying the diurnal curve method to non-flowing waters can be briefly summarized:

k=diffusion constant, $g/m^3/hr$

s_e=DO saturation deficit at sunset, %

s_m=DO saturation deficit at sunrise,%

q_e=rate of change in DO at sunset, $g/m^3/hr$

q_m=rate of change in DO at sunrise, $g/m^3/hr$

r = rate of respiration, $g/m^3/hr$

gp = gross production rate, $g/m^3/day$

dr = daily respiration, $g/m^3/day$

The diffusion rate k is calculated as $\quad k = 100 \cdot (q_m - q_e)/(s_m - s_e)$

Respiration rate calculated for the day as $\quad r = k \cdot (s_e/100) - q_e$

dr is $24 \cdot r$

The diurnal curve of DO rate of change is adjusted, subtracting k

gp is calculated from integral of adjusted DO rate of change curve from sunrise to sunset, adding respiration summed sunrise to sunset.

An assortment of applications and tests of the diurnal curve method to the metabolism of waters have been published since the method's utility was demonstrated in shallow Texas estuaries by Odum and Hoskin (1958), and Odum and Wilson (1962). Several of these papers present findings germane to the present application. McConnell (1962) tested the diurnal method in microcosms and determined that metabolic rates determined thereby were reasonable. He also proposed that for non-flowing waters, equivalent results could be derived by applying the basic assumptions of the method to changes in oxygen concentrations measured at sunset, sunrise, and the following sunset--referred to as the three-point method. Abbott (1967) pointed out the sensitivity of the method to the value of the diffusion constant, K, calculated in

the procedure, and suggested an approach to calculating K less subject to variation. Abbott basically combined measurements from two sequential days in determination of K. Welch (1968), concerned with stratified ponds, stated that diffusion constants calculated by the diurnal method seemed to overestimate diffusion in some systems. Welch applied values of K from 0.01 to 0.5, based on wind speed. He suggested the diurnal curve method would not be sensitive enough for oligotrophic waters. Whitworth and Lane (1969), applied the original method and the three-point modification to pool microcosms, with an arbitrary 0.1 diffusion constant. They found significant differences between the results of the full method and the three-point modification. Juliano (1969) performed various experiments to test various methods of determining the diffusion constant applicable to waters of the Sacramento estuary. Using the diurnal curve method, he found K to vary in an approximately linear fashion from 1.0 at a wind of 4 mph to 3.0 at winds near 24 mph. Juliano also provided a conversion between diffusion in units of $g/m^2/day$, and units more common in engineering contexts. Hornberger and Kelly (1975) stated that determinations of K and respiration using diurnal curve methods will likely be unstable due to noise in the series of measurements made for the procedure. They and others suggest application of mathematical techniques to smooth the data and produce more stable parameter estimates (see Madenjian et al., 1990; Chapra and Di Toro, 1991). In the present investigation, calculations were based on the technique as described by Odum and Hoskin (1958). The only mathematical techniques suggested which appear to offer substantial improvement to the accuracy of the method, those of Madenjian et al., for respiration rate, do not seem amenable to application to a large number of days.

Rate Calculations

Based on the instructions found in Odum and Hoskin (1958), the diurnal curve method was programmed in FORTRAN for solution. Some steps were taken to facilitate calculations. Slopes of rate curves were calculated using three-point central differencing. Samples corresponding to times of sunrise and sunset were determined by choosing sample times closest to sunrise and sunset times determined from astronomical calculations. Gross production was determined as the area under the rate of change curve (adjusted for diffusion) from sunrise through sunset plus respiration during that period.

Preliminary results showed that for many days, calculations resulted in negative diffusion, or negative respiration, or negative production, or rates higher than are realistic. These days are appropriately excluded, as discussed below. However, we wanted to test the calculation procedures to insure some days were not excluded unnecessarily. These tests were also conducted to check the robustness of the results. Odum and Hoskin suggest calculation of the diffusion constant and metabolic rates may be somewhat sensitive to the choice of morning and evening observation times from which values are taken for calculation. Others have indicated that diffusion and respiration may be slightly overestimated in calculations based only on differences between evening and morning dissolved oxygen if night-time respiration rates begin high and decline. Data from Grobbelaar and Soder (1985) shows algal respiration does fall during the night, although it is not clear whether community respiration would also decline. Tests run on Redbluff data compare results based on single points and averages of two and four readings. Using parameters based on the average of two or four points seemed to reduce numbers of wild results, which include negative rates or rates an order of magnitude higher than expected. Averages of daily rates do not show large differences between the methods used (**Table 19**). Here, morning and evening values used in calculations are each averaged from two adjacent readings: sunrise and pre-sunrise, sunset and post-sunset (**Figure 14**).

Since respiration and production calculations both involve adjustments for diffusion, accepting dubious values of diffusion reduces the confidence which can be placed in the other calculated rates. A test was made of the efficacy of replacing calculated K values higher than the means with the site mean value. In addition, values of K between -0.5 and 0.0 were replaced by their absolute value. For Redbluff, this procedure resulted in many more days fitting criteria for acceptable data. However, both respiration and production derived from the modified calculations were lower than those calculated using unmodified K's. Since the calculation of respiration is a function of K, selecting K arbitrarily easily injects observer bias into the results. Determination of K based on other means, such as wind speed, should be explored further, at least to provide criteria for acceptance of calculated values. The averages of K's calculated for each site were 1.81 (Trinity Bay), 1.45 (Redbluff), 1.98 (Dollar Point), and 1.18 (East Bay). These seem to fit

Table 19. Comparison of daily rates calculated from one or averages of two-four points after sunset and before sunrise.

Rate	Single	vs	Two-point	p	Two Points	vs Four Points	p
Respiration	6.34		6.19	n s	8.59†	9.78	n s
Production	2.42		2.25	*	2.33	2.49	n s
Diffusion	1.12		1.60	*	1.59	0.79	n s

* $p < 0.05$, Wilcoxon signed ranks

† Differences between rates for two-point case arise due to differences between data sets as different data pairs are included or excluded using rules for inclusion: $0.01 < diffusion < 10.0$, respiration and production > 0.01.

Figure 14.   Adjacent measurements averaged in calculations
of diffusion and respiration rates.
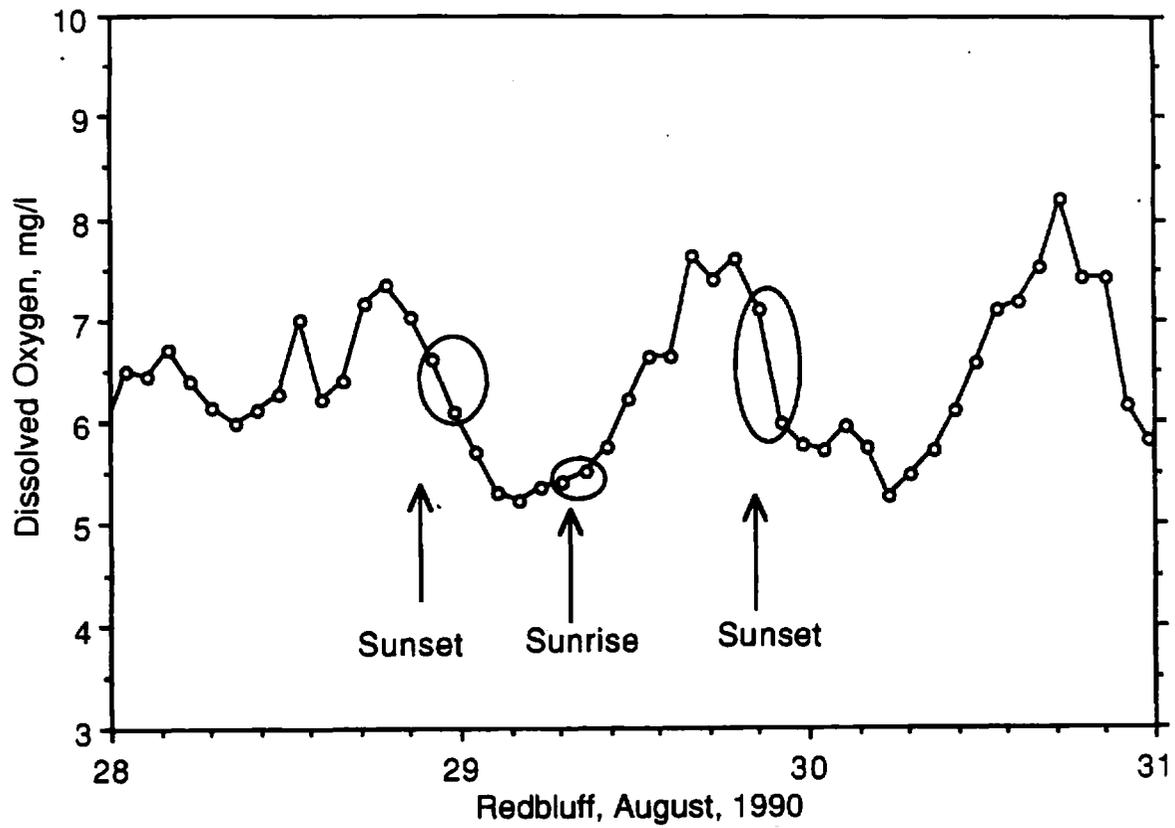
within Juliano's (1969) wind vs K relationship, given the usual windiness of the bay.

Another variation in calculation procedures which seemed likely to provide more robust estimates of K, compatible with the original method and not requiring additional data, follows the suggestion of Abbott (1967). Calculations determining K and respiration for a given day were based on two day's data. As **Figure 15** indicates, this procedure produced more days for which results were within bounds considered valid. However, the two-day modification did not bring more diffusion rates within a moderate range. Averaging data used in the diurnal curve method across adjacent days may be a means of producing reliable results, where day-to-day variation in rates is not the subject of study. This was not developed further for this project.

Automating the calculation of community metabolism from the diurnal oxygen curve has the great benefit of allowing the procedure to be applied to years of collected data, but has the drawback that the methods are applied inappropriately to days which do not fit the assumptions of the method. Among the sites and dates, the data contain days where salinity change indicates movement of water masses, or vigorous tidal action. During these days, Datasonde data would likely not be charting oxygen changes within water masses which shared a common metabolic history. At other times meteorological events, such as high winds and clouds associated with frontal passage, may interfere with or upset the typical diurnal oxygen cycle. Beyond accepting or rejecting an entire record of deployment, no prior selection was made of days within a Datasonde record for which assumptions inherent in the diurnal curve method were likely met. An assumption made in evaluating the results of the diurnal curve calculations is that days for which unrealistic values were calculated (rates negative or diffusion is greater than 10.0) are days for which the diurnal curve method is not suited. The corollary is that days for which diffusion, respiration, and production values all lie in a reasonable range are suited to the diurnal curve calculations. It is likely that neither the assumption nor the corollary is entirely true. More detailed work may result in successful results from days which are here excluded, or exclude some data which only fortuitously appears valid.

Cases were selected from diurnal curve calculations to stand as valid measurements if (a) for a given day, diffusion, respiration, and production all are positive, and (b) if diffusion is less than 10.0 g O_2/m^3/hr. The selection of

Figure 15. Diurnal curve results calculated from averages of sunrise and sunset rates over two days compared with standard method results.
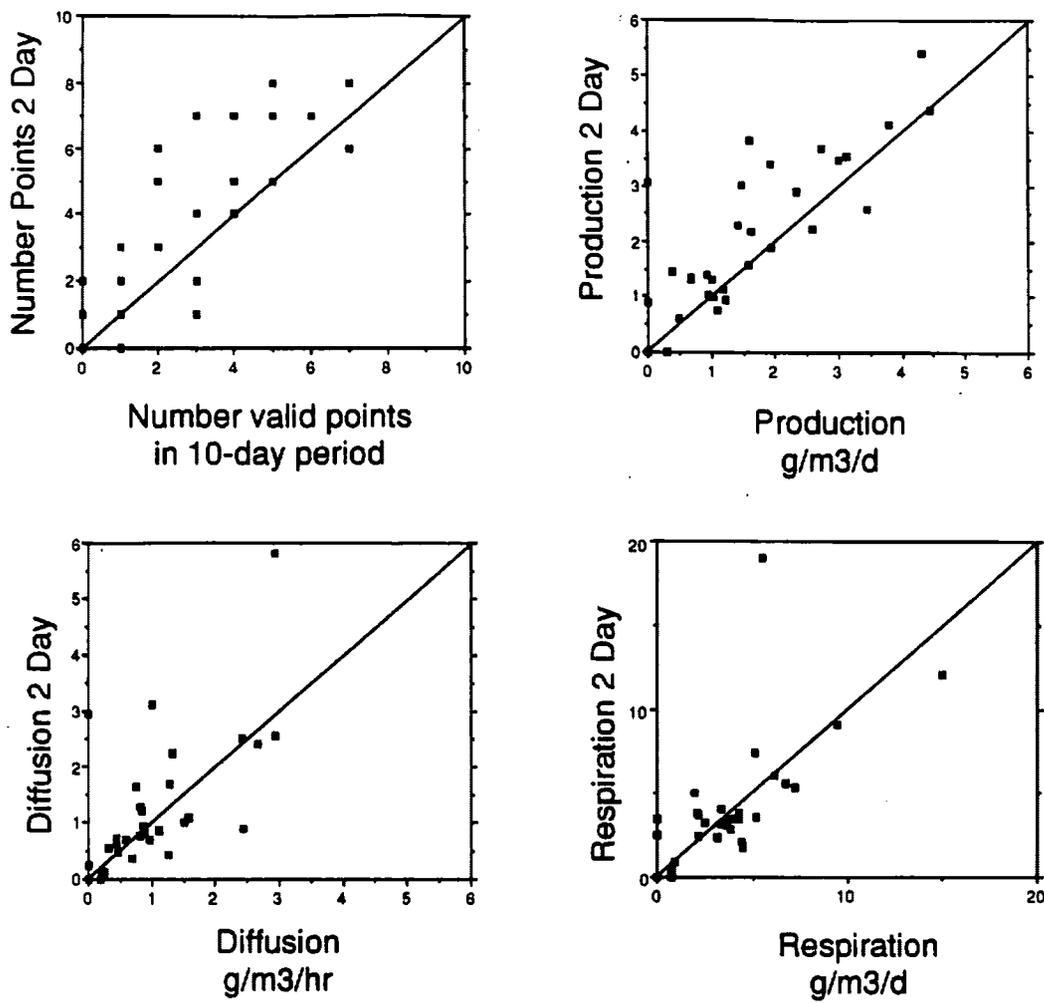
an upper criteria for diffusion rates is somewhat arbitrary. Rates above the 1 - 3 g O_2/m^3/hr range reported by Juliano (1969) may well be appropriate for the depths, wind, and currents of Galveston Bay. However, in this data set, days with diffusion rates calculated to be greater than 10.0 were generally days for which respiration or production rates were much higher than those of neighboring days, often by an order of magnitude. Approximately 5% of days with non-negative data were discarded with this criteria.

The average of valid results from each ten-day data set is here taken to be an appropriate measure of production and respiration for that 10-day span of time. A number of factors contribute to daily variation in metabolic rates. For forthcoming analyses, rates were desired which represented a time span long enough to average away the local influences so that the larger influences could be shown.

Results of Community Metabolism Analyses

Evaluation of Rates Derived from Diurnal Curve Calculations

Diurnal curve calculations yielded nearly 480 measurements of community production and respiration over the study period. **Figure 16** shows availability of rate measurements for each Galveston Bay site.

