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Setting Load Limits for Nutrients and Suspended Solids Based upon Seagrass Depth-limit Targets

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ABSTRACT: Total nitrogen (TN), total phosphorus (TP), and total suspended solids (TSS) loadings [log (kg ha^{-1} yr^{-1})] were regressed against seagrass depth limits (percent of depth-limit targets) to back-predict the load limits or allocations (kg ha^{-1} yr^{-1} or kg yr^{-1}) necessary to meet targeted seagrass depth limits in the Indian River and Banana River (IRBR) lagoons, Florida. Because the load allocations can be applied as total maximum daily loads (TMDL) for the IRBR (U.S. Environmental Protection Agency mandate), the method and results are developed and presented toward that end. The regression analyses indicate that the range of surface-discharge load limits (nonpoint + point source), per watershed area, required to achieve the desired depth limits for seagrass in the IRBR are approximately 2.4–3.2 kg ha^{-1} yr^{-1} TN, 0.41–0.64 kg ha^{-1} yr^{-1} TP, and 48–64 kg ha^{-1} yr^{-1} TSS. This simple regression method may have application to other shallow estuarine lagoons or bays where seagrass growth is limited by light and water transparency, water transparency is strongly affected by watershed pollutant loadings, water residence times are sufficiently long to allow seagrass coverage to respond to and covary with total load inputs, and multiyear monitoring has yielded sufficient variability in both pollutant loadings and seagrass coverages to develop a statistically meaningful relationship.

Introduction

BACKGROUND AND PURPOSE

For the lagoon estuaries of Indian River and Banana River (IRBR; Fig. 1), Florida, as well as for other estuaries, the development of seagrass restoration requirements is an important management objective. Seagrass coverages near major metropolitan areas in the IRBR basin are less than they were in 1943, the earliest year of documented coverage (Steward et al. 2003). The seagrass coverage loss between 1943 and 2003 was 52% (1,147 to 545 ha) in a 35-km reach from just north of Melbourne to Palm Bay, and a 48% loss in the Satellite Beach segment (166 to 86.6 ha; SJRWMD unpublished data; see Fig. 1 for location of segments).

In estuaries where chronic seagrass loss is caused by increased light attenuation, seagrass requirements emphasize subsurface light availability, water transparency, and related water quality variables such as nutrients and suspended matter (Kenworthy and Haunert 1991; Dennison et al. 1993; Moore et al. 1996; Tomasko et al. 1996; Kemp et al. 2004). Light explains about 52% of the depth-limit variability of seagrass in the IRBR (Virnstein et al. 2002).

Total suspended solids (TSS) are primarily responsible for attenuating downwelling light in the IRBR, followed by either color or chlorophyll *a* (chl *a*; as a proxy for phytoplankton biomass) depending upon time and location (Hanisak 2001; Steward et al. 2003). Trefry et al. (1987, In press) found 50% or more of the TSS in the IRBR to be composed of fairly recent (≤ 50 yr) terrestrial organic matter, clays, and silts.

With respect to phytoplankton production, either nitrogen (N) or phosphorus (P) can be limiting in the IRBR system (Hanisak 2001; Phlips et al. 2002). P can be the limiting nutrient in the northern Indian River Lagoon (IRL), whereas N is typically limiting in the central IRL, although both nutrients are generally in surplus (Phlips et al. 2002). In an urban basin of the central IRL (Turkey Creek, Palm Bay), Trefry and Feng (1991) found that N is limiting in the wet season (summer-fall) and P is limiting in the dry season (winter-spring) and at the onset of the seagrass growth season.

Perhaps more significant than phytoplankton production in the IRBR is the large abundance of macroalgae (Virnstein and Carbonara 1985) and epiphytic algae on seagrass blades (Miller 1997; Hanisak 2001), both of which can affect the depth distribution of seagrass (Hanisak 2001; Hauxwell et al. 2001).

These studies, taken together, reveal the need to reduce tripton levels in the IRBR, particularly the input of terrigenous TSS, and to limit loadings of N and P, as well as color or dissolved organic matter (DOM), to promote restoration of seagrass coverage.

In this paper, we present a regression approach for determining watershed areal load limits for total

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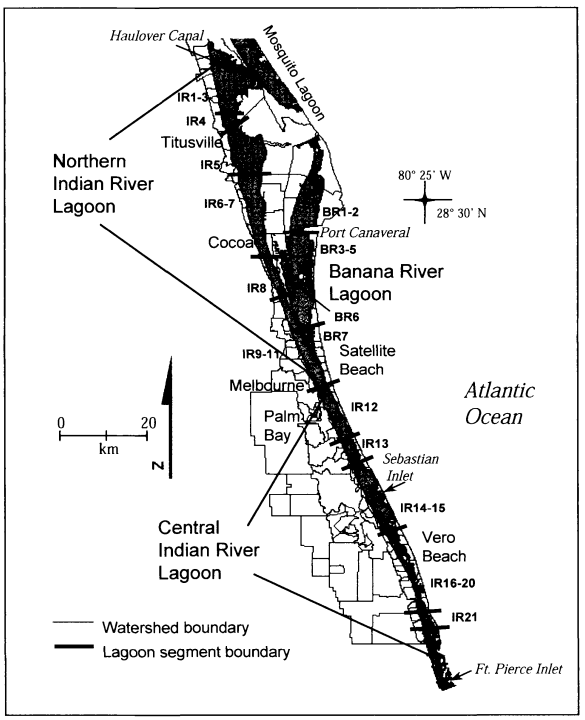

Fig. 1. Indian River and Banana River lagoons comprising three sublagoons (northern Indian River Lagoon, central Indian River Lagoon, and Banana River Lagoon) and 15 segments. Watershed boundaries are also depicted.

nitrogen (TN), total phosphorus (TP), and TSS that correspond to seagrass coverage depth-limit targets. Areal load limits (kg ha^{-1} yr^{-1} or $season^{-1}$) can be used to calculate total watershed load allocations (kg yr^{-1}) or total maximum daily loads (TMDL) required by U.S. Environmental Protection Agency (EPA; Clean Water Act of 1972) to improve impaired waters in the IRBR system.

A similar regression approach was used to predict the change in eelgrass (*Zostera marina*) coverage as a function of N loading rate in Waquoit Bay, Massachusetts (Short and Burdick 1996; Bowen and Valiela 2001; Driscoll et al. 2003). Whereas the purpose of those papers, and their inclusion of a Waquoit Bay regression model, was to demonstrate the effect of N enrichment on seagrass coverage, it is the purpose of this paper to demonstrate that such models can also be useful in setting nutrient and TSS load limits relative to predetermined seagrass coverage targets.

This regression approach does not address color because of constraints in developing loading rates for watershed color or related DOM. A separate study is underway to determine a color load threshold based on possible interrelationships among color, organic carbon, runoff volume, and seagrass salinity and light requirements.

