1-1-1996

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ESTIMATED NITROGEN FLUXES AND NITROGEN-CHLOROPHYLL RELATIONSHIPS IN TAMPA BAY, 1985–1994

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ABSTRACT

Cultural eutrophication driven by anthropogenic nutrient loadings has been identified as a priority problem affecting the water quality of Tampa Bay. Bay management programs therefore wish to control future nutrient loads, an effort that will be complicated by the continuing population growth projected for the watershed in coming decades. The eutrophication management effort is "resource-based," seeking to maintain water clarity in the bay that is adequate to support seagrasses at target acreage levels. Among the factors affecting water clarity, anthropogenic nitrogen loadings and corresponding increases in phytoplankton density appear most amenable to management.

A two-pronged (mechanistic and empirical) water quality modeling approach has been used to develop nitrogen budgets for the bay to support the eutrophication management effort. The two models are driven by the same estimated hydrologic and nitrogen loadings to provide maximum comparability.

Although external loadings are the ultimate source of the nitrogen present in Tampa Bay, the mechanistic water quality model suggests that under current conditions annual nitrogen fluxes associated with estuarine regeneration, recycling and transport processes are larger than annual external loadings. Phytoplankton bioassay data, analyses of measured primary production rates and nitrogen standing stocks, and water quality trends observed in the bay following recent reductions in point source loadings, appear to corroborate the model-based flux estimates. Taken together, the available information suggests that watershed management practices which cause small year-to-year reductions in external nutrient loadings (relative to the size of the existing internal fluxes) may not have immediately detectable effects on bay water quality. If sustained over a number of years, however, management efforts which continue to produce small annual reductions in external loadings should provide water quality improvements by reducing the sizes of the internal nitrogen pools available to support phytoplankton growth.

BACKGROUND

Cultural eutrophication—an unnatural increase in the water body's trophic state driven by anthropogenic nutrient loads—has been identified as a priority problem affecting the water quality of Tampa Bay (Federal Water Pollution Control Administration 1969, Johansson 1991, TBNEP 1996). The Tampa Bay National Estuary Program and other bay management initiatives are therefore attempting to develop strategies for controlling future nutrient loadings, in order to protect water quality despite the increasing population growth, urbanization and other forms of development that are projected to occur in the watershed in coming decades (TBNEP 1996). The eutrophication management effort is "resource-based," seeking to achieve and maintain adequate water clarity to allow seagrass meadows to expand to the acreage levels observed in the bay during the early 1950s (Janicki and Wade 1995; Greening, Morrison et al., this volume).

Reductions in water clarity, as a result of elevated concentrations of phytoplankton and other turbidity sources, have contributed to losses of estuarine seagrass meadows on a worldwide basis (e.g., Giesen et al. 1990, Batiuck et al. 1992, Dennison et al. 1993, Morris and Tomasko 1993, Dixon and Leverone 1995). Over the long term, plants that do not receive adequate sunlight to allow the rate of photosynthetic production to exceed respiration will cease growing and eventually die, leading to reduced seagrass acreage in estuaries where anthropogenic nutrient loadings cause significant reductions in water clarity. In addition to reduced light availability due to increased phytoplankton growth, other water quality-related factors (e.g., shading by epiphytes and drift macroalgae, direct nitrogen toxicity) have been suggested as possible contributing factors in some estuarine seagrass declines.

Recent research indicates that *Thalassia testudinum*, the seagrass species for which water quality goals and nitrogen loading targets are being set in Tampa Bay, requires an annual average of 20–25% of subsurface irradiance for long-term survival in the bay (Dixon and Leverone 1995). In order to restore and maintain seagrass meadows at target depths in different bay segments, these results indicate that water quality in each segment should be sufficient to allow 20–25% of the sunlight which penetrates the water surface to reach the target depth. Light penetration through the water column of Tampa Bay is primarily affected by three factors: phytoplankton density, non-phytoplankton turbidity, and water color (Janicki and Wade 1995). Among these factors, phytoplankton cell density (as measured by chlorophyll-*a* concentration) appears most amenable to management. Water color, which is caused by the presence of dissolved organic compounds in the rivers and streams that flow into the bay, is a natural source of light attenuation to which bay organisms are presumably adapted. Non-phytoplankton turbidity, which is caused by a number of factors including natural estuarine physicochemical processes and natural and anthropogenically induced resuspension of bay sediments, does not appear susceptible to management using current technologies. Chlorophyll-*a* concentrations, however, can be managed by altering the loadings of nutrients (particularly nitrogen) that are discharged to the bay as a result of human activities in the watershed (Johansson 1991, Janicki and Wade 1995, TBNEP 1996).

A 1991 water quality modeling workshop sponsored by the TBNEP recommended implementation of a multi-pronged (empirical and mechanistic) modeling approach to provide technical support for the water quality management process (TBNEP 1996). In accordance with that recommendation, the TBNEP has funded the development of an empirical, regression-based water quality model (Janicki and Wade 1995, 1996) and the Southwest Florida Water Management District's Surface Water Improvement and Management (SWIM) program has funded the application of a mechanistic model (Martin et al. 1996) to Tampa Bay. To ensure comparability, the two models simulate the same ten-year time period (1985–1994) and are driven by the same estimated hydrologic and nutrient loadings (Zarbock et al. 1994, Janicki and Wade 1995, Zarbock et al. 1996). The intent of this approach is that two complementary water quality models, which are based on the same hydrologic and nutrient loadings but very different technical assumptions and mathematical modeling approaches, will be available to serve as cross-checks for one another during the development of water quality and nutrient loading targets.

As summarized by Greening, Morrison et al. (this volume), the following steps are being used by the TBNEP and other resource management programs to develop seagrass-based targets for light attenuation, chlorophyll-*a* concentrations, and nitrogen loadings to Tampa Bay:

1) Identify acreage and depth targets for seagrasses in each bay segment;
2) Identify the light requirements of target seagrass species (e.g., 20–25% of subsurface irradiance for *Thalassia testudinum*);
3) Estimate levels of water column transparency (e.g., light attenuation coefficient, Secchi disk depth) necessary to ensure that adequate sunlight reaches target depths;
4) Estimate chlorophyll-*a* concentration targets that are consistent with water column transparency targets; and
5) Estimate nitrogen loading targets that are consistent with chlorophyll-*a* targets.

This report describes the empirical and mechanistic models used in steps (4) and (5) of the target-setting process, and the nitrogen loading estimates used by the models to simulate the spatial and temporal dynamics of bay water quality.

NITROGEN LOADING ESTIMATES

Estimated monthly nitrogen loadings were developed for the period 1985–1994 by Zarbock et al. (1994, 1996) and Janicki and Wade (1995, 1996), based on the following categories of potential sources:

- general nonpoint sources (stormwater runoff and base flow),
- point sources (domestic and industrial facilities, springs),
- atmospheric deposition falling directly on the bay surface,
- groundwater inputs (with the exception of spring discharges),
- losses of fertilizer products during handling at shiploading facilities, and
- reported spills resulting from other (primarily industrial) human activities.

For comparative purposes, loading estimates were also developed for a "benchmark" less-impacted period (c. 1940) in the recent past, for a potentially more-impacted period (c. 2010) in the near future, and for a "worst-case" period of severe anthropogenic impacts which occurred in the late 1970s prior to the implementation of more stringent treatment levels at a number of municipal wastewater treatment plants and industrial (e.g., fertilizer processing) facilities (Johansson 1991; Zarbock et al. 1994, 1996; Janicki and Wade 1995, 1996).

Annual loading estimates for total nitrogen (TN) during each of these periods are summarized in Figure 1 for four major Tampa Bay segments (Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay and Lower Tampa Bay). The estimated annual loadings show a very large (>400%) increase between the benchmark and worst case periods, a substantial (62%) decline between the worst case and present-day periods, and a projected 7% increase between the present-day and 2010 periods. The latter increase in predicted load is driven primarily by the anticipated growth rate of the human population within the Tampa Bay watershed and airshed (Zarbock et al. 1996).