In **Table 20** are average rates of gross production and respiration calculated for Galveston bay sites in units of O^2 and C, using a 1.2 photosynthetic ratio. Odum and Hoskin (1958) suggest that results of diurnal curve calculations should be left in terms of oxygen, not converted to carbon. Bougis (1976) refers to studies which suggest that the ratio of oxygen released to carbon assimilated may vary between 1.05 and 1.6, depending on the type of carbon compound predominating in synthesis and whether nitrogen sources are predominantly ammonium or nitrate, and suggests 1.2 is an appropriate general figure. Raine (1983) considers the effect of intercellular reduction of nitrate to ammonium on the photosynthetic quotient, and uses 1.25 for comparative purposes.

Production and respiration rates reported by Odum and Wilson (1962) for selected sites and reported by Armstrong and Hinson (1973) for Galveston Bay sites using light-dark bottle method and from microcosms are shown in **Table 21**. Trimmed ranges from the present study are also presented to

Figure 16. Dates with measurements of diurnal metabolism
for each Galveston Bay site
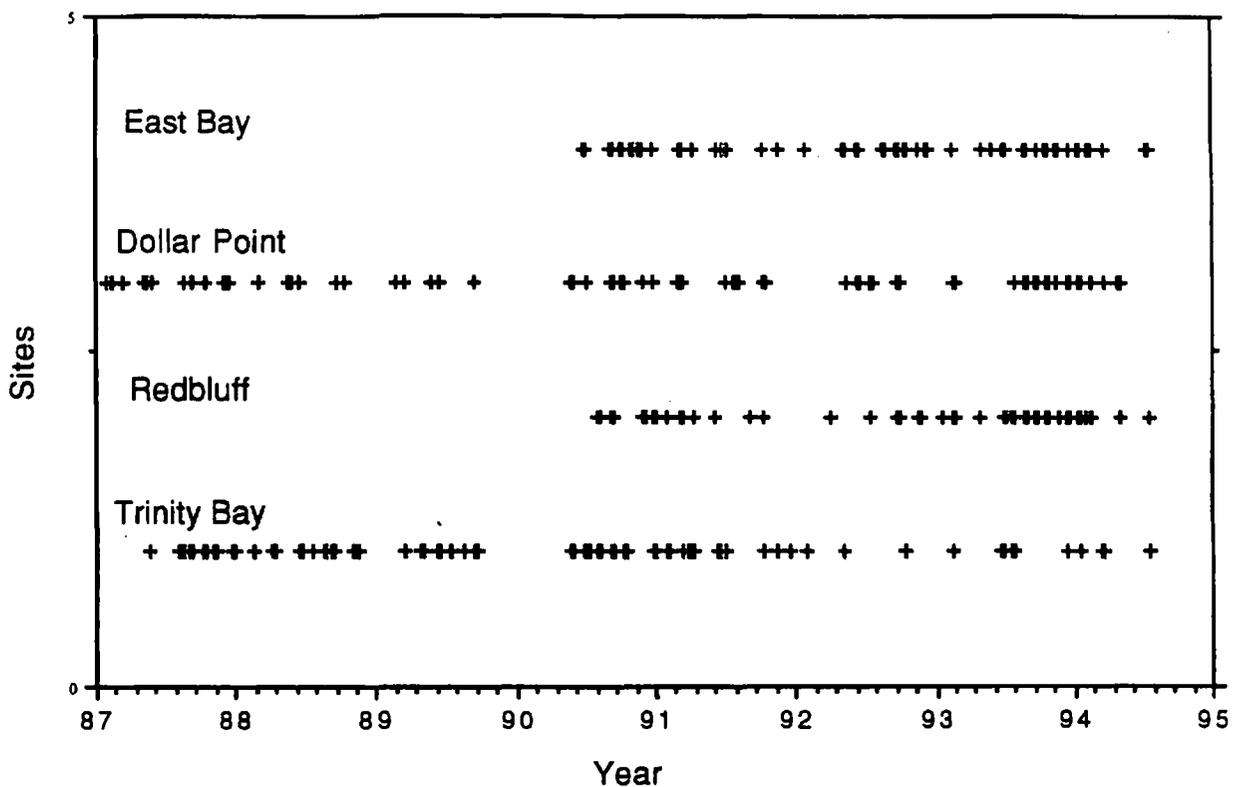
Table 20. Average rates of gross primary production and respiration for sites in the Galveston Bay system.

	$g\ O_2/M^3/d$	$g\ O_2/m^2/d$	$g\ C/m^3/d$	$g\ C/m^2/d$
Production				
Trinity Bay	4.5	11.1	1.42	3.5
Redbluff	3.3	8.9	1.03	2.8
Dollar Point	5.0	15.2	1.56	4.8
East Bay	3.7	6.8	1.17	2.1
Respiration				
Trinity Bay	6.6	16.1	2.1	5.0
RedBluff	4.6	12.5	1.4	3.9
Dollar Point	6.2	18.8	1.9	5.9
East Bay	4.0	7.4	1.3	2.3

Table 21. Production and respiration rates reported here and reported for Texas bays by other investigators.

	Production g $O_2/M^2/d$			Respiration g $O_2/M^2/d$		
This Study. Ranges*						
Trinity Bay	1.8	-	27.7	2.1	-	47.7
Redbluff	2.1	-	18.9	2.2	-	27.7
Dollar Point	3.4	-	29.9	2.2	-	41.5
East Bay	1.7	-	15.0	.9	-	15.6
Armstrong and Hinson, 1973	Light-dark	bottle				
Trinity Bay Jan.27, 1972	0.76			5.2		
Trinity Bay April 27, 1972	2.25			6.17		
Trinity Bay July 25, 1972	9.00			9.87		
	Microcosm diurnal curve					
Trinity Bay (station 26)	1.31			1.35		
Near Redbluff (station 22)	1.52			1.62		
East Bay (station 29)	2.04			2.05		
West Bay (station 14)	1.10			1.44		
Odum and Wilson, 1962	Diurnal curve					
Tres Palacios Bay June 17, 1957	3.8			5.8		
Tres Palacios Bay, July 14, 1960	12.5			0.5		
Lavaca Bay, June 17, 1957	·1.0			7.8		
Lavaca Bay, July 15, 1960	10.4			4.0		
Aransas Bay, May 19, 1957	6.1			11.5		
Aransas Bay, Oct. 20, 1957	6.1			7.8		
Aransas Bay, July 11, 1960	6.8			10.9		
Corpus Christi Bay Jun. 21, 1957	9.4			11.8		

*   10% - 90% trimmed ranges

indicate typical variation observed in daily data. Rates reported from Odum
and Wilson are representative of open water areas in major bays. They report
higher rates over seagrass beds and in waters receiving some influence from
waste discharges. The comparisons indicate that rates we report for the
Galveston system are consistent with the range of rates reported for other
Texas bays. Typical variation in rates for any given site make difficult a direct
comparison between the present data and those of Armstrong and Hinson.
Figure 17 displays seasonal variation in production and respiration for
Trinity Bay.

Stockwell (1989) reports measurements of primary production in Nueces
Bay at a location very near a TWDB Datasonde installation. Datasonde data from
the Nueces Bay site were used to calculate community metabolism. Corrections
for dissolved oxygen drift have not been applied to Nueces Bay data, so the
results should be considered preliminary. Among the dates Stockwell
conducted production measurements, there were five near or within periods
covered by Datasonde deployments. Unfortunately, there were no dates for
which there could be direct comparisons. Comparisons are made between
Stockwell's data and diurnal curve results for the corresponding ten-day
period in **Figure 18**. The method employed by Stockwell was designed to
integrate production over depth for a major portion of the day, but not
including lower-light periods of sunrise and sunset, which the diurnal method
incorporates. Therefore, it might be expected that the C^{14} results extrapolated
to the full daylight period would give higher production than would the
diurnal curve method. There is also the difficulty of putting the C^{14} and O_2
techniques on an equal basis, in terms of actual carbon assimilated, as Raine
(1983) points out. With all these caveats, however, the comparison indicates
that diurnal curve measurements are similar to measurements derived from
C^{14} techniques.

Figure 17. Monthly averages of community respiration and production rates for Trinity Bay, 1987 - 1994.
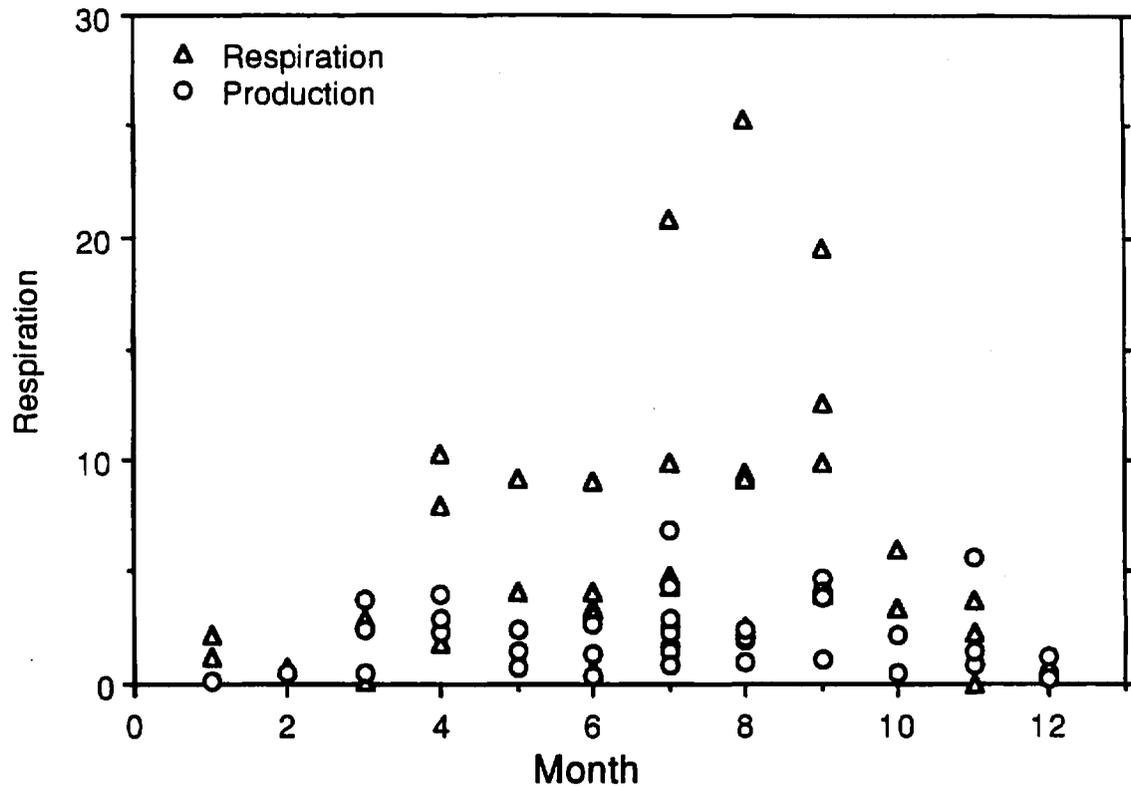
Figure 18. Comparison between production measurements in Nueces Bay
by ^{14}C and diurnal oxygen curve techniques.
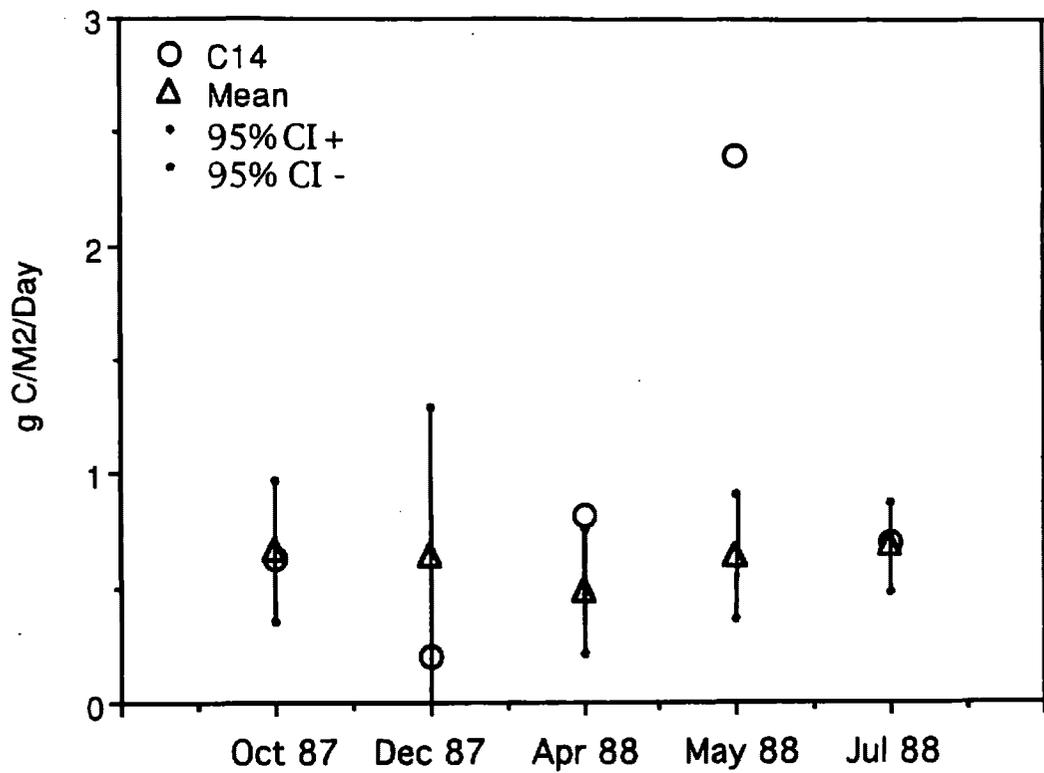
Production measurements by ^{14}C are integrated over depth,
but extrapolated from mid-day measurements to the full daylight period.

Relationships in the Data

The grand averages of rates displayed in **Table 20** indicate an imbalance, respiration exceeding production, which has been noted in many Texas Bays (Odum and Wilson, 1962). In Trinity Bay especially, and at two of the other three sites, respiration often exceeds production (**Table 22**). This implies a source of organic material to fuel the respiration in addition to that supplied by primary production in bay waters. The importance of processing of allochthonous organic material to the economy of a bay has been recognized in numerous other studies. This data reinforces an assessment of the importance of imported material for Galveston Bay, which is included in considerations below.

The respiration/production ratios observed in Galveston Bay sites decrease with distance from the Trinity River mouth (**Table 23**). This illustrates a productivity gradient from the river mouth to the Gulf. The importance of imported allochthonous material appears to decrease as you move away from the river to the sea, and *in situ* production becomes more important.

Correlations among the sites with respect to rates of production and respiration are not strong (**Table 24**). The processing of materials at upper-bay sites, heavily influenced by riverine deposition, might be expected to proceed at different rates than lower bay stations. The weak negative correlations of Trinity Bay respiration rates with rates at other sites may demonstrate that effect. Actually, all the correlations are based on sample sizes less than 30, and many are based on fewer than 20 pairs of valid data. Therefore these correlations may not describe real relationships. For the purposes of this project, lack of strong correlations among stations means that using one site to represent rates throughout the bay is not likely to produce realistic results.

Relationships between community metabolism, inflows, and nutrient loading

Environmental factors which influence production and respiration include parameters related to water clarity and light availability, temperature, and available nutrients. Influential factors also include those which would perturb the system physically, such as major flushing inflows. The present focus is on aspects related to the problem of relating productivity to freshwater inflow and nutrient loading.

Table 22. Numbers of days for which average net production (Net = P - R) was positive or negative during project period.