General Description of the IRBR System and Watershed

The area of interest in the IRBR system extends 150 km along Florida's east central coast and includes two contiguous, yet distinct estuaries, Indian River Lagoon (IRL) and Banana River Lagoon (BRL; Fig. 1). Large spatial variability is a major characteristic of the IRBR system and is well documented with respect to biology and hydrography (Gilmore 1985; Steward and VanArman 1987; Virnstein 1990). Much of the system's biological diversity can be explained by its geographic location and length, comprising a transitional zone between two biotic provinces: the temperate Carolinean province and subtropical Caribbean province. For example, the IRBR is home to seven species of seagrass (*Halodule wrightii*, *Syringodium filiforme*, *Thalassia testudinum*, *Halophila decipiens*, *Halophila engelmanii*, *Halophila johnsonii*, and *Ruppia maritima*), the most diverse assemblage of seagrass species found in any estuary within the United States (IRL National Estuary Program 1996).

The IRBR system is shallow and microtidal. The average depths of the IRL and BRL, relative to mean water level (MWL), are 1.8 and 1.7 m, respectively. Semidiurnal tides propagate through two inlets, Sebastian and Ft. Pierce, located in the central IRL (Fig. 1). There is also an intermittent connection between the Atlantic Ocean and the BRL through a navigational canal-lock facility at Port Canaveral. Flushing of northern IRL is enhanced by wind and tidal excursions of water to and from Mosquito Lagoon through Haulover Canal. Tidal forcing into the system is weak, exerting tidal amplitudes less than 10 cm, except within a few kilometers of the inlets where amplitudes increase two to three-fold (Smith 1987). Weekly wind or seasonal oceanic forcings actually cause greater water level variations: $\pm$ 0.1–0.3 m (Smith 1993, 2001).

Average water residence times vary throughout the system, primarily as a function of distance from inlet. The entire reach of the central IRL from Melbourne southward is $\leq$ 28 km of either Sebastian Inlet or Ft. Pierce Inlet, much closer to inlets and more influenced by oceanic exchange than the northern IRL and BRL. The average residence time in the central IRL can vary between a week to over two months, depending on time of year and distance from an inlet, and is typically 10 to 15 times shorter than in the northern IRL and BRL, respectively (Christian unpublished data).

The mean annual rainfall in the IRBR basin is about 127 cm (Steward and VanArman 1987; Knowles 1995). Rainfall runoff enters the IRBR via

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diffuse surface flow and through creeks, ditches, and outfall structures. From Satellite Beach and Melbourne southward (Fig. 1), the drainage features also include extensive canal networks, most of which were constructed between 1920 and 1940, facilitating agricultural and urban development. The potential annual runoff volume is higher in the central IRL (5,410 m^3 ha^{-1} yr^{-1} or a total of 665 million m^3 yr^{-1}) than in the less developed northern IRL (4,400 m^3 ha^{-1} yr^{-1} or a total of 218 million m^3 yr^{-1}), based on model results derived from 2000 land use and mean annual rainfall (SJRWMD unpublished data). For the IRBR overall, the yearly surface runoff volume under current conditions represents a 17% increase over rates in 1943; the increase in annual nutrient loads since 1943 are disproportionately greater with increases of 30% TN and 50% TP in the northern IRL, and increases of 84–85% TN and 161–183% TP in the central IRL and BRL (Steward et al. 2003).

Surface runoff constitutes over 80% of the external annual load of nutrients and 99% of the TSS load to the IRBR (SJRWMD unpublished data). Treated wastewater loads constitute $\leq$ 2% of the nutrient and TSS loading to the IRBR system (Steward et al. 2003; Florida Department of Environmental Protection unpublished data). The balance of the external nutrient loading to the IRBR is contributed by atmospheric and groundwater sources.

Methods

Pollutant Load Limits—Regression Analyses

The method for setting load limits was based on a simple model for the IRBR system in which seagrass depth limits are inversely related to watershed loadings of TN, TP, and TSS. The model is expressed as a series of linear regressions: seagrass depth limits, represented as percent departures from restoration depth-limit targets (y axis), regressed on log-transformed areal loadings of TN, TP, or TSS (x axis, Fig. 2). The regressions meet the basic assumptions of regression analysis including homoscedasticity, residual independence, and normal distribution. The loadings data were log-transformed not only to produce a normal data distribution, but also because the relationship between loading rates and seagrass depth limits is exponential. The exponential relationship between downwelling light and water quality variables that scatter and absorb light is well established (Kirk 1983; Davies-Colley et al. 1993; Gallegos and Kenworthy 1996), and a similar relationship is expected to exist between pollutant loadings and seagrass depths as mediated by light.

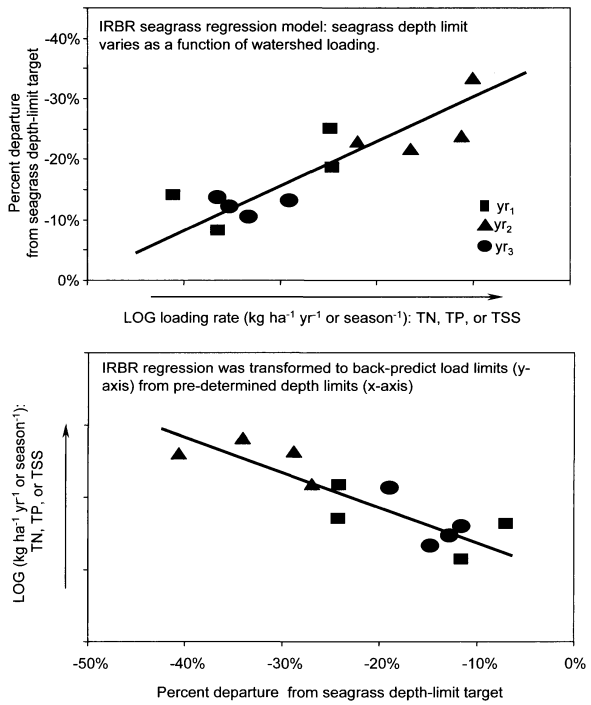

Fig. 2. Conceptual depictions of the IRBR seagrass versus loading model: coverage (percent of depth-limit target) regressed on TN, TP, or TSS loading rate (kg ha^{-1} yr^{-1}, log-transformed) and the model used as tool to determine load limits: log-transformed loading rate (y axis) regressed on percent departure from depth-limit target (x axis).

For the purpose of this study, we back-predicted load limits by regressing watershed loading rates (y axis) on seagrass depth limits (x axis, Fig. 2). The load limits (kg ha^{-1} yr^{-1}), corresponding to the targeted seagrass depth limits, were converted to total watershed allocations (kg yr^{-1}) or TMDL.