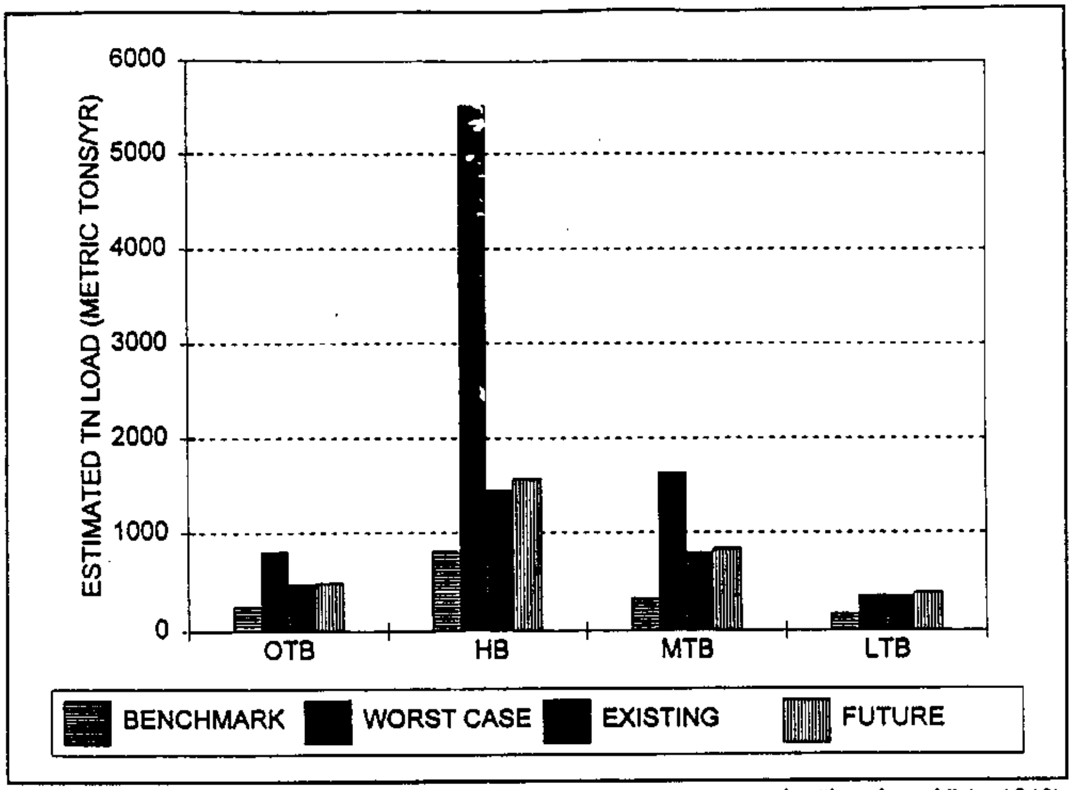

Figure 1. Estimated total nitrogen (TN) loadings to Tampa Bay segments under "benchmark" (c. 1940), "worst case" (c. 1979), "present day" (1985–1994) and projected future (2010) conditions. Source: Zarbock et al. 1994, 1996.

CHLOROPHYLL-*a* AND WATER CLARITY TRENDS

The Environmental Protection Commission (EPC) of Hillsborough County has collected monthly data describing chlorophyll-*a* concentrations and water clarity (Secchi disk depth) levels at 52 sites in Tampa Bay since 1974 (Boler 1995). Annual average values of these parameters are shown for the four largest bay segments in Figure 2.

Throughout the period 1974–1995, chlorophyll-*a* concentrations have been highest in Hillsborough Bay (HB), intermediate in Old Tampa Bay (OTB) and Middle Tampa Bay (MTB), and lowest in Lower Tampa Bay (LTB)(Fig. 2). Particularly high concentrations were observed in HB during the period 1974–1982, and marked reductions occurred in the three upper bay segments (HB, OTB, and MTB) between the early 1980s and early 1990s. These reductions in chlorophyll-*a* concentration, which are thought to have been due primarily to reduced nitrogen loadings from domestic and industrial point source discharges (Johansson 1991, Boler 1995), appear to have begun one year earlier in HB (1983) than in OTB and MTB (1984). Apparent concentration trends in LTB, a bay segment whose water quality is influenced to a particularly large degree by tidal exchange with the Gulf of Mexico, are more difficult to discern, although there is some indication of a weak increasing chlorophyll-*a* trend during the period 1974–1983, a reduction during 1984, and fluctuations around an average value of approximately 5 $\mu g/L$ during the period 1984–1995.

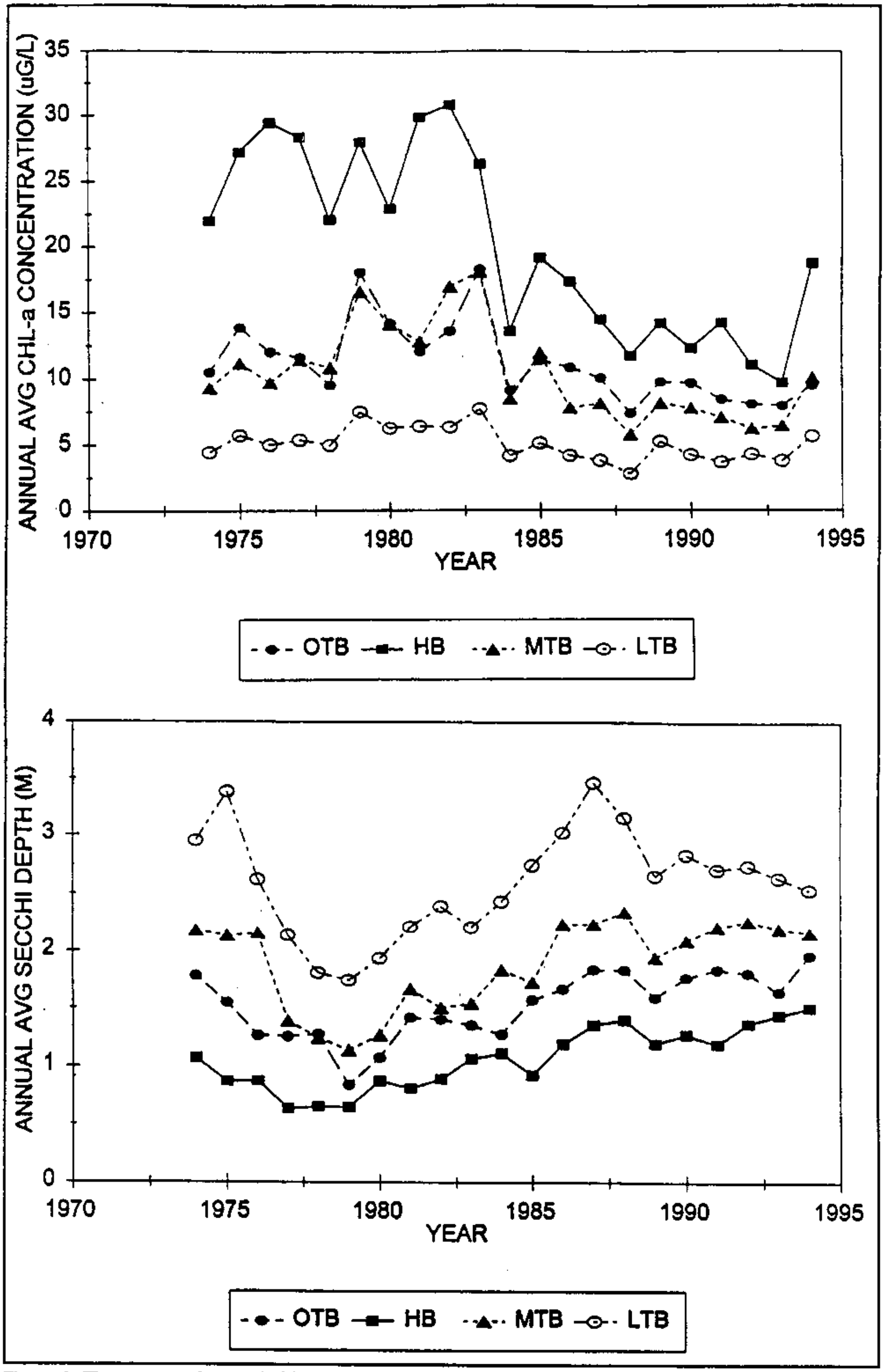

Figure 2. Time series of annual chlorophyll-*a* concentrations (top) and Secchi disk depths (bottom) in four major bay segments (OTB, HB, MTB, LTB), 1974–1995. Source: Boler 1995.