Site	Number Negative	Number Positive
Trinity Bay	117	58
Redbluff	64	46
Dollar Point	91	61
East Bay	54	56

Table 23. Changing average ratio of community respiration to production with station distance from the mouth of the Trinity River.

		R/P from average rates, Table 20	R/P from numbers of days Table 22
Increasing distance from river mouth	Trinity Bay	1.47	2.02
	Red Bluff	1.39	1.39
	Dollar Point	1.24	1.49
	East Bay	1.08	0.96

Table 24. Correlation of production and respiration rates among Galveston Bay sites.

Production

	Trinity	Redbluff	Dollar	East
Trinity	1.0			
Redbluff	0.34	1.0		
Dollar	0.35	0.14	1.0	
East	0.60	0.56	0.41	1.0

Respiration

	Trinity	Redbluff	Dollar	East
Trinity	1.0			
Redbluff	-.10	1.0		
Dollar	-.13	0.54	1.0	
East	0.47	0.43	-.27	1.0

Detailed hydrology was available only through 1990. Therefore, the volumes included in the table and elsewhere are drainage basin inflows generated from Trinity River and Buffalo Bayou gaged records (based on regression of those gages on total gaged+ungaged-diverted inflow, $r^2=.87$). Use of this relationship extends the estuary flow data through the period for which most Datasonde measurements were available. **Table 25** presents correlations of respiration and production with water temperature and freshwater inflow to the estuary. Correlations between rates and inflow for upper estuary sites, Trinity and Redbluff, were usually negative. This is most likely a reflection of the disturbance to the community from flushing flood flows and salinity fluctuations from estuarine to brackish in this area. The mid-bay and lower bay areas, represented by Dollar Point and East Bay, show production positively correlated to the previous flow history over a number of months. However mid and lower bay respiration is negatively related to longer-term cumulative inflows. East Bay respiration does not show a consistent correlation with whole estuary inflows.

Conclusions about community metabolism-inflow relationships should be tempered by limitations in the data. First, as shown in **Figure 16**, there are few rate measurements for the low-flow year, 1988. In addition, the period of record from which data were available to assess community metabolism is dominated by moderate to high inflows (**Figure 19**). Therefore, the correlations above may not be applicable to average inflow conditions in the Trinity-San Jacinto Estuary. However, the correlations may show that during this period of high inflows, the estuary community is at least destabilized, if not depressed from conditions at more moderate inflows. That apparent trend should be put in terms of a hypothesis and tested, for its implications for management.

Given the positive and negative correlations of production and respiration at each site with inflows, determining effects of inflows on the estuary as a whole from **Table 25** is not certain. Whole-estuary response could be based on the sum of volume-weighted average rates from each part of the bay, for all months. Within the data, however, there are few months for which all or many parts of the bay are represented. To avoid this problem, data were used to generate equations descriptive of production and respiration in terms of temperature and inflow (estimated gaged+modeled-diverted, as above), for each site. **Table 26** shows equations developed. Rates of

Table 25. Correlations of production and respiration at Galveston Bay sites with average daily temperature and gaged freshwater inflow. Monthly Galveston Bay gaged+ungaged-diverted inflows were estimated from Trinity River and Buffalo Bayou gaged inflows (regression, $r^2=.87$) and summed to produce the following flow categories: 1mo (the month of the data), 3mo (present and prior two months), 6mo (present and prior five months), 9mo, 18mo, and 36mo. (An * indicates correlations with one high outlier removed).

Production

	Temp	1mo	3mo	6mo	9mo	18mo	36mo
Trinity	.54	-.23	-.15	-.12	-.29	-.35	-.40
Redbluff	.54	-.55	-.32	.07	.24	-.13	-.13
Dollar*	.50	.08	.26	.32	.16	.11	-.08
East	.67	-.09	.27	.43	.13	.32	.08

Respiration

	Temp	1mo	3mo	6mo	9mo	18mo	36mo
Trinity	.53	-.09	-.03	.03	-.25	-.28	-.34
Redbluff	.28	-.43	-.11	.05	-.01	-.33	-.12
Dollar*	.44	-.04	.10	.09	.01	-.06	-.20
East*	.36	-.12	-.02	.04	-.19	-.25	.02

Figure 19. Gaged+ungaged-diversion inflows to the Trinity-San Jacinto Estuary
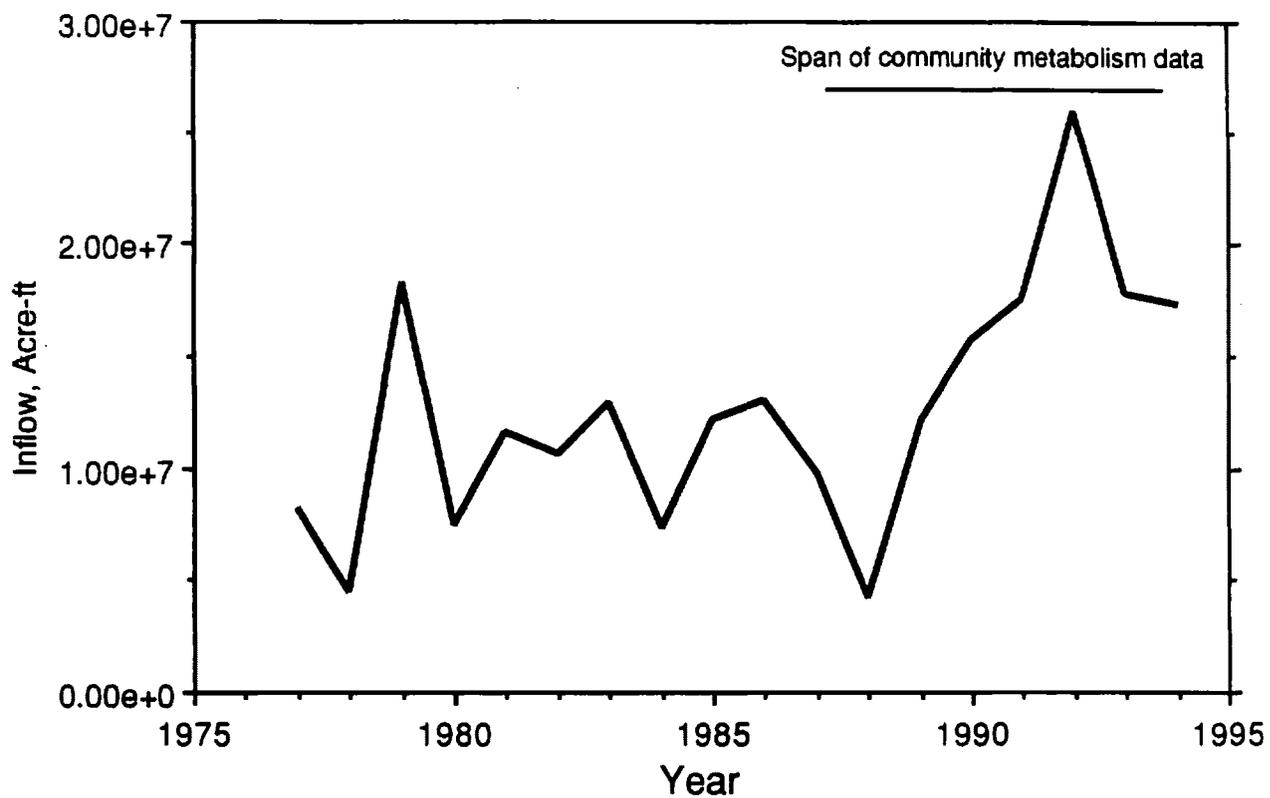
Table 26. Equations quantifying relationship of community production and respiration with temperature and freshwater inflow volume. Production and respiration as g O_2/M^3/day. Inflow is drainage basin gaged+ungaged-diverted 10^6 M^3. Inflows are cumulative one-month (F1), three-month (F3), etc, as in Table 25.

Production   P

Trinity Bay $P = 1.7804 + 0.2826 \cdot T - 0.9824E\text{-}4 \cdot F36$ .37
Redbluff $P = 1.8174 + 0.1181 \cdot T - 0.001055 \cdot F1$ .51
Dollar Point $P = -1.5034 + 0.2204 \cdot T + 0.1015E\text{-}3 \cdot F6^*$ .28
East Bay $P = -1.6118 + 0.2317 \cdot T + 0.3243E\text{-}4 \cdot F6^*$ .46

Respiration

Trinity Bay $R = \exp (0.17106 + 0.0985 \cdot T - 0.2809E\text{-}4 \cdot F36)$ .50
Redbluff $R = 4.14666 + 0.0971 \cdot T^* - 0.001616 \cdot F1$ .24
Dollar Point $R = \exp(0.7294 + 0.0591 \cdot T - 0.1647E\text{-}4 \cdot F36^*)$ .23
East Bay $R = 3.57104 + 0.2043 \cdot T - 0.1716E\text{-}3 \cdot F18$ .31

* Term does not contribute significantly (p > .05)

community metabolism for each area of the bay were then generated from these equations for all months, 1980 through 1994, using available hydrology and average monthly temperatures. These rates, volume-weighted, were then summed to produce whole-estuary rates. East Bay rates were used to represent West Bay. As the correlation table would indicate, the regression equations, while significant, do not explain a large amount of the observed variation. The impact of this is that rates derived from the equations have a smaller range of variation than is actually observed.

The annual estuary metabolism over the 1980-1994 period is illustrated in **Figure 20.** In this figure, the several years of very high inflows seem to depress community respiration markedly, and may have depressed rates of production as well. We can infer that the inflows would bring in a great quantity of dissolved and particulate organic material that should fuel heterotrophic activity, which should increase respiration rates. A reasonable conclusion from the rate drop would be that the flows were so great as to disturb the estuarine community. The figure also shows that there are no successions of low flow years in this data sequence. We might expect that bay metabolism in a low-flow year following a year of moderate-to-high inflows would be heavily influenced by production or processing of organics imported during the previous high-flow period. In fact, if flushing or turbidity associated with inflows are negative factors, we might expect that low-inflow years following high-inflow years would be periods of greatest production. The figure, however, does not show that from the generated rates.

Estuary Comparison

In the absence of a good mix of low volumes and high volumes of inflow in the period of study, it is worth exploring comparisons among bays as a means of estimating the behavior of a bay during a period of low inflows. Preliminary measurements of community metabolism were developed for a Datasonde in Mid-Nueces Bay, as discussed above. Nueces data are considered preliminary because corrections for probe drift have not been applied, but this should not substantially affect the comparison discussed here. The Nueces Estuary has a much smaller inflow volume per unit estuary volume than Galveston Bay (0.7 vs 4.7, annual inflow/bay volume). Nueces Bay metabolic rates show positive correlations with inflows (**Table 27**). It seems possible

Figure 20.   Annual Galveston Bay production and respiration,
estimated from regression vs Inflow
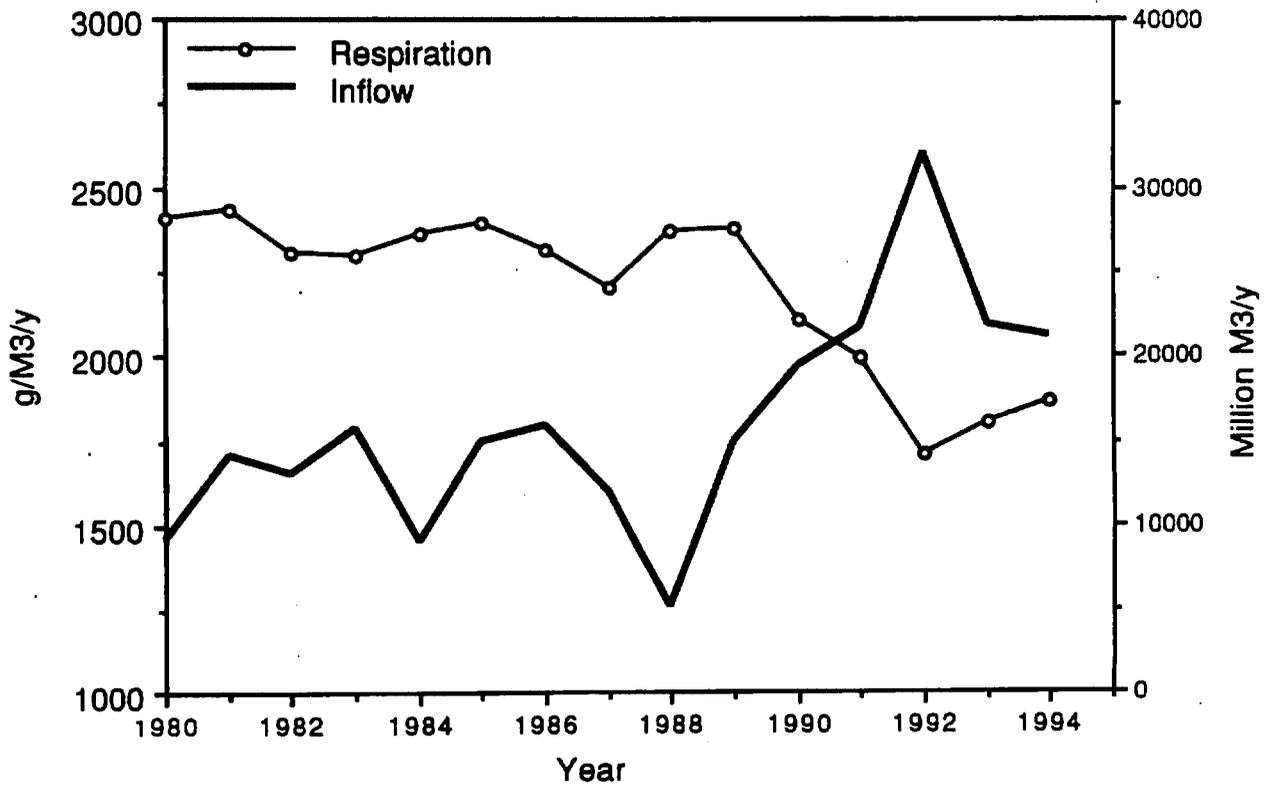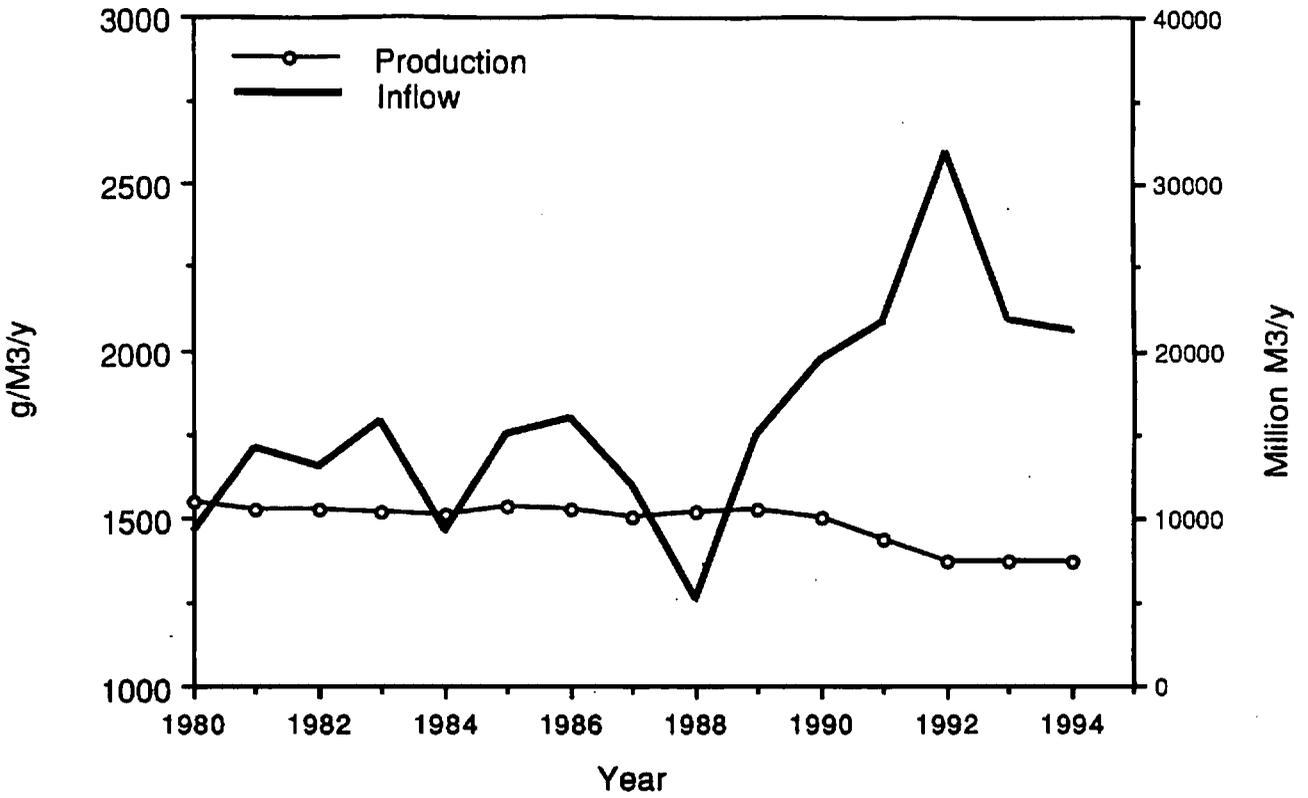
Table 27. Average rates (g $O_2/m^3/d$) of production and respiration in Nueces Bay, and correlations with temperature and gaged+ungaged-diversion inflows. Flow terms defined as in Figure 22.