In order to construct the regression models, the IRBR system was divided into sublagoons and segments (Fig. 1). Segment-specific data sets comprising seagrass depth limits and modeled loading estimates (annual and seasonal) for TN, TP, and TSS were generated for those seagrass mapping years when both types of data were concurrent: 1943, 1996, 1999, and 2001. Each point in a regression plot represents the paired areal load and depth-limit data for an individual segment and mapping year (Fig. 2). A series of regression models were constructed separately for TN, TP, and TSS per sublagoon: northern IRL, central IRL, and BRL.

Segmentation

Segmentation was a two step process. Segment boundaries were located at the 13 causeway bridges

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that span the IRBR where disruptions in hydrodynamic circulation patterns have either been observed or presumed to exist (Evink 1980), and where changes in seagrass coverage patterns were apparent (Virnstein et al. 2003). Variability among segments was assessed with respect to turbidity and salinity, the water quality parameters with the greatest spatial variability. Contiguous segments were aggregated if no significant turbidity and salinity differences were found between them (via kriging and cluster analysis by Sigua et al. 1996). As a final result, 15 segments in the IRBR were established (Fig. 1; six segments in northern IRL, five in central IRL, and four in BRL).

Seagrass Depth-limit Data

Seagrass depth limits were determined from maps of seagrass coverage (March through mid September) overlaid on depth contours. The mapping of seagrass coverage was accomplished by interpretation of aerial photographs, and the depth contours were based on 1996 bathymetry (Coastal Planning and Engineering 1997). Median depth-limit targets were established for each segment based on a deep edge boundary delineating the union of seagrass coverages from seven mapping years (1943, 1986, 1989, 1992, 1994, 1996, and 1999) and the deepest extent to which seagrass had ever been mapped. More detail with regards to the IRBR seagrass depth-limit data and the setting of depth-limit targets is found in Steward et al. (2005).

With respect to the regression models, median seagrass depth limits for each segment and mapping year (1943, 1996, 1999, 2001) were represented as percent departures from that segment's median depth-limit target. Median depth-limit targets range from 1.2 to 1.8 m among the IRBR segments (Steward et al. 2005), and percent departures ranged from −5% to −55% (both occurred in segment IR9-11 in 1943 and 1996, respectively).

Because there were no data available for 2001 seagrass depth limits in BRL and northern IRL, only 1943, 1996, and 1999 were represented in the regression analyses for those two sublagoons.

Pollutant Loading Data

Annual and seasonal loads of TN, TP, and TSS from point (treated wastewater) and nonpoint (surface runoff) sources were estimated for a 12-mo period preceding the date of aerial photography: 1942–1943 loads for the 1943 mapping year, 1995–1996 loads for 1996, 1998–1999 for 1999, and 2000–2001 for 2001. For the seasonal analysis, we chose the wet season months of August through October (August and September are typically peak rainfall months) and the dry season months of February through April from the same 12-mo periods listed above.

Combined point and nonpoint source loads were estimated for the 1995–1996, 1998–1999, and 2000–2001 periods. Only nonpoint loads were estimated for 1942–1943 because there were no point source data for that period. It is assumed that any amount of point source loading during 1942–1943 would be quite low (given the prevalence of rural and undeveloped land use) and would have no bearing on the regression analyses.

Since 1995, point source loads have comprised such a small percentage ($\leq 2\%$; Steward et al. 2003) of the total external loading to IRBR that they do not contribute significantly to the regression models (confidence level $\alpha = 0.15$, stepwise regression analysis). Groundwater load estimations are available only as system-wide extrapolations and could not be incorporated into a stepwise regression. It is believed that the groundwater contribution would be similarly insignificant because it constitutes only 1% of the total external TN loading to the IRBR (Belaineh personal communication; SJRWMD unpublished data). Direct atmospheric loads constitute a larger percentage ($< 17\%$; SJRWMD unpublished data) of the external nutrient load, but, like the point source loads, they do not provide a significant contribution to the regression models ($\alpha = 0.15$).

Although point source loads are considered insignificant, we included them in the regression analyses for the sake of TMDL development. It is a requirement that both point and nonpoint sources are accounted in order to help determine the distribution of the total load allocation or TMDL between the two sources (Clean Water Act, 40 CFR Part 130; Section 403.067, Florida Statutes). Accordingly, the calculation of a total load allocation was made the same as that required for a TMDL, which is arithmetically expressed as follows:

$$\text{Total load allocation} = \text{TMDL} = (\Sigma\text{WLA} + \Sigma\text{LA}) - \text{MOS}$$

in which, ΣWLA = cumulative wasteload allocation assigned to point sources (wastewater treatment facilities), ΣLA = cumulative load allocation assigned to nonpoint anthropogenic sources and natural background sources, and MOS = margin of safety, which accounts for the uncertainty about the relationship of the pollutant load and the targeted resource (e.g., seagrass).

A TMDL does not necessarily need to be developed as a daily load; rather it can be developed according to any period (e.g., annual, seasonal) relevant or critical to the system's targeted resource

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TABLE 1. The major NPDES[a] wastewater treatment facilities that discharge to the IRBR system and their mean annual loads (2000–2005).

Sublagoons and Segments	NPDES Facility	Facility Type	Mean Annual Loads[b] (kg yr^{-1})		
			TN	TP	TSS
Northern IRL					
IR6-7	Cocoa	Domestic WWTP	3,338	816	454
IR8	Rockledge	Domestic WWTP	354	281	112
IR9-11	Melbourne	Reverse Osmosis	2,982	67.6	0
		Total:	6,674	1,165	566
Central IRL					
IR12	South Beaches	Domestic WWTP	68.5	16.8	331
IR12	Melbourne Grant Street	Domestic WWTP	36.7	1.8	40.8
IR14-15	Barefoot Bay	Domestic WWTP	56.7	9.5	68.9
IR16-20	Vero Beach	Domestic WWTP	5,559	420	1,065
IR16-20	Vero Beach	Reverse Osmosis	1,100	155	0
IR16-20	Indian River County Hobart	Reverse Osmosis	883	20.0	0
IR16-20	West Regional	Domestic WWTP	662	36.3	678
IR16-20	Indian River County South	Reverse Osmosis	2,051	163	0
		Total:	10,417	822	2,184
Banana River Lagoon					
BR3-5	Cape Canaveral	Domestic WWTP	992	68.9	843
BR3-5	Cocoa Beach	Domestic WWTP	6,242	861	1,463
		Total:	7,234	930	2,306

[a] The National Pollution Discharge Elimination System is a regulatory program of the U.S. Environmental Protection Agency. The program was delegated to Florida Department of Environmental Protection (FDEP) for management of sewage and stormwater in Florida. The NPDES wastewater facilities included in the point source load estimation are domestic wastewater and reverse osmosis treatment plants that discharge continuously or intermittently to the IRBR system.