In general, trends in water clarity (Secchi disk depth) observed in these bay segments since 1974 have tended to mirror trends in chlorophyll-*a* concentrations, declining from 1974 through 1979 and increasing from 1980 through 1995 (Fig. 2). Plotting the annual average Secchi disk depths versus the annual average chlorophyll-*a* concentrations produces evidence of a curvilinear inverse relationship which can be linearized by log-transforming the chlorophyll-*a* concentration values (Fig. 3).

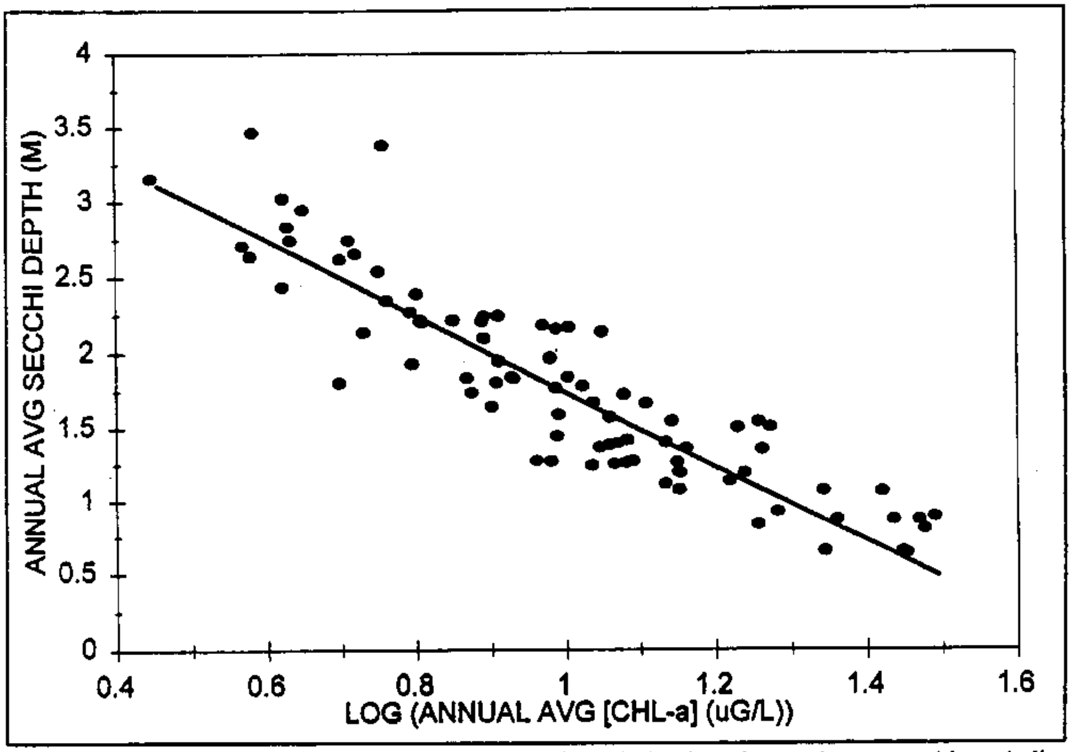

Figure 3. Relationship between annual average Secchi disk depth and annual average chlorophyll-*a* concentration in four major bay segments (OTB, HB, MTB, LTB) during the period 1974–1995. Regression line: Secchi depth=4.2-2.4(log[Chl-*a*]), R^2=0.82. Source: Boler 1995.

RELATIONSHIP BETWEEN WATER CLARITY AND CHLOROPHYLL-*a* CONCENTRATION

Janicki and Wade (1996) used regression analysis to relate monthly EPC water clarity measurements to monthly chlorophyll-*a* concentrations in the four largest bay segments (HB, OTB, MTB, and LTB). Least squares regression methods were used to estimate parameters for the equation:

$$\ln(C_{t,s}) = \alpha_{t,s} + \beta_s\,[\ln(Z_{t,s})]$$

where $C_{t,s}$ = average chlorophyll-*a* concentration at month t and bay segment s,
$Z_{t,s}$ = the depth to which 20.5% of subsurface irradiance penetrates at month t and bay segment s, and
$\alpha_{t,s}$, β_s = regression parameters.

An R^2 value of 0.67 was obtained for this model, using monthly (EPC) data from HB, OTB, MTB, and LTB (Janicki and Wade 1996). The intercept term of the regression was allowed to vary between months to avoid potentially confounding effects of seasonality in the independent and dependent variables.

RELATIONSHIPS BETWEEN CHLOROPHYLL-*a* CONCENTRATION AND NITROGEN LOADING

Empirical Model

For the period 1985–1994, Janicki and Wade (1995, 1996) also used regression analysis to relate monthly EPC chlorophyll-*a* measurements to the monthly nitrogen loading estimates developed by Zarbock et al. (1994). A series of steady-state equations was used to model the intersegmental transport of water and nitrogen, where net transport rates were estimated based on observed salinity data and freshwater inflow estimates (Janicki and Wade 1996). The nonconservative behavior of nitrogen was accounted for in the intersegmental transport rates through a comparison of observed EPC nitrogen concentrations and nitrogen concentrations predicted based on the steady-state equations alone.

The estimated overall nitrogen loads (i.e., nitrogen loads from external sources and internal estuarine processes) to individual bay segments were related to observed chlorophyll concentrations using the regression model:

$$C_{t,s} = \alpha_{t,s} + \beta_s L_{t,s}$$

where $C_{t,s}$ = average chlorophyll-*a* concentration at month t in bay segment s,
$L_{t,s}$ = overall nitrogen load at month t to bay segment s, and
$\alpha_{t,s}$, β_s = regression parameters.

An overall R^2 value of 0.69 was obtained using this relationship.

Mechanistic Model

For the period 1985–1994, Martin et al. (1996) used the U.S. EPA's "water analysis simulation program" (WASP) model to relate monthly (EPC) chlorophyll-*a* measurements to the monthly nitrogen loading estimates developed by Zarbock et al. (1994, 1996). WASP is a dynamic, compartment-based ("box") modeling program which incorporates time-varying processes such as point and non-point source nutrient loadings, boundary exchanges, advection, dispersion, phytoplankton growth and respiration, microbial nutrient transformations, particle settling, and benthic nutrient releases to simulate water quality responses to changing nutrient loads (Ambrose and Martin 1990, Martin et al. 1990). The model is based on the principle of mass conservation, and divides the water body under study into a set of comparably-sized "boxes" or segments within which water quality conditions are relatively homogeneous. Masses of water, nutrients, and other constituents of interest which enter a segment either exit the segment by physical transport and chemical or biological transformation or accumulate within the segment. For water bodies represented by a series of segments, transport can occur between segments or across the system's boundaries. Through detailed bookkeeping of the simulated movement of constituents between segments and accumulation within segments, the model computes predicted concentrations and fluxes of the water quality constituents of interest.

The eutrophication component of the model simulates the transport and transformations of up to eight state variables representing four interacting systems (phytoplankton kinetics, phosphorus cycle, nitrogen cycle, and dissolved oxygen balance) by solving a general mass balance equation for each state variable. The eutrophication state variables and interactions included in the model are summarized in Figure 4.