	Mean	s.d
Production	2.28	1.15
Respiration	2.91	2.38

Correlations	Temp	1mo	3mo	6mo	9mo	18mo	36mo
Production	.57	.49	.54	.34	.03	-.01	-.19
Respiration	.16	.81	.54	.60	.22	.01	-.10

that Nueces and Trinity bays would form two halves of an inflow-rate response curve.

Productivity rates from Trinity and Nueces bays were compiled into annual production by applying measured daily rates from each month to each day of the month and summing all days. Missing months' data were estimated by interpolation between months or using the average rate for that month, where two or more adjacent months were missing. Annual community production are plotted vs. inflows in **Figure 21**. Nueces Bay data, the left-most three points, cover only enough inflow-range to define the tail of the curve. However, this figure shows that there is some potential for understanding relationships of inflows and production in general for bays of the Texas coast. This is developed further in the last section of the report.

Determination of Galveston Bay Nutrient Input Requirements to Support Productivity

A major implication of the weak or negative correlations between freshwater inflow volume and primary production over most of the estuary is that the current ratio between nutrient loading and primary production may not reflect the actual requirements for nutrient loading to support production. Negative aspects of the observed high volume inflows may outweigh the positive influence of the nutrient input. However, there are means of inferring nutrient requirements to support primary production aside from direct correlations.

Nutrient Demand and Supply based on Community Metabolism

Production data has been used itself to estimate the nutrient input required to support bay productivity, based on general nutrient content of organic matter produced (eg. Flint et al. 1983). This approach can also be developed for Galveston Bay. Where this approach has been used, it is often combined with estimation of internal nutrient supply, recycling. The relationship between these, however, can be misconstrued in terms of the long-term system requirements. To some extent, the interplay between nutrient demands from gross primary production and nutrient supply from respiratory processes is internal cycling which does not apply entirely to an

Figure 21. Annual community production, g O2/M3/y, in Nueces and Trinity bays, vs estuary inflow. Three Nueces Bay points are at lower left.
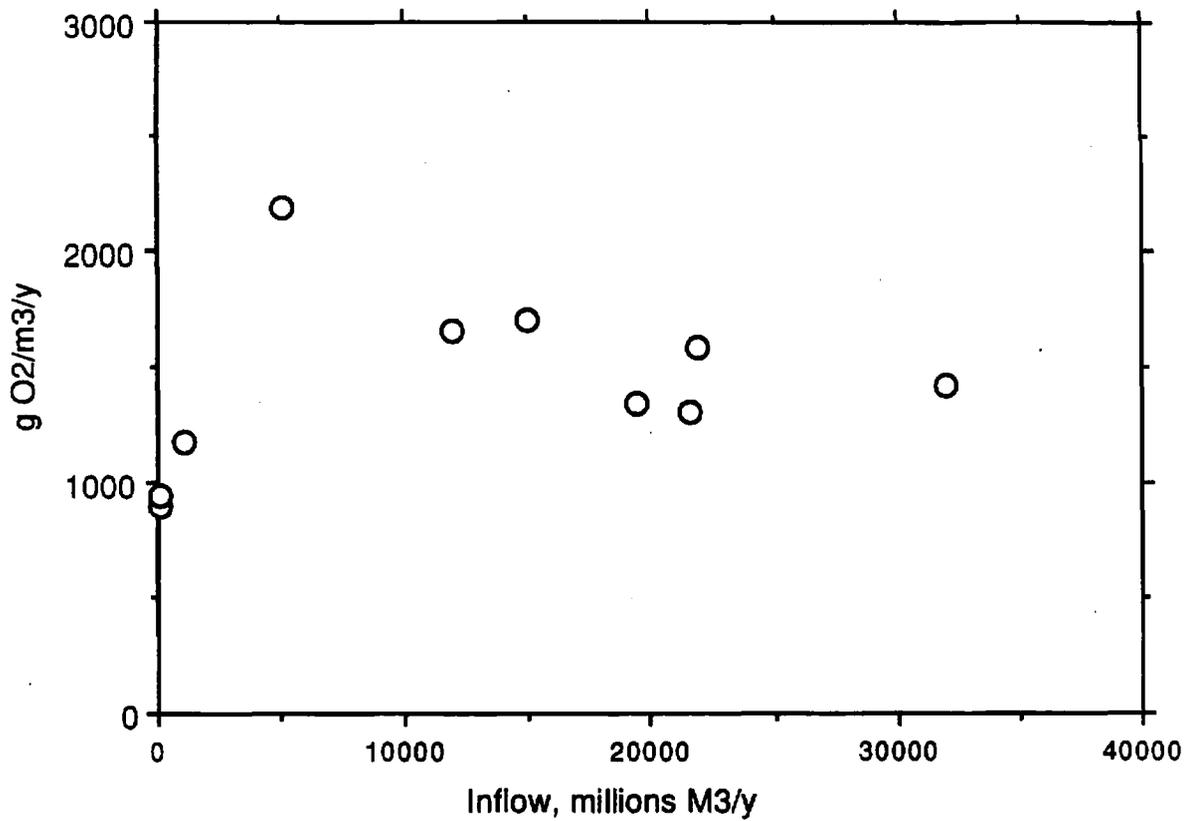
estimation of estuary nutrient requirements. However, there is some connection, and the process does involve the community metabolism data, so some discussion is helpful. **Table 28** presents the data and relationships and results discussed below.

Assuming that the carbon fixed in algal cells during photosynthesis is incorporated into saccharides, lipids, and proteins, in proportions generally found in cells, the amount of other nutrient elements required by cells to incorporate this carbon can be estimated as the proportion of those elements in the cell. Here, the C:N:P ratio of 106:16:1, gram atomic weights, is used, as commonly employed after Redfield et al. (1963). Per gram carbon incorporated, the phytoplankton are assumed to require 0.176 g N and 0.024 g P. This relationship between production and nutrient ratios applied to average Galveston Bay gross primary production results in requirements of $170 \cdot 10^6$ kg/y N and $23 \cdot 10^6$ kg/y P. If the other main source of carbon to the system, terrestrially derived TOC, also becomes incorporated into biomass through heterotrophy, this would require nitrogen also. Nitrogen demand to serve allochthonous carbon is calculated as above and amounts to $24 \cdot 10^6$ kg N/y, probably an overestimate, since some terrestrially derived carbon is refractory. In addition, remember that net production and the net requirement is only a fraction of these.

Respiration and inefficient consumption of the organic matter produced is occurring simultaneously with production. The nutrients released (=regenerated) from this heterotrophy supply nutrients to support the observed gross primary production. The first step in the use of community respiration to infer nutrient regeneration is to partition community respiration between benthic and water column processes. Sediment oxygen demand includes processes such as nitrification, which consume oxygen without the same implication of nitrogen release as biological respiration has, so the sedimentary ratio between oxygen consumption and nitrogen regeneration should differ from that in the water column. Zimmerman and Benner (1994) measured respiration rates averaging 0.006 g/M2/h in intact cores of Galveston Bay sediments. However, those 10 cm diameter cores were small enough perhaps to miss some macrobenthic contributions. Using in-situ chambers in San Antonio and Nueces bays, Montagna et al. (1989) report average benthic respiration rates of 0.06 and 0.04 $g/M^2/h$ respectively. Flint, et al. (1983) found respiration ranged from 0.04-0.10 $g/M^2/h$ in Nueces Bay

Table 28.   Nutrient demand and supply for production developed from community metabolism data.

Assumptions

Respiratory Quotient, CO_2/O_2	1.0 M
Photosynthetic Quotient, O_2/CO_2	1.2 M
Portion algal respiration of gross production	0.15
Cellular ratio N/C	16/106 g-at
Cellular ratio P/C	1/106 g-at
Water column NH_4 regeneration (Gardener et al. 1994): (μM NH_4/h) = -0.0085 + 0.114 · (μM O_2/h)	
Sediment ratio NH_4-N release/O_2 consumed (Gardener et al. 1994)	0.04
Bay area	1391. · 10^6 M^2

Gross Primary Production

Average bay production	4.2 g O_2 /M^3/d
minus algal respiration	3.5 g O_2 /M^3/d
per area	6.1 g O_2 /M^2/d
Production as C	1.9 g C /M^2/d
Demand for N	0.33 g N /M^2/d
Demand for P	0.05 g P /M^2/d
Bay N gross demand, per day	465 · 10^6 g/d
per year	170 · 10^6 kg/y
Bay P gross demand, per day	63 · 10^6 g/d
per year	23 · 10^6 kg/y

Nutrient Regeneration

Average Bay respiration	6.4 g O_2 /M^3/d
per area	11.1 g O_2 /M^2/d
Respiration assigned to sediment	2.4 g O_2 /M^2/d
N regeneration from ratio	0.1 g N /M^2/d
Respiration of water column	8.7 g O_2 /M^2/d
N regeneration from equation	0.32 g N /M^2/d
Total N regeneration	0.4 g N /M^2/d
Total P regeneration	0.1 g P /M^2/d
Bay N gross regeneration	574 · 10^6 g/d
per year	210 · 10^6 kg/y
Bay P gross regeneration	80 · 10^6 g/d
per year	29 · 10^6 kg/y

N regeneration/demand	1.24
P regeneration/demand	1.25

sediments. Hargrave (1969) reviewed respiration of sediments from a variety of aquatic habitats and found that much of the variation could be explained as a function of temperature alone. Applying Hargrave's function to the annual temperature cycle of Galveston Bay waters produces an average respiration of 0.03 g O_2/M^2/hr. For Galveston Bay, benthic respiration may be in a range of 0.05 to 0.10 g O_2/M^2/h. The higher value is applied in calculations. From benthic respiration, nitrogen regeneration can be calculated. A simple ratio is employed here, 0.04 g ammonia N to 1.0 g O_2 consumed, from Gardner, et al. (1994) studies of shelf sediments near the mouth of the Mississippi. This is similar to the ratio which can be calculated from data in Rossen (1986). Phosphorus regeneration was linked to nitrogen via Redfield stoichiometry. Actually, however, changes in salinity and sediment resuspension may play the major role in making sediment P available to primary producers (Pomeroy et al. 1972).

In **Table 28**, water column respiration is community respiration minus the benthic portion. This respiration includes everything from algal respiration to that of demersal fish and crustaceans. Typical studies of water column regeneration have focused on activities of specific species, such as copepods (Conover and Corner, 1968). For this analysis, ammonia regeneration accompanying Galveston Bay water column respiration was calculated by applying the relationship reported for waters off the Mississippi delta by Gardner et al. (1994).

From volume weighted average community respiration and the assumptions and further information above, water column and sediment regeneration supplies $210 \cdot 10^6$ kg/y N and $29 \cdot 10^6$ kg/y P. This supply is 120% of the demands of gross primary production. Although compiled for the year, these figures are most relevant to a short time scale. On any given day, the primary production may be supported entirely by regeneration. However, a substantial portion of regeneration on a given day is probably derived from the processing of that day's production. The real question in use of regenerated nutrients is how much production is supported by regeneration from nutrients accumulated in the system in previous years. The impact of regeneration in accounting for primary production over a longer time frame is limited by the finite pool from which water column and sediment nitrogen can be regenerated. Consider a sediment nitrogen content of 65 g/m^2 based on data compiled for nitrogen burial, a net flux from the sediment of 0.01 g/m^2/d,

after Gardener et al. (1994) as above, and using the rate of denitrification from **Table 12.** Based on that information, if sediment N were not replaced, Galveston Bay available sediment N would be depleted in 80 days. The upshot is that contrasting regeneration rates with gross production rates, at least in this situation with a negative net production, does not provide a basis for a nutrient requirement without further information.

We can explore the idea that the *net* yearly requirement of the estuary for nitrogen and phosphorus is related to the gross requirements stated above approximately as net biomass production is related to gross photosynthesis. It may be possible to arrive at an estimate of system biomass production from photosynthesis from trophic conversion efficiencies. Trophic efficiencies in trophic biomass conversion may be on the order of 15% in coastal systems (Ryther, 1969, Parsons and Takahshi, 1973, Table 29). If we can consider production of second or third order consumers as net production, net production would be .0225 to .0033 times gross production. Then, net nitrogen and phosphorus requirements would be in the range of $1/100^{th}$ and $1/1000^{th}$ the gross requirements. For nitrogen, this approach would result in 560-$3800 \cdot 10^6$ g/y TN needed to support Galveston Bay net production. These figures are 1/2-3 times the 1988-1990 nitrogen loss to fisheries, which indicates the trophic efficiency approach has potential. If the approach is taken to define nutrient needs from net production, then nutrient needs to support other system processes must also be considered in developing a system requirement. The weakness of adequately defining net production is, however, an order-of-magnitude problem.

Net system production is a quantity which is hard to measure. Defining it as the difference between community production and respiration is ambiguous when respiration commonly exceeds production. A rational alternative approach to the problem would be to assume that fish yield and escape represent net production, and match the contained nutrients against nutrient loading minus burial, dissolved exports, etc. And this is what has been done in the analysis of nutrient budgets.

Discussion on the Application of the Diurnal Curve Technique

Data presented in the application of diurnal curve techniques to measure production and respiration in Galveston Bay are generally if not definitively important to the evaluation of nutrient requirements of Galveston Bay. The

production and respiration data constitute a productivity history for several years. The large respiration rates clearly demonstrate that a substantial proportion of production of the bay depends on assimilation of organic material inputs from outside the bay. Stated another way, primary heterotrophs compete with autotrophs for available inorganic nutrients. A corollary to this is that accounting for the income of organic material to the bay may be as important under current conditions to understanding tertiary production in the bay as is accounting for nutrients which affect primarily phytoplankton production. Galveston Bay receives a much larger amount of organic material (as reflected in total organic carbon measurements) than other bays on an aerial or volumetric basis, 141 $g/m^2/y$ and 88.1 $g/m^3/y$ (Table 4.3.2 in Longley, 1994).