[b] Mean annual loads were calculated from data obtained from FDEP. The mean annual loads were applied as wasteload allocations (WLA) for total maximum daily load (TMDL) development. Please note that the actual point source annual loads of 1995–1996, 1998–1999, and 2000–2001 were used in the regression models, not the mean annual loads.

or stress response (Greenfield personal communication).

Point source loads, annual and seasonal, were calculated using the National Pollutant Discharge Elimination System (NPDES) wastewater treatment facility database (flows and concentrations) maintained by the Florida Department of Environmental Protection (FDEP). The NPDES facilities considered in this study are listed in Table 1.

Data on nonpoint loadings (flow × concentration) from major tributary streams or canals are available (e.g., Eau Gallie River and Crane Creek in segment IR9-11, Crane and Turkey creeks in IR12, Sebastian River in IR14-15, and the Vero canals in IR16-20). Those data do not account for the substantial loads conveyed by minor streams, outfalls, and diffuse runoff. Nonpoint load estimates, annual and seasonal, were generated using watershed loading models. Two models were evaluated for use in this study: the Pollutant Load Screening Model (PLSM; Adamus and Bergman 1995) and the Hydrological Simulation Program—Fortran (HSPF; Bicknell et al. 2001).

PLSM is a Geographic Information System (GIS)-based spreadsheet model that relies upon spatial data layers (e.g., land use, basin boundaries, average rainfall, soil hydrology) and static, empirically derived runoff coefficients and concentrations assigned to spatial data layers. HSPF has similar, spatial data requirements as PLSM (land use, basin boundaries, rainfall time series), but it is more complex than PLSM in that it provides continuous simulations of the watershed's physical and chemical processes, such as pollutant build-up, wash-off, in-stream transport, and decay over hourly time steps.

PLSM was calibrated against observed data (1990–2003) from several IRL watersheds that consist of the major land use types found throughout the IRBR basin (Green and Steward 2003; CDM, Inc. unpublished data). HSPF was hydrologically calibrated to multiple watersheds for the period 1994–2001 (Adkins et al. 2004), but only one watershed (Sebastian) was used to calibrate pollutant loads (Bergman et al. 2002). Even so, HSPF generated pollutant load estimates for all the IRBR watersheds that fell within the range of literature values for central and south Florida (Adkins et al. 2004). The accuracy of the PLSM and HSPF models are rated as fair to very good (Table 2) according to a model reliability index provided in the *Application Guide for HSPF* (Donigian et al. 1984).

PLSM and HSPF model outputs were produced for each segment's watershed for each of the four mapping years. PLSM and HSPF model runs for each of the seagrass mapping years incorporated contemporaneous rainfall and land use, with related soils hydrology, in the following manner: 1943: 1943 land use (LU) data were used with 1942–1943 rainfall data (gaged); 1996: 1995 LU, 1995–1996

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TABLE 2. The percent differences between PLSM and HSPF results and measured values in the IRBR, and assessment of accuracy relative to acceptable ranges of percent differences (shown in parentheses) according to Donigian et al. (1984). PLSM was evaluated for all major parameters shown in the table; HSPF could only be evaluated for runoff volume.

Parameter	PLSM	HSPF
Runoff volume	−20%, Fair (15–20%)	−13%, Good (10–15%)
TSS	1%, Very good (< 15%)	Insufficient data
Nutrients	−1% TN and 9% TP, Very good (< 20%)	Insufficient data

rainfall (gaged); 1999: 2000 LU, 1998–1999 rainfall (gaged); and 2001: 2000 LU, 2000–2001 rainfall (Doppler radar, central IRL only).

Two sets of modeled loads from PLSM and HSPF were used to create two separate sets of regression analyses. More specifically, for each segment, the nonpoint loading estimates from each model were added to the point source loadings, converted to watershed areal loading rates (kg ha^{-1} yr^{-1} or $season^{-1}$), and then log-transformed for use in the regressions. Although the results between PLSM and HSPF were comparable (Green and Steward 2003; Fig. 3), there were sufficient differences between the models' results so that one or the other yielded stronger regression statistics (r^2 and p values) for a given sublagoon or pollutant. Whichever type of regression, PLSM or HSPF, was statistically stronger was the one selected for determining the final annual or seasonal load limits.

Calculation of the Load Limits and Total Load Allocations (or TMDL)

The load limits and total load allocations (kg ha^{-1} yr^{-1} or $season^{-1}$, and kg yr^{-1} or $season^{-1}$, respectively) were derived from the regression models relative to a −10% (shoreward) departure from the seagrass depth-limit target. The −10% departure criterion, rather than the target itself, was used at the behest of FDEP, with concurrence by the USEPA. The criterion is predicated upon Florida's water transparency standard (Chapter 62–302.530, Florida Administrative Code), which states that the "depth of the compensation point for photosynthetic activity [as it relates to seagrass depth limits in this case] shall not be reduced by more than 10% as compared to the natural background value." Natural background is defined as "the condition of the waters [water transparency in this case] in the absence of man-induced alterations based on the best scientific information available. . ." (Chapter 62–302.200(14), p. 4 Florida Administrative Code).

To obtain load-limit values, the log-transformed loading rates must be back-transformed. By simply taking the inverse log of the y-axis loading rate, the result underestimates the true mean value derived from the regression equation. This happens because the distribution of back-transformed y values based upon x values is no longer normal (Helsel and Hirsch 1992). To obtain a more accurate estimate of the mean value, we used a nonparametric method described by Duan (1983) as "smearing." This method uses the mean of the regression residuals (expressed in their original units) as a multiplier applied to the regression line value. For any back-transformed relationship to x, the value of y is derived from the following equation:

$$\hat{y} = f^{-1}[b_o + b_1\log(x)] \times \frac{\Sigma f^{-1}(e_i)}{n}$$

in which, e_i = residual errors, f^{-1} = inverse function of the transformation, $b_o + b_1\log(x)$ = regression line equation, and n = sample size.