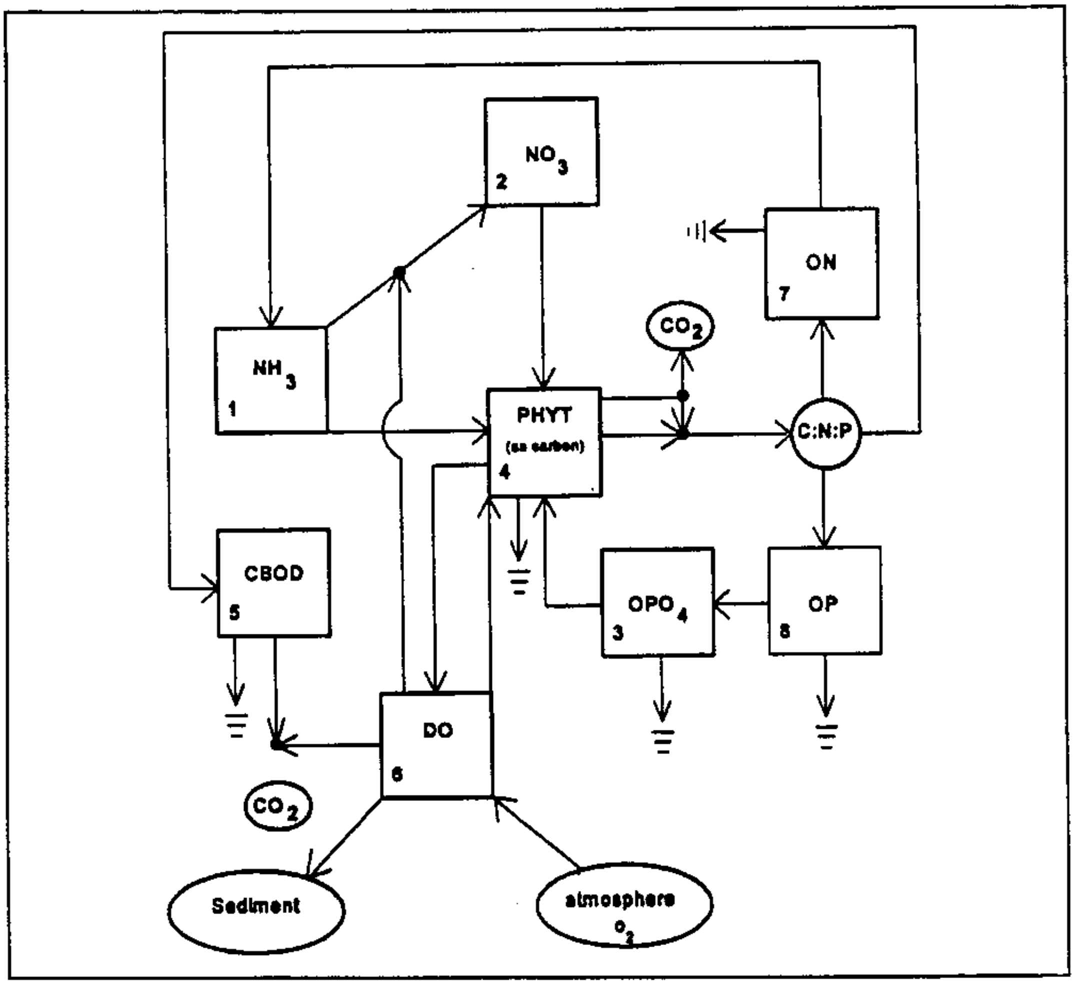

Figure 4. WASP model state variables and simulated interactions. Source: Martin et al. 1996.

Phytoplankton growth is simulated in the WASP model using a postulated maximal growth rate and computed "multipliers" which may range in value from zero (no algal growth) to one (maximal algal growth) depending on the simulated availability of light (photosynthetically active radiation) and the macronutrients nitrogen and phosphorus. The computed multipliers for these factors, from a representative subset of model segments in OTB, HB, MTB, and LTB, are shown in Figure 5. The computed multipliers indicate that light availability is the primary factor limiting phytoplankton growth rates in the Tampa Bay model.

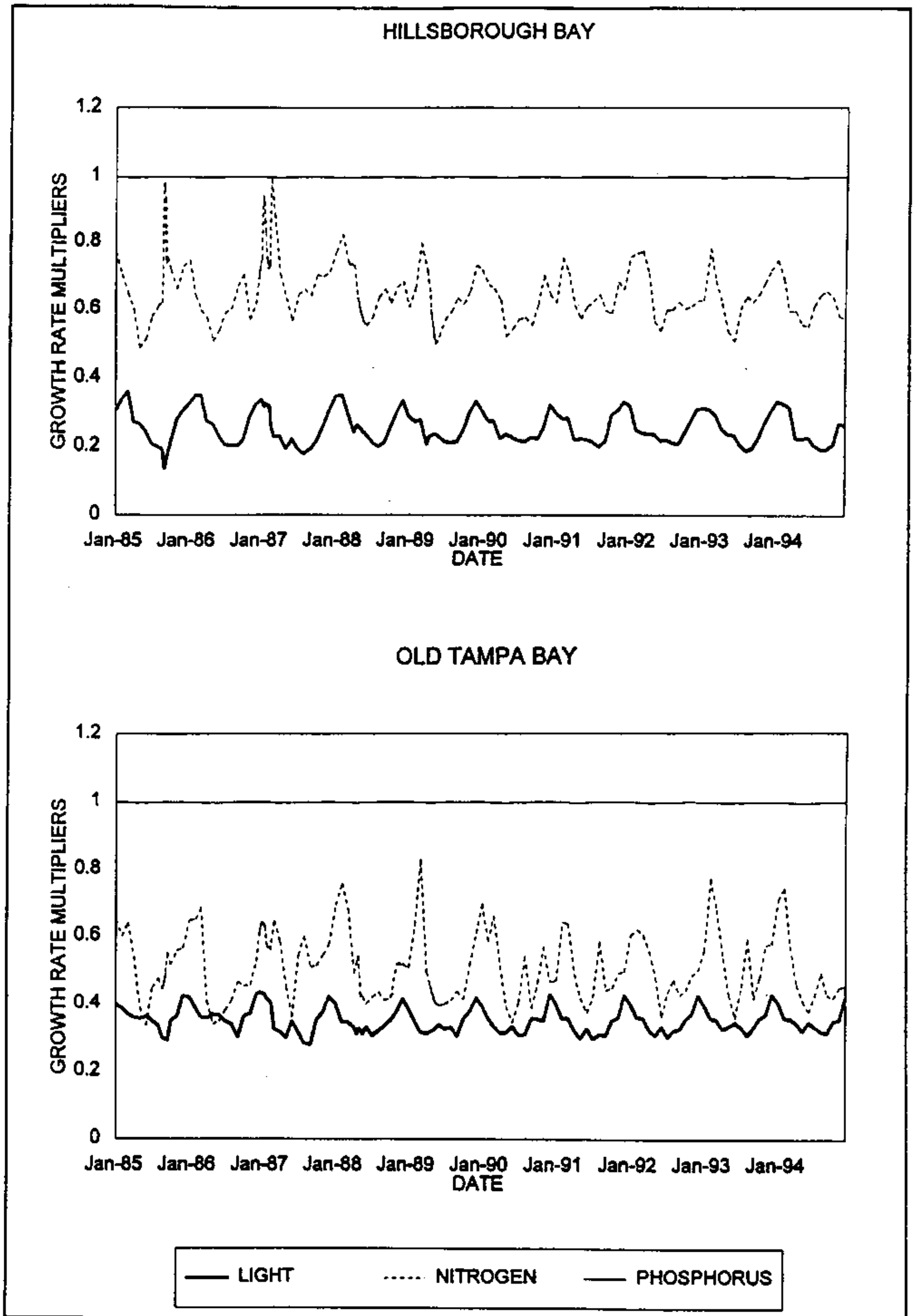

Figure 5. Predicted algal growth multipliers for macronutrients (N, P) and light in selected Tampa Bay model segments. Source: Martin et al. 1996.