Evaluation of the data presented here on the heterotrophic metabolism of the bay should be tempered with caution: The assumption that respiration rates during the day are similar to those at night is an important determinant of respiration rates. Algal respiration rates during the entire 24-hr cycle may be highest at sunset, as suggested by data of Grobellaar and Soder (1985) and Fogg (1965, Table 14). Respiration rates of other components of the community should match activity cycles, and there are probably both diurnal and nocturnal peaks.

There are areas where further study could improve the estimates of nutrient requirements from community production and respiration data. The stoichiometry applied, relating atoms of C, N, and P, has a large impact on the ultimate results. Zimmerman and Benner (1994) report that as much as 20-40% of oxygen consumption by their sediment cores is due to nitrification, not oxidation of carbon. Therefore, a stoichiometric equation of sediment oxygen respiration to organic matter decomposition and hence nitrogen release would probably overestimate nitrogen release. It is unclear whether the sediment rate calculated here is an overestimate, however, since this portion of respiration was derived through the use of an empirical ratio. However, it should be relatively easy to collect data to provide a better relationship of benthic respiration to benthic ammonium release for Galveston Bay.

The use of community production data to determine nutrient requirements for Galveston Bay requires more information about the system. Specifically, adequate data are required on trophic connections and conversion efficiencies. Improvements would also require additional work on

generalization of production rates from point data to other inflow and seasonal regimes. Application of generally accepted trophic conversions to gross nutrient requirements does allow evaluation of general response of bay production to changes in nutrient loading. Application of a mathematical trophic model should allow more predictive use of this kind of analysis.

Section III. A Summary and Exploratory Model of Galveston Bay Nitrogen

Introduction

A dynamic model can test quantification of concepts and components of system behavior, such as embodied in the nutrient budget and inflow-production relationships. The purpose of this section is to present a model which incorporates major results of the previous two sections and to use the model to explore implications of the interactions of components for management of Galveston Bay environmental conditions. The model is built to match the level of resolution and system organization addressed in the preceding sections. That is, it is general in scope, dealing with the bay for the most part as one unit, and modeled from yearly inputs (the time step is quarterly). It is a device to test our understanding of system level factors and responses on a large scale.

There are other models, including the TWDB ESTECO model (TDWR, 1976), which build a system of interacting parts based on physical factors, chemical reaction rates, and physiological and trophic parameters. These models allow a focus on underlying mechanisms which determine system behavior. Even in these complex models there are by necessity simplifications. Constructing such a model requires decisions on which processes deserve most effort. It is hoped that the model presented here will show major areas where detail is really needed to understand how nitrogen inputs and dynamics effect the Galveston Bay ecosystem.

The STELLA® Gal Model

A model of components involved in the Galveston Bay nitrogen budget was developed using the STELLA II software for Macintosh (High Performance Systems, 1992). The software package allows a graphical construction of the model without mandatory user involvement in programming simulation

engine details. Model run modes for sensitivity analysis are easy to invoke, a key feature in its use in this context.

Figure 22 illustrates the model structure, making use of STELLA symbolic elements:

Rectangle: a stock or reservoir, accumulates material from flows, subtracting losses.

Circle: a converter, transforming material or information according to quantitative relationships.

Thin arrow: information transfer

Thick arrow with attached converter: flows of materials, incorporating information to effect changes in units.

Cloud: undefined sources or sinks.

There are basically two discrete segments of the model, a nitrogen stock segment and a primary production segment. Both segments respond to the same driving freshwater inflow. Although many other factors determine system behavior, this single factor has enough general effects to be a valid main focus, as well as being of management concern. The nitrogen stock segment derives loss from removal or biological production from the primary production segment. There could be a complementary incorporation of nutrient flows into the primary production segment, except that this is subsumed in the general inflow effect. Other possible linkages for this model structure are mentioned in the discussion.

Table 29 further defines the model, providing details on the relationships involved. There are a number of assumptions and conventions incorporated into these relationships which are important to understand why the model behaves as it does. Where not identified, nitrogen relationships are based on **Table 12**.

Bay content

The structure of the nitrogen portion of the model is a change in stock driven by various contributors to a total loading and various contributors to a total loss. With respect to the nitrogen budget section of this report, the *bay content* reservoir represents the pool of biologically available nitrogen in the bay: sediment, biota, and water column. The initial value is the sum of nitrogen in a 10-cm sediment layer, average water volume content, and

Figure 22. Diagram of STELLA Galveston nitrogen budget model.
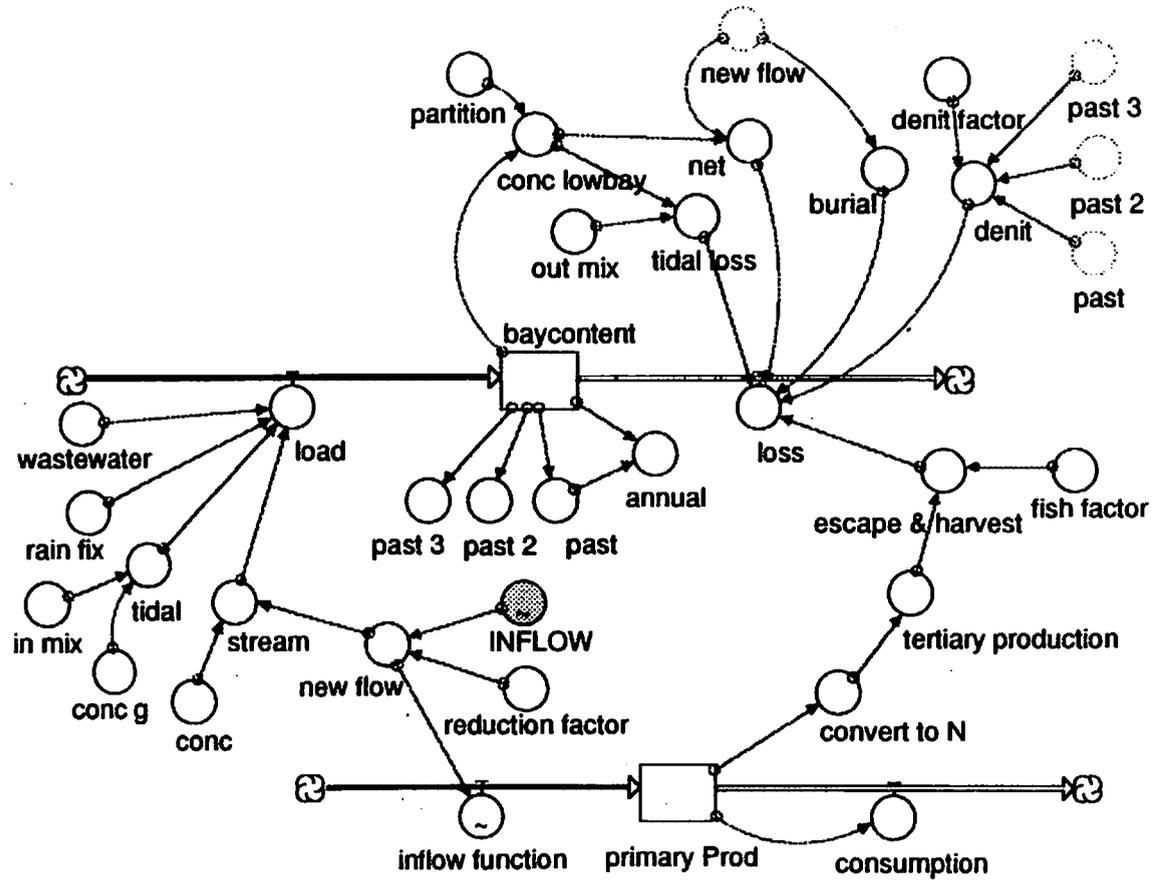
Table 29. Equations and data included in the Galveston Bay STELLA nitrogen model. Mass quanitities are in units 10^6 g N.

Model Element	Equation
baycontent	=baycontent (t/dt)+(load-loss)·dt
	Initial value 220,000
load	=stream+tidal+rainfix+wastewater
loss	=net+tidalloss+escape&harvest+denit+burial
Primary Prod	=Primary Prod (t/dt)+(inflow function-consumption)·dt
	initial value = 50000
inflows	inflow function from graph on new flow
outflows	consumption=primary prod
annual	= baycontent-past
burial	= (new flow ·0.1394+28.84)·2280/2585
conc	2.49 g/m^3
conc g	0.1 g/m^3 .
conc lowbay	= if(baycontent >0) then (baycontent/2409)·partition
	else 0.0
convert to N	=primary prod ·0.17
denit	=if years > 3 and (past+past2+past3 > 0) then
	((past+past2+past3)/3)·denit-factor + 370,
	else 370
denit-factor	=.0162
escape&harvest	=tertiary production·fish-factor
fish-factor	=0.7
in-mix	=out-mix - 0.02
net	=(-206.67+1.0622·new flow)·conc lowbay
new flow	=inflow·reduction factor
out-mix	=0.15
partition	=0.01
past	=delay(baycontent,1)
past2	=delay(baycontent,2)
past3	=delay(baycontent,3)
rainfix	=700+560
reduction factor	=1.0
stream	=newflow·conc
tertiary production	=convert to N·0.15·0.15
tidal	=in-mix·208520·conc g
tidalloss	=out-mix·208520·conc lowbay
wastewater	=7300
INFLOW	=input data set, yearly volumes, 10^6 M^3

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average fisheries harvest and escapement. Actual data on changes within Galveston Bay total biota and sediment compartments, are not available, and explicit inclusion of these compartments was not possible in the nitrogen budget (Section I). Instead, changes in these compartments were confounded with any error in calculations and listed as a remainder. The long-term budget exercise was focused on a time frame over which changes in internal storage would be minimal. Modeling, however, provides us the opportunity to consider the importance of cumulative year-to-year changes in these internal storage compartments. *Bay content* is the reservoir which holds that accumulation. Of course, *bay content* also contains whatever residual error accumulates from application of the relationships which drive the model. We thus need to make an assumption about the importance of error here. We assume the error is not biased negative or positive, and that it is not large enough to determine results. *Bay content* carries over the system's previous history in present nutrient dynamics.

Primary Production

For simplicity, physical and nutrient effects of freshwater inflow are not separated here, in the way production is generated from freshwater inflow. We expect, therefore, that production should increase with increasing inflow at least up to the point at which flushing removes nutrients and products faster than rates of biological production. For Galveston Bay itself, community production data of the previous section seemed to cover the relationship just on the threshold of flushing impacts, whereas our main interest is in the behavior of the system at lower inflows. Therefore, production rates calculated for the bay are related to the more general relationship for which we have a little information in prior **Figure 21**. That data was re-expressed as the proportion of each measurement to the maximal observed. In the model, the equation fit through those points is used as the inflow relationship, modifying maximum expected production (as g $C \cdot 10^6$/bay/y). Consumption serves to keep this reservoir from continuing to accumulate beyond the yearly cycle, as required to maintain the annual inputs to the nitrogen section of the model.

Other Aspects

In the diagram, *annual* is where the figure can be found which corresponds to the remainder of the annual nitrogen budget. **Table 12.** *Annual* serves a comparative purpose, and is derived by separating past accumulations in *bay content* from present contributions. This allows the model to show each year's contribution to the cumulative stock.

Past bay content, implemented by a 1-time step delay function, is the bay reservoir accumulation prior to present calculations. *Past-2* and *past-3* are similarly determined previous years accumulations, to introduce lagged effects.

The *reduction factor* converter is a locus for testing effects of alterations in inflows to the model, a constant by which input historic inflows are multiplied.

Tidal volume (for both input and losses entrainment) are constant with respect to inflows and symmetric, ebb and flood. *In-mix* and *out-mix* are the portions of tidal volumes exchanged which contribute actual mass transfer.

The *partition* coefficient determines the portion of *bay content* which is dissolved or suspended matter, vulnerable to transport loss. This mass divided by bay volume becomes the outflow TN concentration, *conc low bay*.

Fisheries harvest and escapement is calculated from primary production, applying a 15% efficiency (Ryther, 1969) of converting algal biomass to consumer biomass up each of two consumer levels. *Fish-factor* is the portion of tertiary production which annually leaves the bay. This factor was determined iteratively to obtain values in the range of those used in the nitrogen balances. (Over the range of inflows in the data set, the result is an increasing harvest with increasing inflow, **Figure 23.**)

Denitrification *(denit)* is calculated as a base rate plus an amount determined by multiplying the average of the previous three years' *baycontent* by a constant, determined through sensitivity testing. The constant was chosen to bring calculated denitrification within the range of values used in **Table 15.** The form of this relationship is at this point speculative, but reasonable.

Burial is based on a linear relationship between inflow and results of burial calculations presented in the nitrogen budget table. The product of equation is reduced slightly by a ratio of adjusted to calculated burial from the long-term budget. Implicit here is an assumption that the amount of burial is

Figure 23 Relationship of nitrogen loss via fisheries to freshwater inflow volume in simulation results.
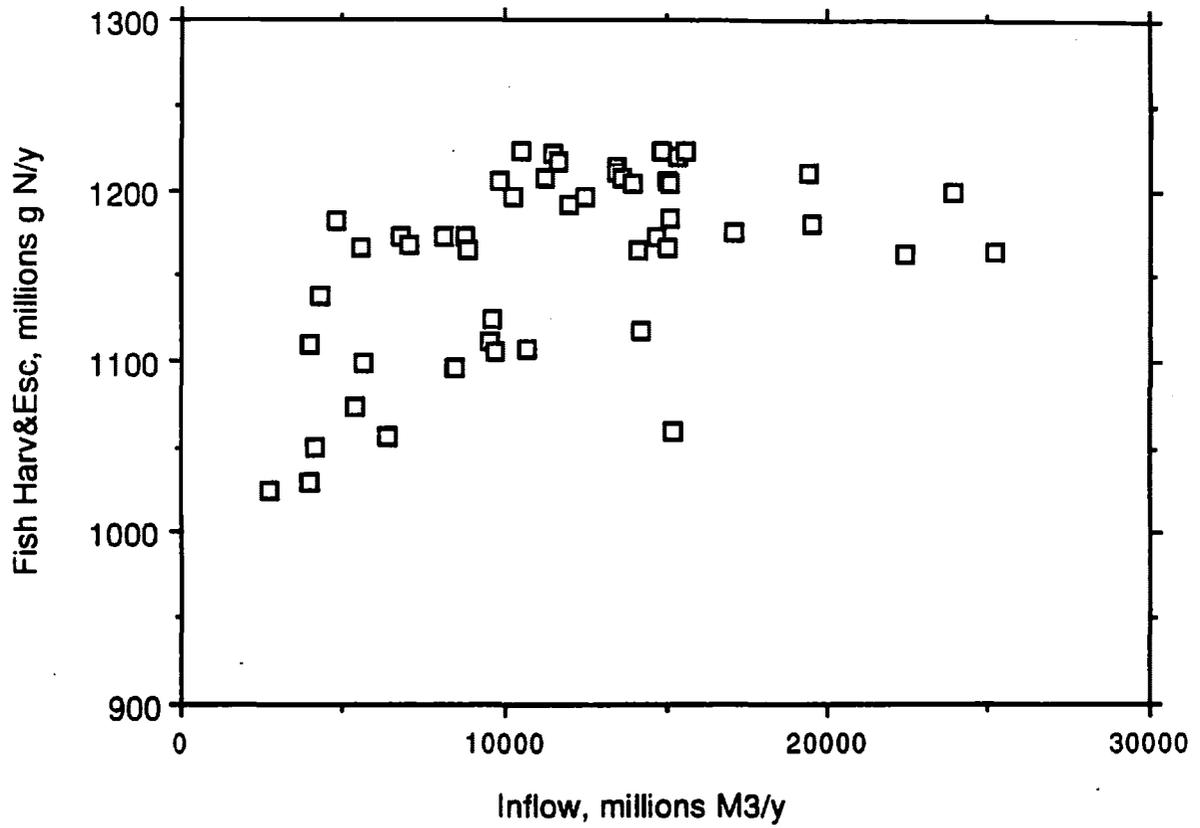
determined by inflow control of sedimentation more than by sediment nitrogen storage. The nitrogen at a depth removable by burial probably reflects an accumulation over a longer span of time than nitrogen subject to denitrification.