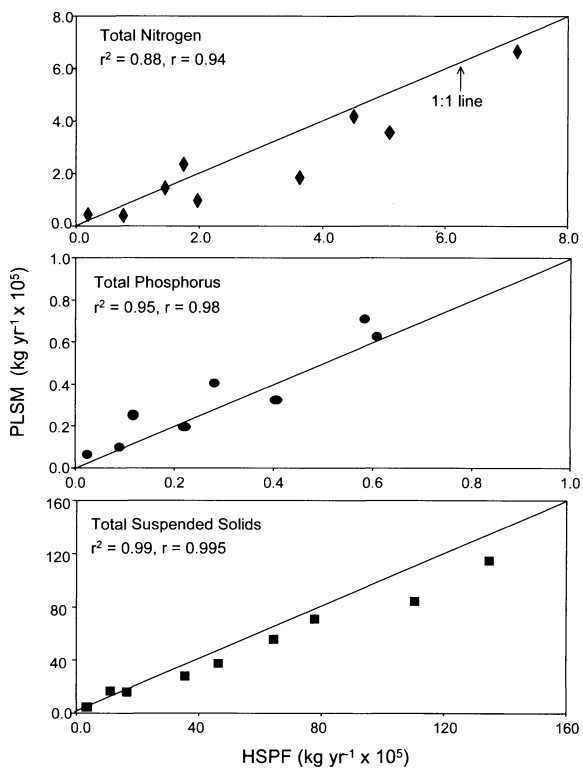

Fig. 3. PLSM versus HSPF regression plots with 1:1 lines indicating the close comparison between PLSM and HSPF annual loading results for all IRBR sublagoons and all applicable mapping years.

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Load limits were converted to sublagoon total load allocations or TMDL by multiplying the load limits by the sublagoon watershed area. An annual-based TMDL was distributed between a point-source wasteload allocation (WLA) and nonpoint load allocation (LA) by setting the WLA equal to the facilities' six-year mean annual load (see Table 1), and then subtracting that WLA from the TMDL to obtain the LA.

The sublagoon LA can be distributed among its segments according to each segment's relative percent contribution toward the sublagoon's current nonpoint load (estimated by either PLSM or HSPF using 2000 land use and 30-yr average annual rainfall). For any segment that contains any of the NPDES facilities listed in Table 1, those facilities' mean annual load can be added to its distributed LA to obtain a segment TMDL. The sequential steps in calculating a sublagoon TMDL and a segment TMDL (MOS notwithstanding) are arithmetically summarized below.

(1) $\text{Load limit}_{\text{sublagoon}} \times \text{Watershed area}_{\text{sublagoon}} = \text{TMDL}_{\text{sublagoon}}$

(2) $\text{TMDL}_{\text{sublagoon}} - \text{WLA}_{\text{sublagoon}} = \text{LA}_{\text{sublagoon}}$

(3) $\text{LA}_{\text{sublagoon}} \times (\text{current nonpoint load}_{\text{segment}} \div \text{current nonpoint load}_{\text{sublagoon}}) = \text{LA}_{\text{segment}}$

(4) $\text{LA}_{\text{segment}} + \text{WLA}_{\text{segment}} = \text{TMDL}_{\text{segment}}$

A TMDL margin of safety (MOS) must also be considered for the benefit of the resource pursuant to the Clean Water Act, 40 CFR 130.2. The MOS may be implicit if conservative model assumptions were used or certain characteristics of the water body have been altered, causing improvements in the targeted resource (e.g., an increase in flushing rates that helps improve water transparency and seagrass coverage). Alternatively, an explicit MOS was evaluated using the regression models; specifically, the lower limit of a specified confidence interval calculated about the regression line.

Results

Pollutant Load Limits (kg ha^{-1} yr^{-1} or season^{-1})

The regressions developed for each sublagoon reveal that a significant correlation exists between loading rate (TN, TP, and TSS) and the percent departure from the seagrass depth-limit target. Collectively, the regression models indicate that the range of watershed load limits (nonpoint runoff + point source) required to achieve, on average, −10% of the targeted seagrass depth limits are approximately 2.4–3.2 kg ha^{-1} yr^{-1} TN, 0.41–0.64 kg ha^{-1} yr^{-1} TP, and 48–64 kg ha^{-1} yr^{-1} TSS (Fig. 4). These load limits are substantially lower than current watershed loads (PLSM estimated: 2000 land cover, 30-yr mean annual rainfall) - 4.8–7.1 kg ha^{-1} yr^{-1} TN, 0.77–1.2 kg ha^{-1} yr^{-1} TP, and 105–197 kg ha^{-1} yr^{-1} TSS - but are still above predevelopment loads (PLSM: undeveloped watersheds, 1943 land cover, and mean annual rainfall) by 0–42% for TN, 13–26% for TP, and 82–176% for TSS.

Nearly all the annual regressions used PLSM loads because the PLSM produced a stronger set of regression statistics than did HSPF. The exception was the northern IRL TP regression, which improved using the HSPF loads. Some of the regressions were further improved by the removal of statistical outliers.

The central IRL regression statistics (r^2 and p) substantially improved when the 1943 data (all segments) and the Sebastian (IR14-15) segment data (all years) were excluded. The rationale for excluding these data sets is Sebastian Inlet's absence in the early 1940s and its presence in recent decades. Sebastian Inlet was not made a permanent opening until 1948 (Mehta et al. 1976). Before then, the inlet was often closed. Water residence times and the relationship between pollutant loads and seagrass distribution in the central IRL were likely to have been much different in 1943 than in recent decades. The present load-to-depth-limit relationship in segment IR14-15 (which includes Sebastian Inlet) is apparently much different than what it is in other segments. The data indicate that oceanic-induced flushing and dilution have sufficiently mitigated the effect of increased pollutant loadings from the Sebastian River watershed as seagrass coverage in IR14-15 has progressively expanded from 540 ha in 1943 to 1,320 ha in 2001.

Both the dry and wet season regression analyses yielded fairly strong correlations ($p \leq 0.017$; r_{TN} values of 0.58–0.90, r_{TP} values of 0.66–0.84, r_{TSS} values of 0.69–0.76). The exception was the central IRL dry season analyses, which showed such a poor correlation for all pollutants ($r \approx 0.3$) that dry season load limits for the central IRL were not considered. The seasonal load limits (kg ha^{-1} 3-mo-season^{-1}), as related to the −10% departure from the depth-limit target, are presented in Table 3.

Total Load Allocations or TMDL (kg yr^{-1} or season^{-1})

For developing sublagoon and segment TMDL, we performed the four sequential calculations as prescribed in the Methods. The results are presented in Table 4. An example of how sublagoon and segment TMDL were calculated is provided below using the annual TN load limit for the BRL sublagoon and segment BR3-5.