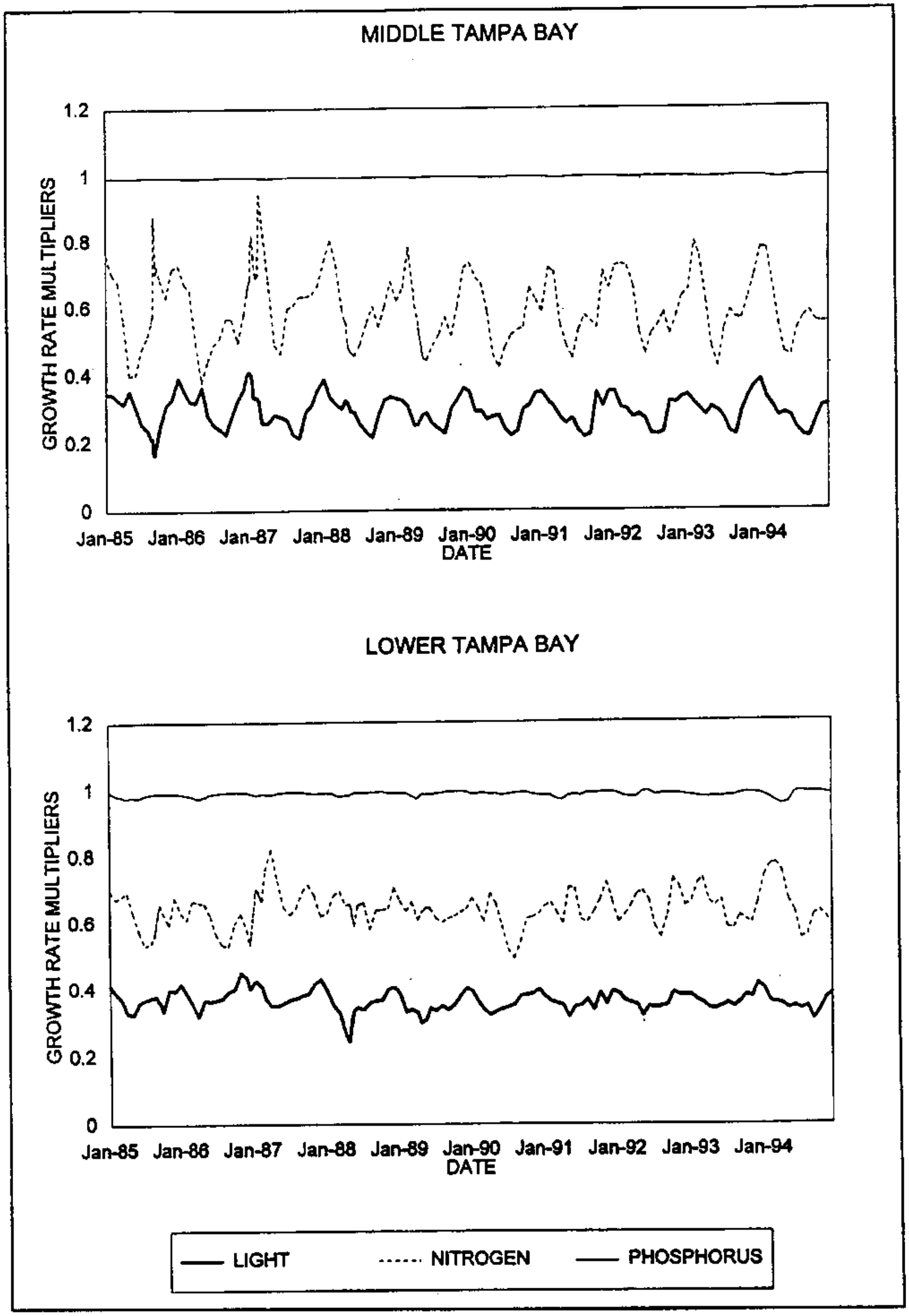

Figure 5 continued. Predicted algal growth multipliers for macronutrients (N, P) and light in selected Tampa Bay model segments. Source: Martin et al. 1996.

Although light appears to be the chief limiting factor in this model application (Fig. 5), the model also predicts a positive relationship between nitrogen loading and chlorophyll-*a* concentration within the bay (see Figure 6 below). Simulated increases in inorganic nitrogen loadings, such as those that occurred in Tampa Bay as a result of ammonia spills during 1985 and 1987, also produced substantial increases in model-predicted phytoplankton growth rates and chlorophyll-*a* concentrations (Martin et al. 1996). These responses indicate that nitrogen availability plays an important role in determining phytoplankton growth rates within the model (Martin et al. 1996). Phosphorus is present in sufficiently high concentrations in all modeled bay segments that its availability does not appear to limit phytoplankton growth.

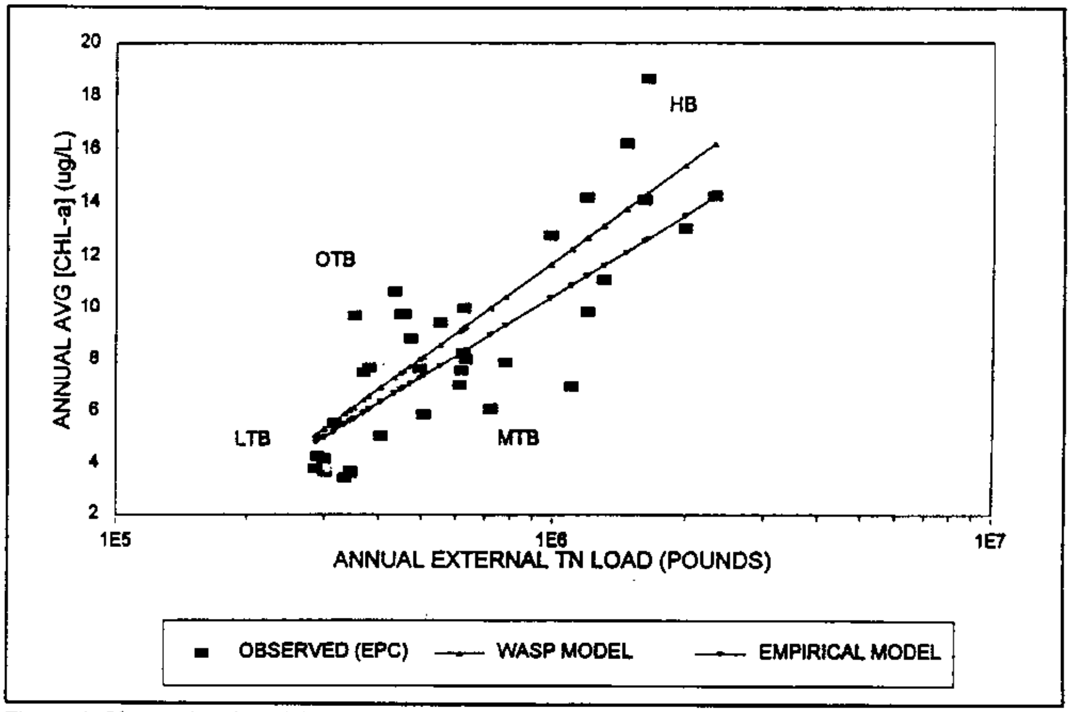

Figure 6. Observed and model-predicted relationships between annual TN loading and annual average chlorophyll-*a* concentration in major Tampa Bay segments, 1986–1994 (1985 omitted due to gaps in measured chlorophyll-*a* values). Regression equations: (empirical model) chlorophyll-a=-51.67 +4.49(log TN load), R^2=0.66; (WASP model) chlorophyll-a=-61.74+5.32(log TN load), R^2=0.61. Sources: Boler 1995; Janicki and Wade 1996; Martin et al. 1996.

Comparison of Model Predictions: Chlorophyll-*a* Concentration vs Total Nitrogen Load

Observed and model-predicted relationships between annual average chlorophyll-*a* concentrations and estimated annual nitrogen loadings for the four largest bay segments are shown in Figure 6. At the annual time scale Hillsborough Bay received the largest estimated TN loadings and produced the highest observed chlorophyll-*a* concentrations, while Lower Tampa Bay was the site of lowest external loadings and chlorophyll-*a* concentrations. Old Tampa Bay and Middle Tampa Bay exhibit intermediate values of both parameters. Relationships predicted by the empirical and WASP models were summarized by regressing predicted (average annual) chlorophyll-*a* concentrations against predicted annual nitrogen loadings. In general, both models appear successful in describing the relationship between estimated total nitrogen loadings and chlorophyll-*a* concentrations observed in the major bay segments during the 1985–1994 period of interest (Fig. 6).