Flow data input to the model are 1941-1990 gaged+modeled-diverted (GMD) total annual volumes. A linear relationship estimates total water-balance inflows required by some parameters, like *net* outflow.

Tidal volumes are averaged from 1988-1990 simulations of circulation, from **Table 10**, and assumed symmetric and constant. Instead of a function relating entrainment outflow to inflow, entrainment outflow is based on tidal volume and a concentration which varies with bay content. Paragraphs below discuss the use of feedback from *bay content* in the model.

Results and Discussion

Model Behavior

Figure 24 demonstrates the model output of nitrogen input/output balance over the historical period of inflows for simulations using a lower bay concentration partition of 0.01, stream flow concentration set at 2.49, and fisheries loss at 70% of tertiary production. Constants in this model were adjusted so that the components they controlled were consistent with the prototype long-term budget, **Table 15**. The *annual* balances in **Figure 24** are equivalent to the remainders shown in **Table 12**. *Bay content* in the figure, the cumulative balance, demonstrates system inertia in response to inflow variation, dampening some fluctuations in annual balances, accumulating or declining with trends. Years with large positive balances maintain *bay content* through the following low-balance years. This is consistent with our expectations of real-world behavior. Note that the first few years of simulation may contain some artifacts from initialization and use of lagged terms. **Figure 25** shows simulation results as a scatter plot of annual balances against inflows. The fit line crosses the zero-balance line near the median inflow, $12450 \cdot 10^6$ M^3/y, which indicates model consistency with the long-term budget.

Figure 24. Annual nitrogen balance and bay nitrogen content simulated over 50 years, using 1941-1990 inflows.
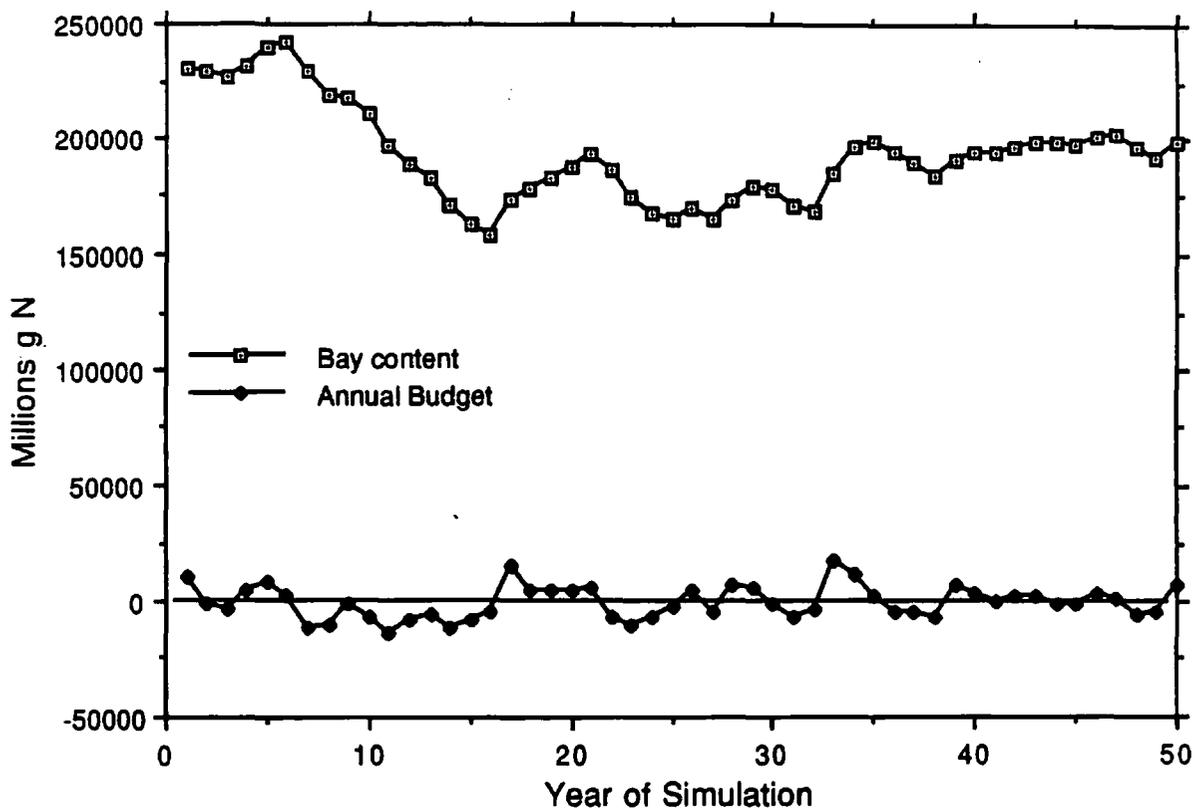
Figure 25. Annual Nitrogen balance from simulations, showing linear fit line and annual inflow at balance = 0.
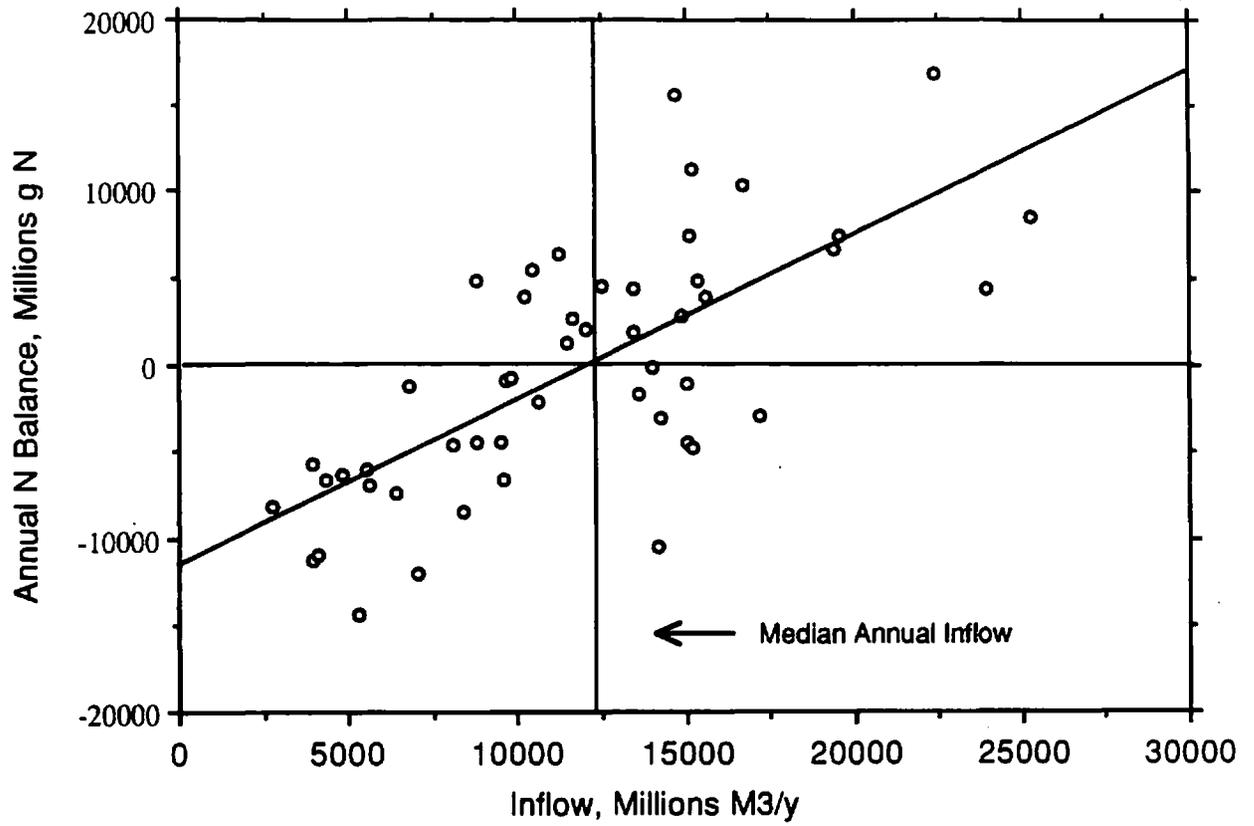
Responsiveness

Model behavior is decidedly influenced by the degree of feedback incorporated in some loss relationships, particularly in determination of lower bay concentration and in denitrification. We expect both rate and parameter would be influenced by ecosystem condition or nutrient richness, although the exact relationship might involve several variables. That is, we expect that an estuary with lower nitrogen stock held in sediments would show lower denitrification rates than would be found in an estuary with higher stocks. We also expect an estuary with high suspended and dissolved nitrogen would have higher transport nitrogen loss than an estuary with less in its water column. Unfortunately, existing denitrification data are not adequate to empirically determine relationships involved. **Figure 5** suggests that from year to year, inflow is an influence on lower bay concentrations, as annual averages. Whether the concentrations vary with bay nitrogen storage, as in the model, would require more study. Sensitivity analyses, such as described below, were used to adjust parameters involved in linkage between bay content and loss rates.

Table 30 contrasts relationships included in responsive and non-responsive versions of the model. Two important parameters are given constant value in the non-responsive model, denitrification and lower-bay concentration. Lower bay concentration helps determine both net and tidal outflow. **Figure 26** shows that annual balances of responsive and nonresponsive models differ in average magnitude and slightly in amplitude. **Figure 27** shows a clear qualitative difference between the models. In the non-responsive model, bay nitrogen storage cannot recover from a series of negative annual balances. The model exhibits unstable behavior. In the real estuary, processes and interactions which result in system feed-back responsiveness are likely much more complex than portrayed in the responsive model. Even so, results demonstrate the model shows realistic behavior when incorporating some capacity for adjustment. There is a clear need for more information on relationships governing denitrification rate and outflow total nitrogen concentrations.

Table 30. Comparison of terms differing between responsive and non-responsive models

Responsive Model

conc lowbay   = if(baycontent >0) then (baycontent/2409)·partition
                        else 0.0
denit   =if years > 3 and (past+past2+past3 > 0) then
                        ((past+past2+past3)/3)·denit-factor   +   370,
                        else 370
denit-factor   =.0162

Non-Responsive Model

conc lowbay   =.91

denit   =3200

Figure 26. Sensitivity of annual balance to incorporating feedback from bay content in calculation of denitrification and lower-bay TN concentration.
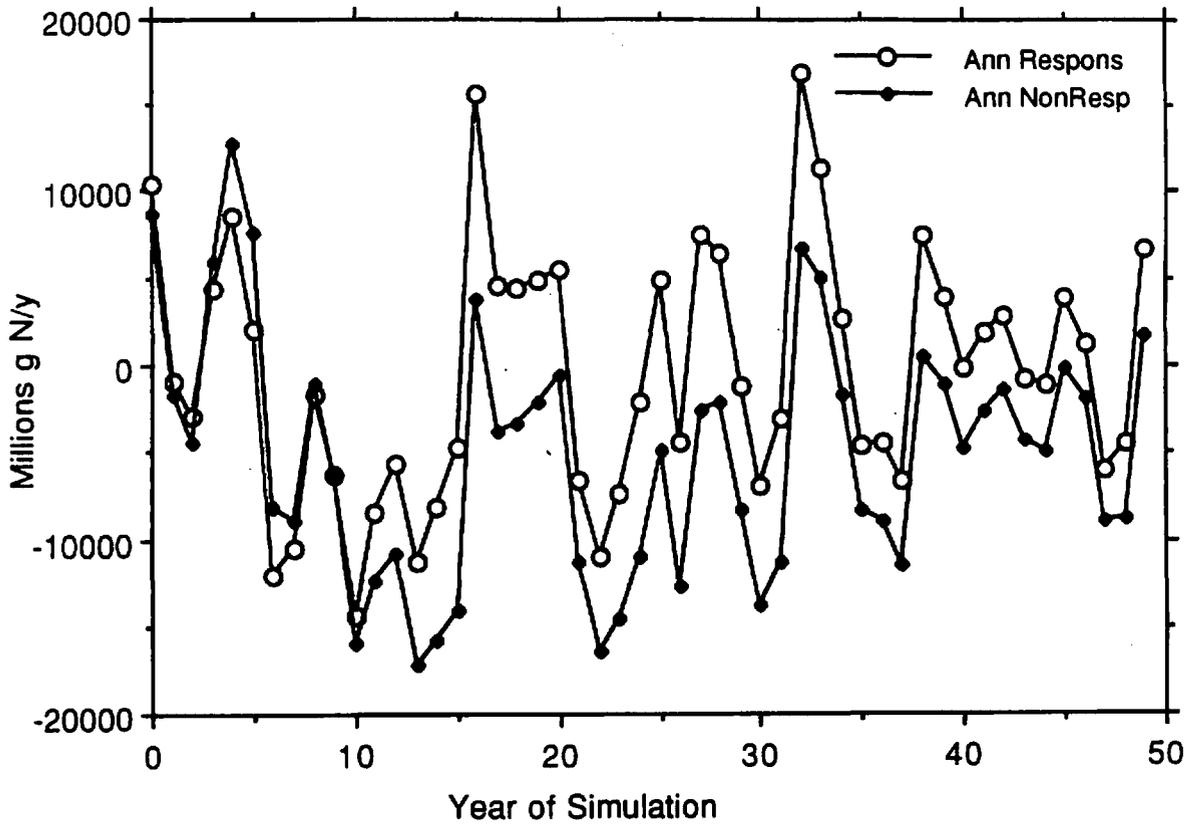
Figure 27. Sensitivity of bay content to incorporating feedback from bay content in calculation of denitrification and lower-bay TN concentration.
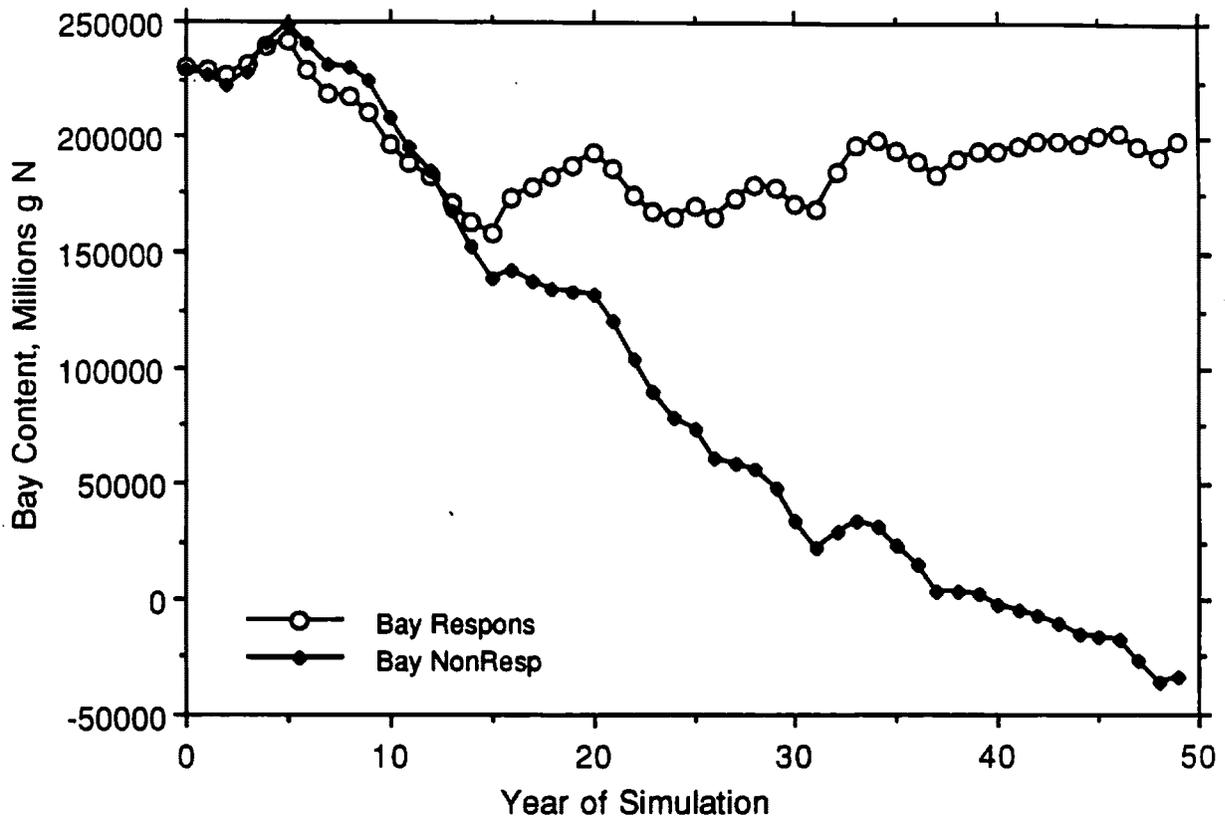
Model Sensitivity