Multiplying the TN load limit established for the BRL sublagoon (2.44 kg ha^{-1} yr^{-1}; Fig. 4) by the sublagoon watershed area (20,797 ha) yields a sub-

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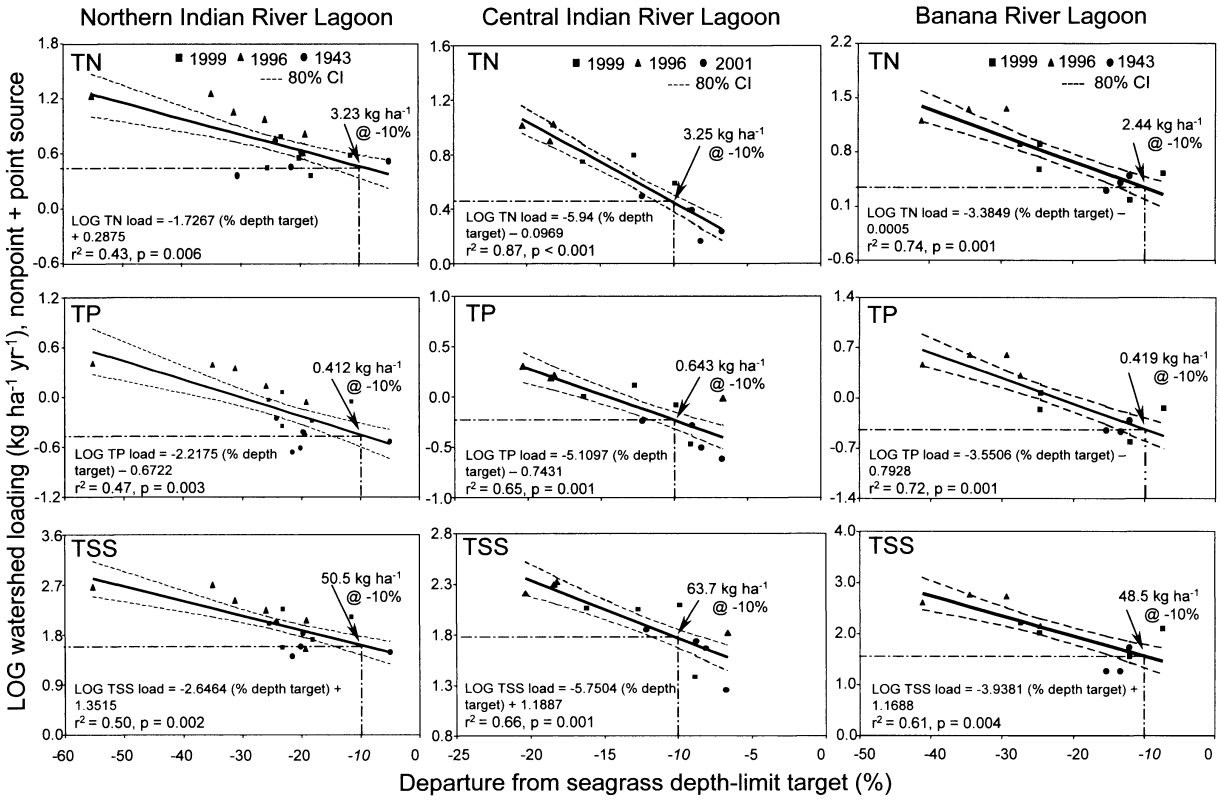

Fig. 4. Regression plots of the log-transformed loading rate (y axis) regressed on percent departure from depth-limit target (x axis) per sublagoon for TN, TP, and TSS. Also shown are each sublagoon's mean annual load limits for TN, TP, and TSS based on the −10% departure from the depth-limit targets, the 80% confidence intervals, and the regression equations and statistics. PLSM-estimated loads were used in all the regressions models except for northern IRL TP where HSPF-estimated loads were used because they provided stronger regression statistics. The 1943 data and segment IR14-15 data were excluded from the central IRL analysis for reasons explained in the Results.

lagoon load allocation of approximately 50,800 kg yr^{-1} TN. Subtracting the TN WLA for the BRL sublagoon (7,234 kg yr^{-1} as a combined allocation for the two NPDES facilities; see Table 1) from the TN total load allocation (50,800 kg yr^{-1}) provides the nonpoint LA: 43,570 kg yr^{-1}. Under current conditions (2000 LU), segment BR3-5's nonpoint TN loading is 25.5% of BRL's nonpoint loading of 131,143 kg yr^{-1} TN. Multiplying that percentage by the sublagoon LA (43,570 kg yr^{-1}) provides a LA of

TABLE 3. Mean seasonal load limits (kg ha^{-1} $season^{-1}$) and allocations (kg $season^{-1}$) per sublagoon. Seasonal limits and allocations include both nonpoint loads + point source loads. The HSPF load regressions were used to calculate wet season TP and TSS load limits in both the northern Indian River Lagoon (IRL) and Banana River Lagoon. The remaining load limit values were calculated using the PLSM load regressions.

	TN		TP		TSS	
	kg ha^{-1}	kg $season^{-1}$	kg ha^{-1}	kg $season^{-1}$	kg ha^{-1}	kg $season^{-1}$
Wet season (August–October)						
Northern IRL (54,790 ha)[a]	1.15	63,008	0.112	6,136	12.0	657,480
Central IRL (115,008 ha)	1.90	218,515	0.308	35,422	33.8	3,887,270
Banana River Lagoon (20,797 ha)	0.84	17,469	0.128	2,662	14.9	309,875
Dry season (February–April)[b]						
Northern IRL	0.358	19,615	0.064	3,507	6.50	356,135
Banana River Lagoon	0.157	3,265	0.027	562	3.47	72,166

[a] Watershed land area excluding the Lagoon and its tributaries.
[b] Dry season load limits and allocations for the central IRL were not considered because of poor correlation statistics.

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TABLE 4. Total load allocations (nonpoint + point source) by sublagoon and segment. A sublagoon's total load allocation was distributed among its segments according to the segment distribution of current (ca. 2000) sublagoon loads. An asterisk (*) indicates that the segment allocation includes the allocated point source load from Table 1.

IRBR Sublagoons and Segments	Total Load Allocations[a] (kg yr^{-1})		
	TN	TP	TSS
Northern IRL	176,861	22,599	2,769,601
IR1-3	39,850	2,626	317,296
IR4	6,124	802	119,660
IR5	37,159	4,813	578,404
IR6-7	37,558*	5,785*	512,382*
IR8	7,525*	1,510*	130,334*
IR9-11	48,645*	7,063*	1,111,525*
Central IRL	373,822	73,991	7,334,652
IR12	102,987*	19,355*	2,389,923*
IR13	12,704	1,833	142,490
IR14-15b	147,448*	28,706*	2,230,223*
IR16-20	108,190*	23,538*	2,484,940*
IR21	2,493	559	87,076
Banana River Lagoon	50,816	8,718	1,009,330
BR1-2	19,263	2,927	341,098
BR3-5	18,344*	3,344*	350,668*
BR6	6,966	1,377	190,741
BR7	6,243	1,070	126,823

[a] The total load allocations presented in this table can serve as the sublagoon or segment TMDL if the MOS is implicit.

[b] IR14-15 was excluded from the regression analyses because of its much greater flushing rate (due to its close proximity to Sebastian Inlet) as compared to other IRBR segments. This reason for exclusion could also be used to argue for some upward adjustment of the load allocation or TMDL for IR14-15. Other factors outside the scope of this study would need to be considered to determine what the level of adjustment could be.