SIMULATED NITROGEN FLUXES

The detailed bookkeeping performed by the WASP model as part of its mass balance calculations also allows the user to estimate the simulated fluxes (in units of mass/time) of nitrogen forms and other water quality constituents within and between model compartments. These flux estimates can be used provide insights regarding interactions that exist between external nutrient loadings, advective and dispersive transport, and internal regeneration and recycling (e.g., through algal and microbial food webs) in the Tampa Bay model. To the extent that the model resembles the actual bay, such insights may prove helpful to resource managers in the development of water quality management targets and strategies.

Total Nitrogen

Simulated annual fluxes of total nitrogen for the four largest bay segments (HB, OTB, MTB, and LTB), averaged over the period 1985–1994, are shown in Figure 7. Major sources of total nitrogen for the water column include the annual external load from the watershed, and microbial and benthic processes which recycle nitrogen within the water column and between the water column and the underlying sediments. The major loss term for the water column is advective and dispersive transport, which moves nitrogen (in the form of algal cells and particulate and dissolved nitrogen) between bay segments and between the bay and the Gulf of Mexico. When summed over the four segments, the estimated annual loss of total nitrogen via advective and dispersive transport (15,400 metric tons/year) is about 4.5 times larger than the estimated annual load (3,400 metric tons/year) received from the watershed, suggesting that the overall mass of nitrogen in the bay may be relaxing to a lower level following several decades of unusually large external loadings.

Phytoplankton growth and respiration processes represent a small loss term for each bay segment, which is the (net) difference between the inorganic nitrogen removed from the water column by growing algal cells and the (primarily) organic nitrogen returned to the water column as a result of cell respiration and death. The fact that the estimated net flux due to phytoplankton growth and respiration processes is negative presumably reflects a permanent loss of some phytoplankton-associated nitrogen from the water column each year as a result of transport from the system or deep burial in the sediments.

Summation of these fluxes produces a relatively small (1,500 metric tons/year) net loss term, indicating that, on average, the mass of total nitrogen in the modeled water column of the largest bay segments declined by this amount annually during the period 1985–1994.

Inorganic Nitrogen

The total nitrogen fluxes shown in Figure 7 include several nitrogen (e.g., organic nitrogen) forms which are not readily available to phytoplankton to support cell growth and associated changes in water-column chlorophyll-*a* concentrations. Because phytoplankton cells preferentially remove dissolved inorganic nitrogen (e.g., nitrate, ammonia) from the water column to support growth, fluxes of inorganic nitrogen may be more relevant than total nitrogen fluxes for Tampa Bay management efforts.

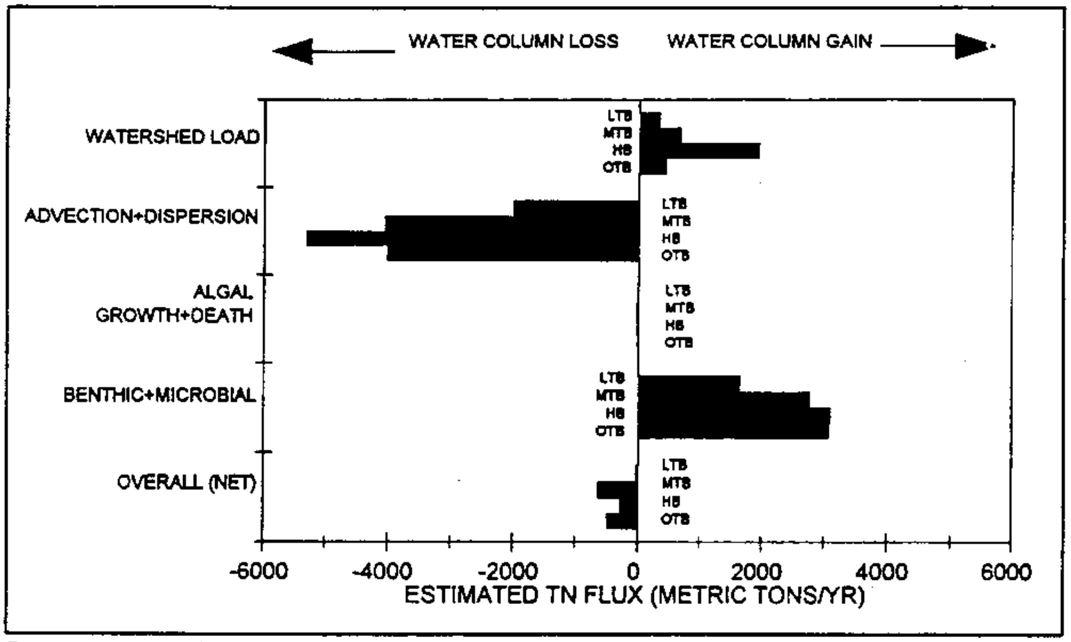

Figure 7. Annual fluxes of total nitrogen estimated by the WASP model. Source: Martin et al. 1996.

Simulated annual fluxes of dissolved inorganic nitrogen for the four largest bay segments, averaged over the period 1985–1994, are shown in Figure 8. Benthic and microbial processes which transform organic nitrogen to inorganic nitrogen and release it to the water column represent the major water column source term (58,900 metric tons/yr). Among these processes, microbial mineralization of organic nitrogen to ammonia, and benthic ammonia releases, contribute the highest flux rates. Estimated fluxes due to nitrification are an order of magnitude smaller than those associated with ammonia production, a model result which may explain the large ammonia:nitrate ratios typically observed in the water column of Tampa Bay. Estimated external loadings of inorganic nitrogen from the watershed represent a relatively minor source term (2,300 metric tons/yr).

Phytoplankton uptake represents the major loss term for inorganic nitrogen from the water column (60,300 metric tons/yr), while advective and dispersive transport cause minor losses (2,300 metric tons/yr). Simulated losses via denitrification are small (20 metric tons/yr).

Although external loadings from the watershed and airshed represent the bay's ultimate source of nitrogen, the comparable magnitudes of the microbial/benthic source term (58,900 metric tons/yr) and the phytoplankton-uptake loss term (60,300 metric tons/yr) suggest that microbial and benthic processes are providing most of the inorganic nitrogen fueling phytoplankton growth on an annual basis.

Summation of the estimated fluxes produces a small net loss term (1,300 metric tons/year), the model's estimate of the annual change in the mass of inorganic nitrogen present in the water column of the four largest bay segments during the period 1985–1994.

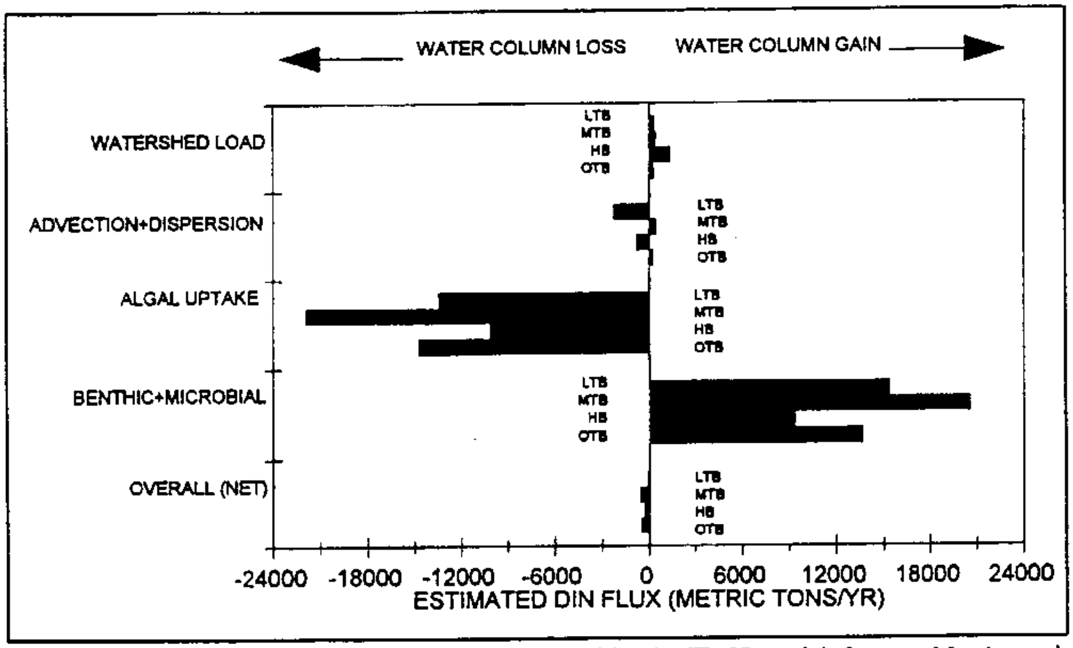

Figure 8. Annual fluxes of inorganic nitrogen estimated by the WASP model. Source: Martin et al. 1996.