Reservoirs and terminal *converters* are the loci of sensitivity testing. These are components which do not in turn depend on other components. In the model, we display some parameters explicitly as converters so that sensitivity of results to those values can be tested. Candidates for sensitivity testing include, on the input side, wastewater input (mass), rain + fixation (mass), Gulf nitrogen concentration, mixing rate, and stream concentration. On the loss side, sensitivity testing could be run on partition, mixing rate, proportion of storage loss to denitrification, and fisheries loss. Sensitivity of model results to changes in inflow is accomplished through changing the reduction factor.

Figure 28 presents simulation sensitivity results on a constant input term, *wastewater*. As expected, the axis of annual variation moves up and down according to the magnitude of the constant input. The contribution of wastewater to total variation is not great in comparison to other effects, below.

Sensitivity to *partition*, determining outflow concentrations, is shown in **Figure 29**. As *partition* is changed from .0075 to .025, average lower bay concentrations change from 0.75 to 0.80. With a low partition coefficient, less of the system nitrogen is available for transport loss. Higher partition coefficients increase the amount of nitrogen which can be lost through transport. Operationally, this parameter is adjusted using the results of sensitivity testing, so that resulting lower bay concentrations match observed average values.

There is a real system relevance of the partition coefficient parameter. It stands to some extent for the degree to which system nitrogen is subject to transport loss. A system with high planktonic nitrogen--high populations of phyto- and zooplankton and particulates--would have a higher partition coefficient than a system with a larger proportion of nitrogen tied up in seagrass and other attached or sedentary fauna and flora. The model shows that a strongly plankton-based system would have lower standing stock than a system less so, as would be expected. For shallow estuaries, such as those of Texas, in which much of the system nitrogen pool is in surface sediments and the volume/area ratio is low, the range of this parameter is probably limited.

Figure 28. Baycontent sensitivity to wastewater input variation.
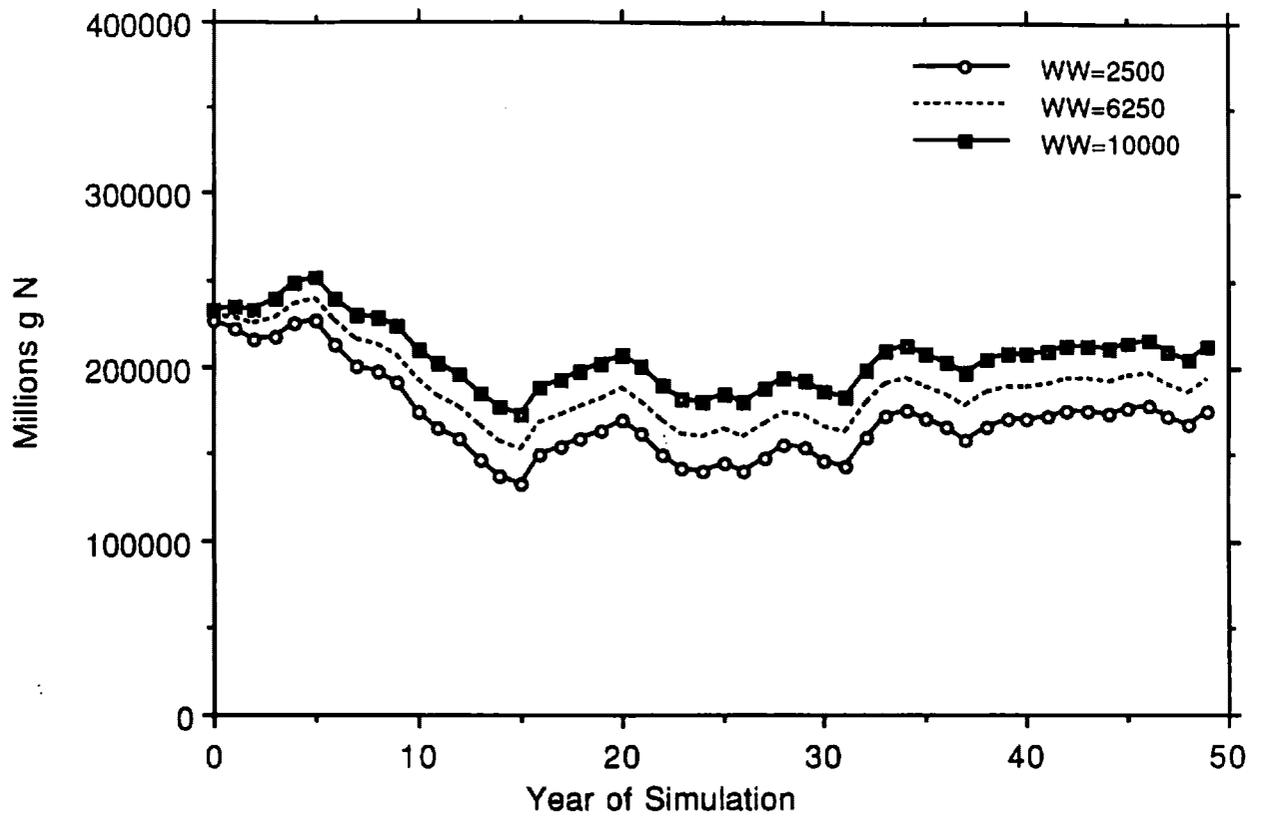
Figure 29.   Sensitivity of bay content to the partition coefficient.
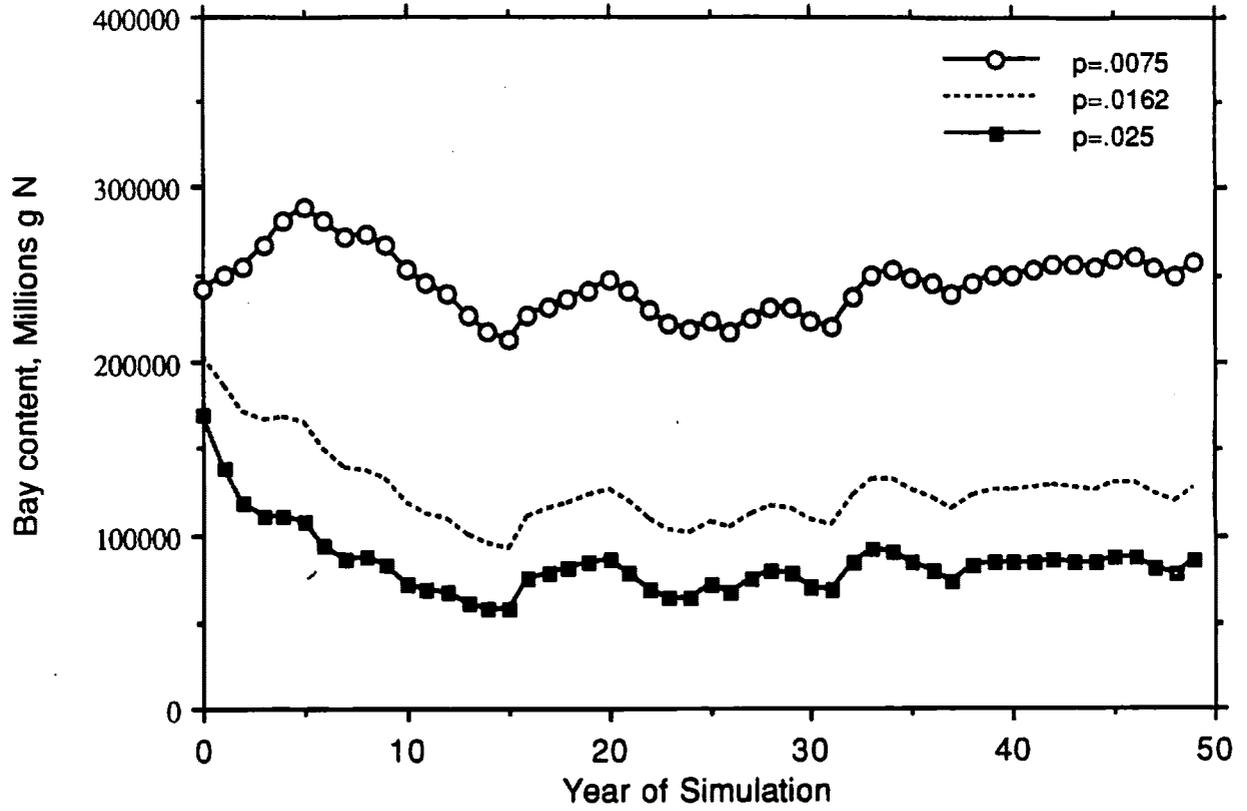
Sensitivity to freshwater inflow

A test of sensitivity of the nitrogen system to freshwater inflow can be accomplished through sensitivity testing on the *reduction* factor. **Figure 30** and **Table 31** display results. In the context of sensitivity of bay content to small magnitude variation in other parameters, bay content does not seem dramatically influenced by large changes in freshwater inflow. Of course, the magnitude of changes to specific species populations could be severe.

Bay-Gulf mixing is important in establishing the mean system storage. **Figure 31** demonstrates sensitivity of *bay content* to tidal mixing. In the figure, outflow mixing is maintained larger than inflow mixing, on assumptions explained earlier. Small values of mixing tend to maintain storage within the system, whereas large values increase outflows more than inputs, lowering system storage. Though the geometry of the bay opening and tidal energy level may chiefly determine mixing rates, it is possible that freshwater inflow rate may also influence mixing, through turbulence or shear. In the real world, qualitative changes in the system which could accompany changes in salinity as mixing rates change, such as increases or declines in aquatic vegetation or oyster beds, could alter the actual outcome somewhat.

Tidal mixing is undoubtedly a function of the dimensions of openings between the bay and Gulf, and may be influenced by internal circulation as well. The construction of the Houston Ship Channel and Texas City Dike may have had significant impact on bay-Gulf exchange. If these changes affected an increase in exchange rates, they may have significantly reduced the bay's capacity to retain nitrogen. If true, this could weaken the applicability of pre-modern nitrogen loading to sustain characteristic production in the bay. The Houston Ship Channel is now scheduled to be widened and deepened. Modeling studies have been undertaken and have not indicated major alteration in Galveston Bay salinities (which would indicate changes in exchange rates) (Berger et al., 1995). Our calculations show that the channel project will result in a cross-section area at the end of the Galveston Bay entrance jetties 103% that at present. Model sensitivity tests using a 105% increase in mixing show *bay content* could decrease to 97% of present values with the project. Given uncertainties concerning partition coefficient, mixing coefficient, and other assumptions which reduce the reliability of the exact results, the model does

Figure 30.   Bay content sensitity to reduction in freshwater inflows.
Inflows as percentage of normal.
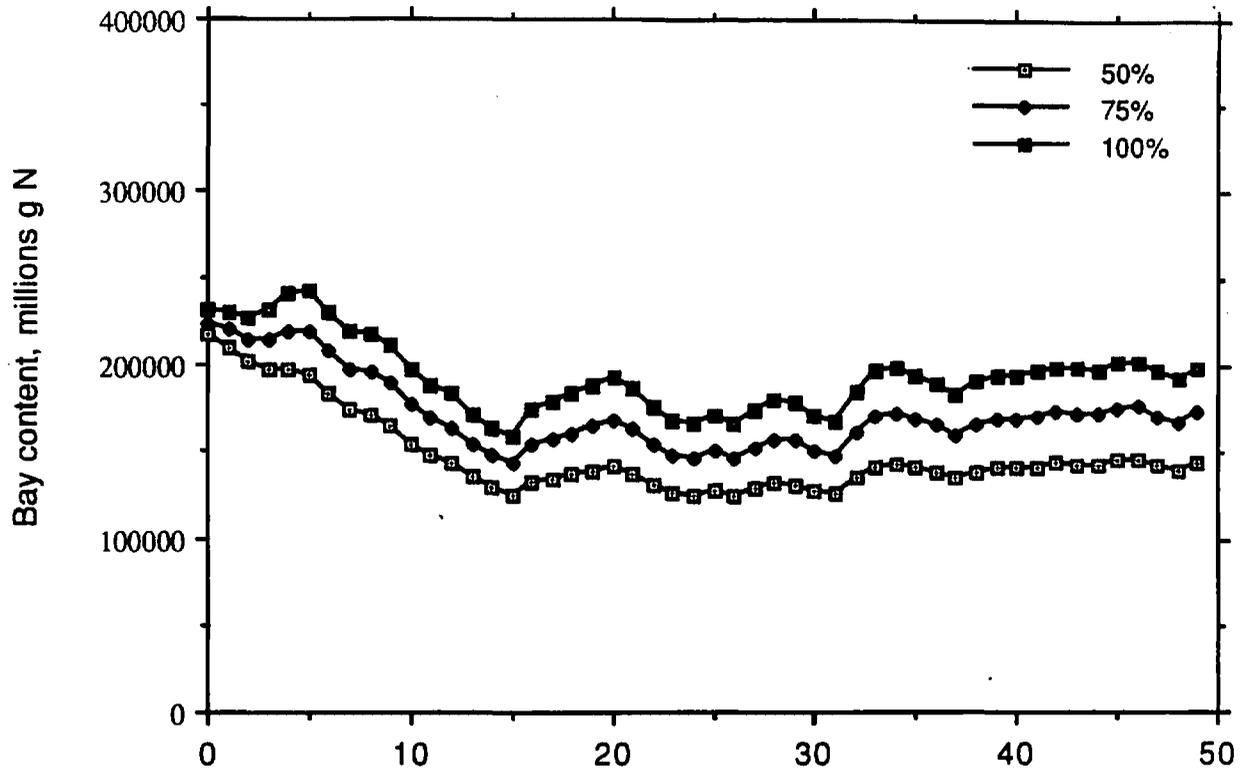
Table 31. Sensitivity of baycontent to inflows. Units are $10^6 \cdot g$ N/y.