11,110 kg yr^{-1} TN for segment BR3-5. Summing the segment LA (11,110 kg yr^{-1} TN) and the segment WLA (7,234 kg yr^{-1} TN; both facilities are in BR3-5) results in 18,344 kg yr^{-1} TN as a segment load allocation for BR3-5 (which can serve as a segment TMDL if the MOS is implicit; see Table 4).

The sequence of calculations above can also be applied to the seasonal load limits (Table 3) to determine seasonal load allocations or TMDL.

In order to achieve the total annual load allocations (Table 4), the percent reduction in current (ca. 2000) loads for the sublagoons would need to be approximately 34–63% for TN, 55–66% for TP, and 52–75% for TSS. The BRL would require the highest levels of load reduction because its load allocations are lowest and, save for BR1-2, its segment basins are intensively developed with high areal loading rates (e.g., 8.2 kg ha^{-1} yr^{-1} TN in BR6 and $>$ 10 kg ha^{-1} yr^{-1} TN in BR3-5 and BR7).

The TMDL Margin of Safety (MOS)

The results in Table 4 can serve as the annual TMDL if the MOS is implicit as defined in the Methods. A case could be made for an implicit MOS given that the sublagoon load limits generally approximate the estimated loads for 1943 (Steward and Green 2006). Sebastian Inlet was not open in 1943, but is a permanent channel today, which provides flushing that lessens the effect of pollutant loads.

An explicit MOS, as described in the Methods, can be the lower limit of a specified confidence interval (CI) about the regression line. The application of a CI lower limit assumes that the certainty of achieving a seagrass depth-limit target is increased beyond that expected from the regression line's load-limit value. For the sake of reasonableness, some constraint on a lower load limit needs to be decided. This approach provides a statistical context in making that decision. If a 66% CI is chosen, its lower limit becomes the new load target with a 66% probability that the true mean annual load target is not lower. The lower limits of two different CIs (66% and 80%) are presented in Table 5 and are compared with the mean annual load limits (kg ha^{-1} yr^{-1}) and total load allocations (kg yr^{-1}) calculated from the regression line.

Discussion

The annual load regressions indicate that TSS and nutrient loads from surface water point and nonpoint sources account for most of the variability in the depth distribution of seagrass in central IRL and Banana River, and up to 50% in the northern IRL. An important causal link implicit in these models is light attenuation, which sets the equilibrium depth limits of seagrass distribution and is apparently affected by watershed loadings of TN, TP, and TSS. It is important to note that, in addition to land use changes over time, the wet

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TABLE 5. TN, TP, and TSS load limit (kg ha^{-1} yr^{-1}) and load allocation (kg yr^{-1}) results from the regression confidence interval (CI) approach that can be used to define an explicit margin of safety (MOS) for TMDL development. For each parameter and sublagoon, the lower limit values relative to the 66% and 80% CI are presented in comparison to the mean values derived from the regression line.

	TN		TP		TSS	
	kg ha^{-1} yr^{-1}	kg yr^{-1}	kg ha^{-1} yr^{-1}	kg yr^{-1}	kg ha^{-1} yr^{-1}	kg yr^{-1}
Northern IRL						
Mean annual load limits and allocations (Fig. 4, Table 4)	3.23	176,861	0.412	22,599	50.5	2,769,601
66% Lower limit	2.32	127,113	0.277	15,177	31.0	1,698,490
80% Lower limit	2.16	118,346	0.255	13,971	28.2	1,545,078
Central IRL						
Mean annual load limits and allocations (Fig. 4, Table 4)	3.25	373,822	0.643	73,991	63.7	7,334,652
66% Lower limit	2.90	333,523	0.520	59,804	51.2	5,888,410
80% Lower limit	2.78	319,722	0.491	56,469	49.1	5,646,893
Banana River Lagoon						
Mean annual load limits and allocations (Fig. 4, Table 4)	2.44	50,816	0.419	8,718	48.5	1,009,330
66% Lower limit	1.76	36,603	0.289	6,010	26.2	544,881
80% Lower limit	1.62	33,691	0.265	5,511	23.2	482,490

and dry years provided the high and low watershed (nonpoint) loadings of nutrients and TSS represented in the data. Without such variability in loadings, meaningful regressions would not be possible. Although other pollutant factors (e.g., other loading sources, color, resuspended sediment) contribute to the variability in seagrass depth-limit coverage, it is evident that reductions in watershed loadings have a significant positive effect.

By adding the atmospheric and groundwater loads to the watershed load limits, we can establish total allochthonous load limits. For the IRBR overall, the total allochthonous load limits, normalized to estuary area, are 14 kg ha^{-1} yr^{-1} TN, 1.8 kg ha^{-1} yr^{-1} TP, and $<$ 190 kg ha^{-1} yr^{-1} TSS. Valiela and Cole (2002), Hauxwell et al. (2003), and Driscoll et al. (2003) present data on other estuaries that indicate a N load limit $<$ 12–20 kg ha^{-1} yr^{-1} would be necessary for seagrass recovery or maintenance. Valiela and Cole (2002) compiled reported data on worldwide seagrass loss over a range of N loads and found significant loss in seagrass coverage when N loads per estuary area exceed 20–30 kg ha^{-1} yr^{-1}. Hauxwell et al. (2003), reviewing the same data, observed that $>$ 50% loss of seagrass can occur when N loads are 50–100 kg ha^{-1} yr^{-1}. The IRBR system conforms to this apparent worldwide trend (Fig. 5). Similar types of data on TP or TSS from other estuaries were not found in the literature.

A relatively long water residence time (months) is an important reason why this regression method is generally applicable for much of the IRBR. Where residence time is very short (days), as in the Sebastian segment (IR14-15), there was no correlation between seagrass coverages and loads (the reason for excluding IR14-15 from the regression models). Phlips et al. (2004) observed that the phytoplankton (chl *a*) response in Sebastian was insensitive to relatively high N and P inputs from its watershed; for the IRBR overall, the key factors affecting change in phytoplankton standing stock are water residence times and watershed inputs. Water residence time probably explains some of the differences among sublagoon load limits (Fig. 4). Load limits are highest in the central IRL and lowest in the BRL where average R_{50} values (time it takes to flush 50% of a conservative tracer) are shortest (10 d) and longest (160 d), respectively (Christian unpublished data). The variability in the relationship between seagrass coverage and nutrient load rates among estuaries worldwide (Fig. 5) is partly a function of water residence time (Valiela and Cole 2002).