BIOASSAY RESULTS

Rodriguez (1991) examined potential nitrogen limitation in Tampa Bay and the 12 ppt salinity zone of the Little Manatee River by performing monthly nitrogen addition bioassays using water samples from the two systems. Her results demonstrated linear relationships between the final yield of chlorophyll-*a*, particulate carbon, and particulate nitrogen over a range of inorganic nitrogen additions. Additional examination of Rodriguez's (1991) data, including corrections for chlorophyll-*a* generated using ambient inorganic nitrogen present in the water samples, produced highly significant linear relationships between annual average chlorophyll-*a* yields and nitrogen additions (Vargo 1996) (Fig. 9). Yield per unit nitrogen addition was greater in the Little Manatee River than in the Tampa Bay bioassays, primarily because of higher yields during the wet season, but the difference is not significant. Yields observed in the monthly bioassays varied seasonally at both locations (Fig. 10). Chlorophyll yield during the summer wet season was lower than during the winter dry season in the Tampa Bay bioassays, while the opposite was true for the Little Manatee River samples. Yields ranged from 0.08 to 2.9 μg Chl/μM nitrogen added, with median values of 1.0 and 1.96 μg Chl/μM nitrogen for the Tampa Bay and Little Manatee River samples, respectively. These values are comparable to estimates from a variety of other systems (e.g., Thomas 1970, Graneli 1978, Gowan et al. 1992). Rodriguez's (1991) results provide experimental support for the model-based hypothesis that nitrogen is the primary limiting nutrient in Tampa Bay (and the 12 ppt salinity zone of the Little Manatee River), as well as demonstrating a direct relationship between inorganic nitrogen inputs and changes in chlorophyll-*a* concentration.

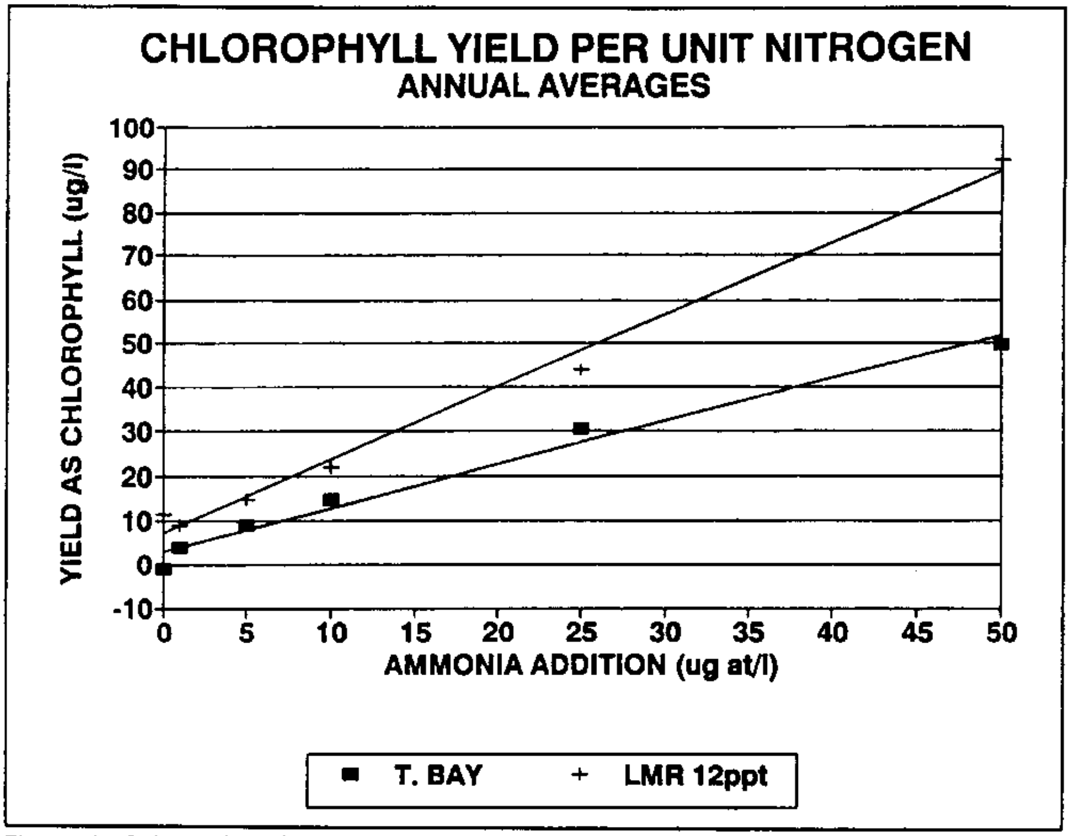

Figure 9. Relationships between the annual average chlorophyll yields and added ammonia concentration determined in laboratory bioassays using natural water samples. Regression equations: (Tampa Bay) Yield=3.09+0.98(N), R^2=0.98; (Little Manatee River) Yield=7.41+1.64(N), R^2=0.99. Source: Rodriguez 1991; Vargo 1996.

A second element in conceptualizing the response of Tampa Bay phytoplankton populations to external nitrogen loadings is an understanding of the relative proportions of available nitrogen contributed by external loadings and estuarine regeneration and recycling processes. One method of estimating these values, based on WASP model output, is shown in Figure 8. As an independent assessment, Vargo (1996) used a conversion factor [DIN = 0.47(TN)] to convert the estimated annual total nitrogen (TN) loadings reported by Morrison (1992) to dissolved inorganic nitrogen (DIN) loadings. These estimated dissolved inorganic nitrogen loadings were then compared to the nitrogen masses estimated to be necessary to support measured nitrogen uptake (Vargo et al. 1993), primary production (Johansson et al. 1985), and dissolved inorganic nitrogen standing stocks (based on EPC monitoring data). As shown in Table 1, the estimated dissolved inorganic nitrogen loads appear to provide a relatively low and highly variable fraction (0.9% to 17%) of the nitrogen required to support the phytoplankton populations observed in the water column of Tampa Bay. The remaining nitrogen required to support water column phytoplankton production appears to be supplied by the regeneration of inorganic nitrogen through microbial and benthic processes within the estuary. At a conceptual level, results of the bioassay experiments and associated analyses by Rodriguez (1991) and Vargo (1996) thus tend to corroborate the model-based flux estimates shown in Figure 8.