Inflow	*Baycontent*	
% of Historic	Mean	s.d.
50	147400	24010
63	159840	22800
75	171560	21950
88	182620	21400
100	·193070	21110

Figure 31. Sensitivity of bay content to tidal mixing rates.
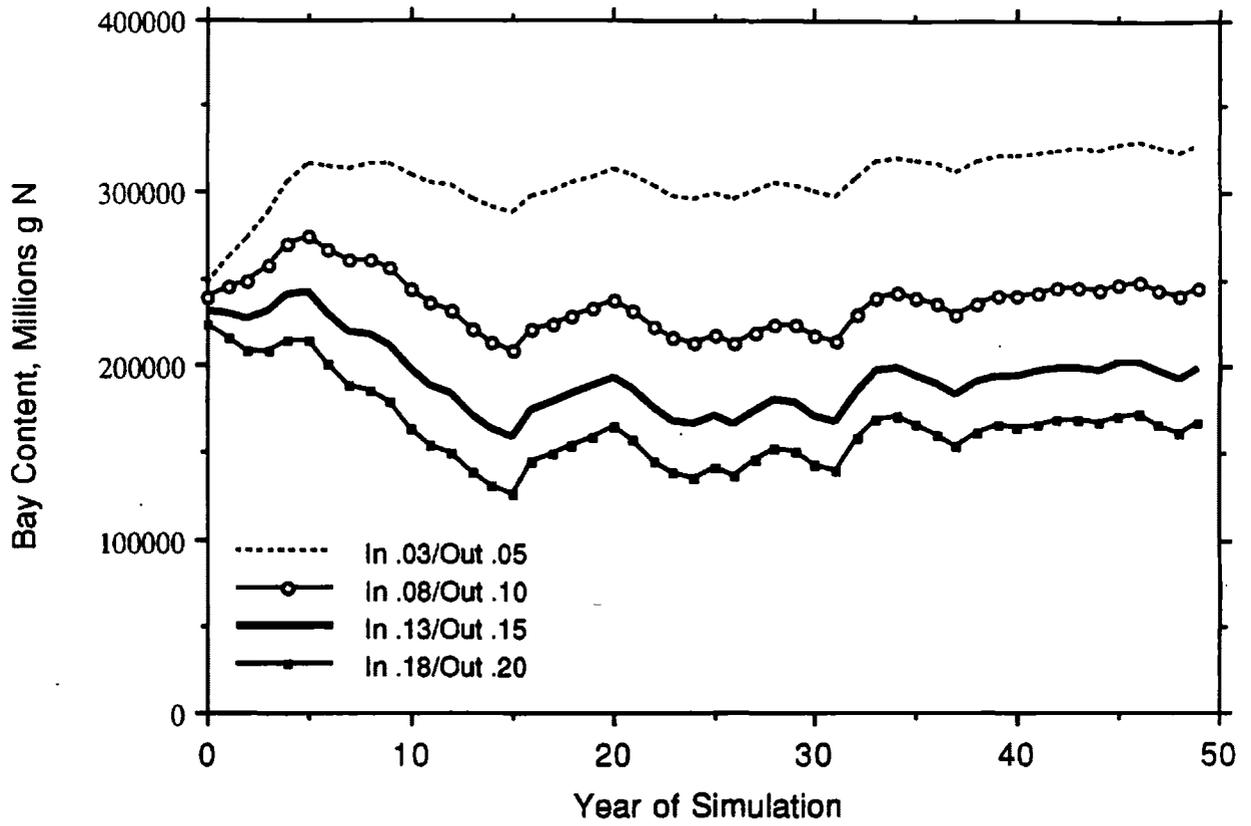
not suggest a nitrogen-level system response to the changes in channel dimensions.

Further Work

Strong dependence of the model behavior on relationships between compartment storage and parameters of loss, show areas where further data gathering or enlightened analysis is required for support. Linear formulations in particular are not likely to be realistic.

Although we report new information on system production, additional work would help bring production and fisheries into the model in a more satisfactory way. An obvious place to start concerns the interaction between available nutrients and turbidity in controlling production, and the influence of freshwater inflows on both factors. The way the model is set up, a linkage between system ecological health and biologically controlled process rates such as denitrification and fraction of harvest-able fish may be accomplished through feedback from production.

Comparative work across bays may be required to support application of fisheries or productivity data to long-term system behavior. This is because most data on record applies only to short-term variation, not to the shift of the axis of variation up or down as we observed in simulation results. There just haven't been long enough periods of consistently higher or lower inflows to allow an individual estuary to adjust biologically.

Modeling offers the prospect of projecting system behavior to conditions not occurring in the estuary. One area which we had hoped to explore is a situation of excessive nutrient supply to the estuary, leading to symptoms of eutrophication. **Figure 32** shows a module inserted into the main model structure which would link development of nuisance algal blooms to conditions of nutrient availability and system flushing. This builds from the index "dissolved concentration potential" developed by NOAA(1989) to determine susceptibility of estuaries to pollution. There are some apparent opportunities for enhancement of the basic model which can probably best be dealt with by a more detailed, physio-chemical process model. The model presented here has potential for testing index-type formulations as predictors of estuarine performance. The above applications have shown that the simple model can reveal important implications of the linkages involved in nutrient balances.

Figure 32. Model diagram showing module for indication of conditions
promoting algal bloom formation.

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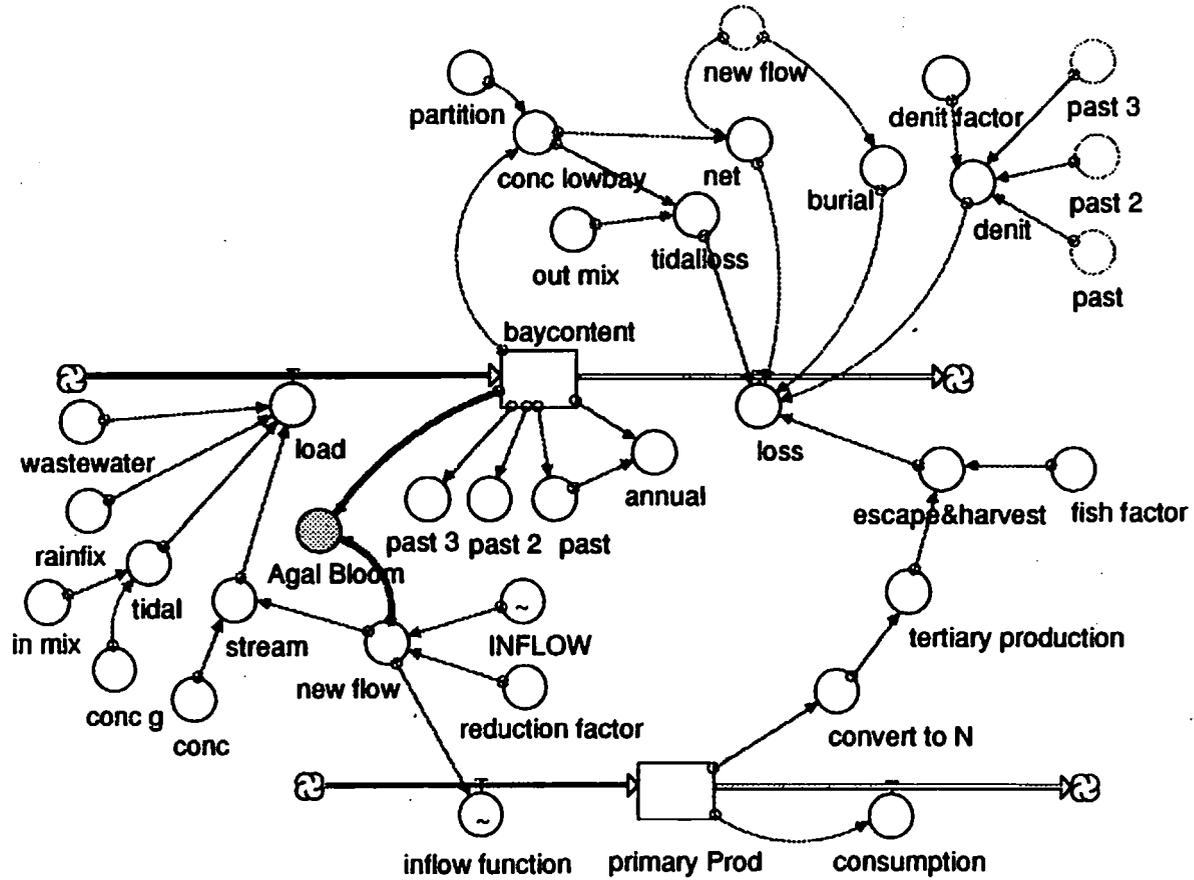
fig 32

Section IV. Galveston Bay Nutrient Requirements, Conclusions

Nutrient Loading Requirement for Biological Production in Galveston Bay

There are several general-level questions which need to be addressed to establish a supportable nutrient requirement for estuarine productivity. First, whether or not the quantity and quality of present level of production is desirable. By quality of production, we recognize that a high productivity of noxious algal blooms may be as much a negative concern as a high productivity of shrimp is a positive concern. Second, would another form of production be as acceptable. This could include considerations of the production of non-harvested features of the system, such as submerged vegetation, as well as possible shifts in proportions of fisheries species. Third, does present production show evidence of strong dependence on nutrient loading, or do other factors exert stronger control once nutrient loading exceeds some threshold level. This report does not build from clear knowledge of the answers to these questions for Galveston Bay. Some decisions need to be made in a broader context. The analyses do touch on each aspect, however, and the findings should indicate a real need for careful consideration of those questions.

The nutrient status of Galveston Bay with respect to nitrogen need is not entirely clear. Comparison of fisheries production versus loading among Texas Bays seems to demonstrate a definite linkage, with Galveston high in both categories. However, analysis of community primary production and respiration indicates that much of the production may be based on metabolism of organic material derived from the terrestrial drainage, not based on primary production within the system. This would lessen the amount that secondary production would be reduced if lower nutrient loadings reduced bay primary production. In addition, relatively weak regressions of production on freshwater inflow suggest little diminishment of bay production with reduced loading, at least over the range of inflows tested. The caveat here is that modeling illustrates the ability of the system to buffer over-all production against year-to-year loading variation. That buffer capacity is likely related to average loading levels. Therefore, over an extended period of years, lower loading could reduce that buffering capacity and might not keep production up to present levels.

The approach to the establishment of nutrient requirements here is through analysis of system inputs and losses of nutrients. Input requirements must balance losses. Some sinks are constant, but other losses are related to amount of input or amount of some other factor, such as freshwater inflow. Nutrient budgets prepared for years of high and low freshwater inflow and nutrient loading provided a basis for describing system response to a range of nutrient loadings and losses. Loss of nitrogen to the Gulf of Mexico was the most important sink, denitrification was second in importance, and losses to burial and fisheries harvests and escapement were variously third and forth.

From nitrogen budgets compiled for years of high- and years of low nitrogen inputs, we constructed a long-term nitrogen budget. This budget balances at a median nitrogen loading. Therefore, nitrogen loading to maintain system processes at present rates would seem to be that median loading rate, $37,700 \cdot 10^6$ g N/y, from the drainage basin. However, this static evaluation does not indicate whether a new balance is possible with lower loading and adjustments by the biological and geochemical systems. Given the range of loading levels among Texas bays which support healthy estuarine communities, changes to Galveston Bay production with moderate decreases in loading may not be deleterious. Modeling results showed only moderate sensitivity of system nitrogen retention to large reductions in loading. The model is dependent on feedback relationships in portions of the nitrogen cycle which have not yet been quantified for Galveston Bay but which seem reasonable.

An alternative nitrogen requirement for Galveston Bay can be stated from our understanding of historical loading levels. An annual input of $14,900 \cdot 10^6$ g N/y will provide Galveston Bay sufficient nitrogen for biological production characteristic of the system prior to major anthropogenic increased inputs. This nitrogen load is proposed as a minimum target input for present management. Phosphorus inputs of $2,100 \cdot 10^6$ g P/yr are proposed as a minimal input target consistent with the nitrogen recommendation. Nitrogen budget analysis showed that if long-term nitrogen inputs were to be reduced to average around the proposed minimum, both the nutrient richness of lower-bay waters and loss of nitrogen in bay sediments would need to reduce substantially, but not unrealistically, to maintain a long-term nitrogen balance. Thus, the proposed minimal nitrogen loading assumes some

qualitative changes in the system in the direction of increased system nitrogen use efficiency.

Limitations in data and knowledge of linkages required for modeling prevented an exploration of upper limits for nutrient inputs for ecosystem health. Considering either proposed minimum loading requirements or upper limits in actuality require decisions on what constitutes a desired ecosystem condition. Altering average nutrient input levels are likely to produce shifts in quality of production and shifts in proportions of commercial species consistent with changes in bay water quality.

Related Findings

Rates of community primary production were derived from analysis of diurnal oxygen concentration curves. Rates of demand for nutrients to support this productivity were derived from production and compared to calculated rates of nutrient supply. This analysis demonstrated the richness of Galveston Bay with respect to nutrients, the supply exceeding the demand commonly in all parts of the bay and all seasons. Based on calculations from community respiration rates, the rate of nutrient supply regenerated metabolically by bay consumers was usually sufficient to meet the short-term needs of production. High respiration rates also indicate that a great deal of Galveston Bay productivity is fueled by the processing of organic material brought into the system from terrestrial sources. Therefore, adequate consideration needs to be given to changes to inflows or other system parameters which would affect organic carbon input in addition to concerns with nitrogen, phosphorus, etc. The respiration and production analysis suggests better information on benthic nutrient processes, on feeding structure, and on trophic conversion efficiencies should permit us to develop an alternative nutrient requirement based on system production itself.

We tested variation of primary production rates for a strong relationship with nutrient loading from freshwater inflow as a means to directly establish nutrient requirements. However, the low strength of inflow relationships which developed may instead support the hypothesis that primary production in the bay is limited more by the prevailing turbidity of the water than by nutrient requirements. Production and respiration regressions on inflow were most successful when they incorporated influence

of inflows during several or many months preceding rate measurements. This suggests the importance of an inflow regime setting up biological communities which perform in concert, as opposed to a quick response to nutrient inputs. The negative slope of inflow-production relationships observed for some sites suggests that, for Galveston Bay, the negative flushing effects of high inflows are stronger than the concomitant positive effects of higher inflow's nutrient contribution. This is consistent with the hypothesis that phytoplankton are limited by light availability or other factors from being able to make the most use of nutrient input.

Reducing nitrogen input to the system by half, according to the Galveston STELLA model, will reduce the system total nitrogen content only by one-fourth. This is largely the result of predicted adaptations in system nitrogen processing which reduce losses. Evidence of the system responsiveness required to buffer the impact of reduced nitrogen inputs should not be difficult to discover and is necessary to support model conclusions. The STELLA nitrogen model also shows the system nitrogen economy is most sensitive to changes in tidal mixing. This is consistent with results of nitrogen budget calculations, which show tidal mixing as a major vehicle for nitrogen loss. Stated another way, accurate knowledge of tidal mixing would be most important to refinement of the budget, eliminating a large potential source of error.

It was necessary to incorporate some data from another estuary in the STELLA model to adequately portray the full range of production rates response to inflow variation. Incorporating comparative work from other estuaries on other processes in the model should reinforce the model and conclusions which can be drawn from it. The STELLA model would be enhanced with strengthened relationships. As it is, this model may be used to apply the major results of this study to assess impacts of resource management options.

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