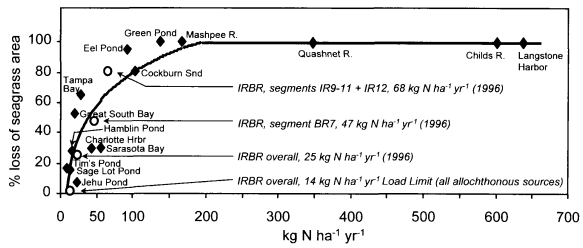

Fig. 5. Percent loss of seagrass area versus nitrogen loading, normalized to estuary area, for selected estuaries worldwide (♦; compiled by Valiela and Cole 2002) and for the IRBR (○; this study). The line of best fit corresponds only to the data compiled by Valiela and Cole (2002). The IRBR points consist of the estimated nitrogen load limit for the IRBR overall and the 1996 data for IRBR overall and for certain segments that exhibited the highest nitrogen loadings and worst seagrass coverage of the 1996–2001 period.

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Although we did not perform any analysis to estimate lag periods in the response to nutrient loads, the regression analyses indicate that growth-season seagrass coverage responds to long period loadings of up to one year. A similar temporal response between nutrient input and seagrass coverage or growth was observed for Tampa Bay (Janicki and Wade 1996), Chesapeake Bay (Magnien et al. 1992), and Patuxent Estuary (Kemp and Boynton 1984). The majority of the nutrient load entering the IRBR is organic and a considerable amount of that is particulate (SJRWMD unpublished data). The time required to break down and remineralize the organic fraction of the load is certainly within the response time for seagrass growth. We conclude that annual load limits are both reasonably sufficient and practical for meeting seagrass targets in the IRBR. The annual load regressions have a slightly higher p value than the seasonal regressions, and it is usually more practical to implement nutrient load management measures on an annual basis. Seasonal load allocations can be useful as design criteria for surface runoff treatment projects (with a likely emphasis on wet season criteria), which would help ensure that annual TMDL are met.

Correlation analysis and regression models have been used before to predict seagrass coverage or depth distribution, or set seagrass water quality criteria (Janicki and Wade 1996; Short and Burdick 1996; Tomasko et al. 2001; Kemp et al. 2004). Such empirical approaches may be most successful in bays and lagoons where residence times are moderately long and allochthonous pollutant loading is a major factor affecting light attenuation and seagrass coverage. These characteristics are certainly apparent in the IRBR system.

The general modeling paradigm used in Tampa Bay, Chesapeake Bay, and other estuaries takes advantage of a series of fundamental relationships: load → concentration (usually chl *a*) → light attenuation → seagrass light requirement → seagrass coverage.

For the IRBR, that paradigm has been functionally simplified whereby the three intermediate relationships are implicitly addressed by the direct empirical relationship between the first and last step: load → seagrass coverage (as a function of depth).

The back-prediction of load limits from seagrass depth limits, without the full numerical development of intermediate relationships, appears to be a logical alternative for the IRBR system given the weak correlation between loads and water quality (e.g., light attenuation, chl *a*). For many estuaries, the lack of correlation between loads and water quality is quite common because of the large variability typically inherent in water quality data. Tomasko et al. (1996), studying Sarasota Bay estuary, observed that seagrass was a better indicator of pollutant loads than traditional water quality parameters.

In shallow lagoons, nutrient inputs can take multiple assimilation pathways whereby the dominant responses are typically in the attached macroalgae, drift macroalgae, and epiphytes, while the phytoplankton (chl *a*) response is minimal or not detected (Fong et al. 1993; Harlin 1995; Nixon et al. 2001). All those different algal responses can affect seagrass coverage and may explain why chl *a* does not correlate well with either nutrient loads or seagrass coverage in the IRBR while nutrient loads do correlate with seagrass coverages. One purpose served by quantifying the intermediate relationships is that it helps determine water quality targets, a management requirement for many estuaries including the IRBR. To satisfy that requirement, one can also look to the regression method presented here, which suggests that a reference site-year method can be applied. That is, water quality data derived from certain segments in certain years when desired depth limits were met could be used to set targets for water transparency and other relevant parameters.

Application of this regression model approach to other estuaries would require simultaneous and covariant data sets of pollutant loadings and seagrass depth limits (or, alternatively, areal coverages), representing sufficiently large variability among years or water bodies. We assume that most seagrass management programs could readily develop the requisite seagrass coverage or depth-limit data sets, but would find it more difficult and costly to develop nonpoint pollutant load estimates over time or across water bodies. Cost and effort can be reduced by application of GIS-based watershed models, like the PLSM, that can generate reliable annual load estimates and are steadily improving in user performance and the availability of regional parameter data and pollutant export coefficients. HSPF or other continuous simulation models may be a better option if daily or monthly load estimates are desirable.

As previously mentioned, a load versus seagrass coverage regression would be most applicable to lagoons and bays with long residence times. Kurtz et al. (2006) advocate the use of stressor response paradigms in classifying types of estuaries (i.e., grouping estuaries with similar properties including biological response to nutrient loading) for the sake of simplifying the process of developing load limits or TMDL for 937 impaired estuaries in the U.S. Estuaries of a particular classification type can be subjected to similar data analysis for TMDL de-

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velopment rather than expending time and effort to develop a unique approach for each water body. Estuaries physically classified as lagoons and bays (Pritchard 1967; Dronkers 1988), including wave-dominated embayments and sloughs, can be further grouped by residence time (R_{90} = weeks to months). Such a group of estuaries could be considered as belonging to one stressor response type to which some variation of the load versus seagrass regression approach may be applied toward TMDL development. This approach may also have value outside the U.S. given that shallow lagoons and bays occupy more than 13% of the world's coastal zone by area (Kennish 1986; Alongi 1997).

The regression models provide reasonable estimations of the annual or seasonal average load limits of TN, TP, and TSS required to meet seagrass depth-limit targets. Surface water runoff is the major management focus for implementing load reductions given that runoff contributes the majority of the nutrient and TSS loadings to the IRBR. The SJRWMD and the National Estuary Program (USEPA), assisted by local jurisdictions, are planning to build regional surface water storage and treatment facilities (e.g., detention reservoirs, created wetland treatment areas) over the next 15 years to help meet segment-specific load targets. Reductions in watershed drainage volume and rates of discharge necessary to meet nutrient and TSS load targets will also serve to minimize the color effect on the seagrass light requirement. It is also important for wastewater facilities to maintain or reduce their current effluent loadings of N and P.

The regression approach presented here could be applicable to any estuary in which the variance in seagrass depth limits is explained best by loads rather than by water quality variables. The most likely type of estuaries where this approach could work are shallow lagoons or bays in which seagrass growth is limited by light and water transparency, water transparency is strongly affected by allochthonous pollutant loadings, water residence times are sufficiently long to allow seagrass coverage to respond to inorganic and organic allochthonous loads (enabling covariance between the independent and dependent parameters), and the data show sufficient variability in both pollutant loadings and seagrass coverages in order to develop a statistically meaningful regression. This last point underscores the value of long-term or multiyear data in the management of estuaries.

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