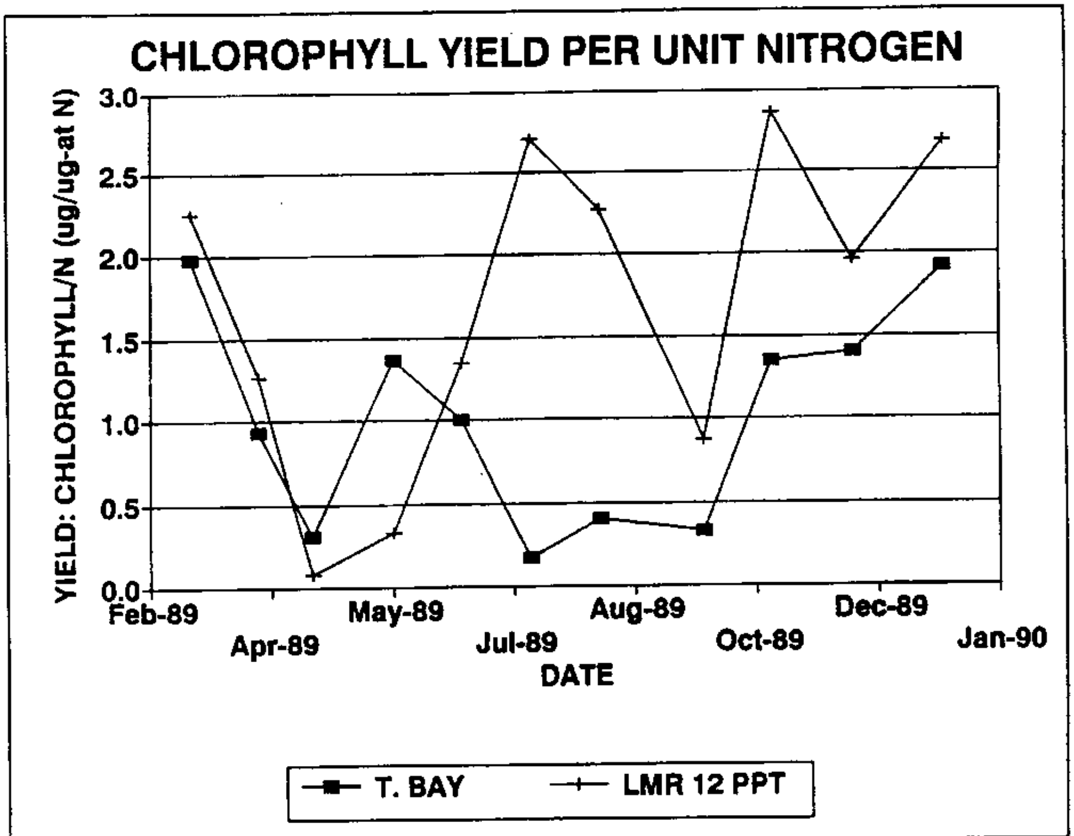

Figure 10. Monthly values of chlorophyll-*a* yield (μg) per unit nitrogen (μM) for bioassay samples from Tampa Bay (TB) and the 12-ppt salinity zone of the Little Manatee River (LMR). Source: Rodriguez 1991.

RESOURCE MANAGEMENT IMPLICATIONS

The fact that estimated annual nitrogen loads during the period 1985–1994 (Fig. 1) have remained substantially larger than the estimated "benchmark" (ca. 1940) values, despite the very large load reductions that occurred between 1979 and 1985 (Johansson 1991), reflects the magnitude of human population growth that has taken place in the watershed and airshed since the benchmark period. Annual nitrogen loads are projected to increase an additional ~7% bay-wide between the 1985–1994 period and the year 2010 (Fig. 1), a trend which, if continued over time, will make a substantial amount of additional nitrogen available to support increased chlorophyll-*a* concentrations (Fig. 6) and reduced water clarity levels (Fig. 3) in the bay in future years.

Results from the mechanistic water quality model (Fig. 8) and empirical studies (Table 1) suggest that the mechanisms underlying observed relationships between nitrogen loading rates and chlorophyll-*a* concentrations in the bay are relatively complex. Only a small proportion of the phytoplankton standing stock (as estimated by chlorophyll-*a* concentration) observed in the bay at any given time appears to be supported by loadings of "new" nitrogen recently delivered from the watershed. A much larger proportion is apparently supported by the regeneration of inorganic nitrogen within the bay, through microbial and benthic pathways, from pools of recycled organic nitrogen. Taken together, the modeled flux rates and bioassay results imply that short-term changes in external nitrogen loads, particularly if they are small relative to the size of the internal nitrogen pools and occur following a prolonged period of elevated loadings, may not produce immediately detectable changes in chlorophyll-*a* concentrations. An example of this situation appears to have occurred in the bay during the period 1980–1983. Although the City of Tampa's municipal wastewater treatment plant at Hookers Point upgraded to advanced wastewater treatment nitrogen discharge standards in 1979, producing a marked reduction in annual nitrogen loadings to Hillsborough Bay (Johansson 1991), reductions in chlorophyll-*a* concentrations were not observed until 1983 in Hillsborough Bay and 1984 in other bay segments (Fig. 2). As noted by Johansson (1991), the observed lag between nitrogen load reduction and chlorophyll-*a* response appeared to represent the time period necessary for the bay's internal processes to equilibrate to the new level of nitrogen input following several decades of quite elevated anthropogenic loads.

Table 1. Nitrogen requirements and nitrogen loadings for three regions of Tampa Bay. Loadings calculated as total nitrogen (TN) or dissolved inorganic nitrogen (DIN = 0.47[TN]).

		METRIC TONS/YEAR					
SEGMENT	PROCESS	N REQUIRED[a]	N LOADINGS[b] TN	N LOADINGS[b] DIN	STANDING STOCK[c] DIN	PERCENT REQUIREMENT MET[c] TN	PERCENT REQUIREMENT MET[c] DIN
HB	N uptake[a]	8520	983	462		11.5	5.4
			3060	1440		35.7	16.9
	1° prod.[d]	8200				11.9	5.6
						37.3	17.5
					1780		26–81
MTB	1° prod.	8600	550	260		6.4	3.0
			2530	1190		29.4	13.8
LTB	N uptake	21210	450	210		2.1	0.9
			2470	1160		11.6	5.4
	1° prod.	6900				6.5	3.0
		12700				3.5	9.1
					2610		8–44

[a]Calculated from C:N ratio of 5.7:1 by weight for primary production values (Vargo 1996). N uptake rates from Vargo et al. 1993.
[b]Estimated loadings from Morrison (1992), representing high and low flow years.
[c]From Vargo 1996.
[d]Range of primary production values from Johansson et al. 1985.

Unlike short-term fluctuations in external nitrogen loads, sustained directional (increasing or decreasing) loading changes which are sufficient to alter the sizes of the internal nitrogen pools should produce changes in bay water quality that are observable over time. External loads from the watershed and airshed represent the ultimate source of the nitrogen present in the internal pools, and the water quality improvements observed in the bay during the 1980s (Fig. 2) have demonstrated the system's long-term responsiveness to sustained directional loading changes.

Unfortunately, neither the empirical nor mechanistic water quality models addressed here are capable of simulating long-term changes in some internal flux rates (e.g., sediment nutrient release rates) that would be expected to occur in response to long-term changes in external loadings. This problem is not unique to the Tampa Bay modeling effort, and while efforts to model estuarine sediment/water column interactions have been initiated and improved in recent years, accurate prediction of long-term interactions, and the availability of data to test the accuracy of model predictions, remain problematic. Because of this limitation, it should not be assumed that the available Tampa Bay models will provide unbiased predictions of water quality conditions (e.g., chlorophyll-*a* concentrations, Secchi disk depths) in the bay following long-term increases or decreases in nitrogen loadings. In fact, because both models are calibrated to current (1985–1994) conditions, when nitrogen concentrations in the bay appear to be relaxing to lower levels following several decades of elevated external loads, their predictions concerning future water quality conditions may contain some inherent bias. Because neither model actively simulates changes in sediment nutrient exchange rates, both must assume that today's rates (for which the models are calibrated) will continue into the future. In reality, however, a prolonged period of increasing external loads would presumably cause (after some time lag) an increase in sediment release rates, while a prolonged period of declining external loads would eventually cause sediment release rates to fall. As a result, both models may tend to underestimate the water quality benefits that would occur as a result of long-term load reductions as well as the negative water quality impacts that would occur as a result of long-term load increases. This is a potentially important issue from a water quality management perspective, and should be kept in mind when interpreting any long-term predictions generated by the models. Relatively frequent model recalibration (e.g., on a three- to five-year cycle) would be helpful to reduce the likelihood that an inherent model limitation might contribute to an incorrect watershed management decision.

ACKNOWLEDGMENTS

The projects summarized here were funded by the Tampa Bay National Estuary Program (TBNEP) and the Southwest Florida Water Management District's Tampa Bay Surface Water Improvement and Management (SWIM) program. The authors thank the staff of the Environmental Protection Commission of Hillsborough County for providing access to that agency's water quality database, and the TBNEP's Technical Advisory Committee for guidance during the development of the empirical and mechanistic water quality models.

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