

General empirical models for estimating nutrient load limits for Florida's estuaries and inland waters

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Abstract

We developed empirical models that can predict maximum allowable, nutrient loading rates or load limits (L_M) for maintenance of mesotrophy in Florida's fresh and estuarine waters. Water residence time (τ_w) explains most of the variation in L_M previously set by the U.S. Environmental Protection Agency and Florida's environmental agencies for total phosphorus (TP) and for total nitrogen (TN). We found that a single model for TP and for TN fit data from both estuarine and freshwater systems. Further, we show that the models are consistent with or analogous to the critical loading model of the Organization of Economic Cooperation and Development (OECD 1982): $L_c = [\text{TP or TN}]_c \times z / \tau_w (1 + \tau_w^{0.5})$. The $L_M - \tau_w$ model for TP is sufficiently similar to the OECD model as to indicate that either model could provide a reasonable fit to mesotrophy in both temperate and subtropical systems. The models can facilitate the development of total maximum daily loads for estuaries and inland waters of Florida and may have broader regional utility. They may also aid the development of nutrient concentration criteria.

Florida, as well as the rest of the nation, is seriously challenged by the increasingly complex problems stemming from anthropogenic effects on its natural aquatic systems. Cultural eutrophication of freshwater systems and estuaries, as a consequence of nutrient enrichment, is one such effect that has heightened public concern, giving rise to many academic and governmental programs whose collective goal is to improve their trophic status and restore designated uses. Work over the last 20 yr and well into the foreseeable future is focused on applying a large and growing body of information on the relationships between nutrient levels and primary production or trophic status toward the development of nutrient criteria intended to reverse or prevent cultural eutrophication of Florida's waterbodies. This is the intent of the total maximum daily load (TMDL) and nutrient criteria programs managed by the U.S. Environmental Protection Agency (U.S. EPA) and the Florida Department of Environmental Protection (FDEP), and the Pollutant Load Reduction Goal (PLRG) program managed by Florida's regional water management districts (1972 federal Clean Water Act; Chapter 99-223, Laws of Florida; Chapters 62-302 and 62-40, Florida Administrative Code or F.A.C.).

The purpose of the TMDL program and its complementary PLRG program is to establish the maximum nutrient load that a waterbody can assimilate and meet Florida's narrative nutrient criterion (nutrient concentrations shall not "...be altered so as to cause an imbalance in natural populations of aquatic flora or fauna" [Chapter 62-302.530 {48} {b}, F.A.C.]), and other applicable criteria or response end points. Although there are differing response end points—dissolved oxygen (DO), chlorophyll *a* (Chl *a*), water transparency, etc.—with respect to setting nutrient loading limits in Florida, they all seek to prevent or reverse eutrophy and have converged on some common goals: limitation of macroalgal or phytoplankton production and promotion or

restoration of submersed rooted macrophytes. If a common trophic condition is the goal for nutrient load limits, then comparative analysis of a nutrient load-limit data set (e.g., TMDLs), including mediating factors such as hydrology, may yield relationships common within and even across system types. Nutrient load–trophic relationships are well documented in limnology and are being developed for estuaries (Boynton and Kemp 2000; Kelly 2008).

There has been substantial progress in characterizing lakes and reservoirs, as a group, through models relating critical nutrient concentrations or loads to transitional points along a trophic gradient (e.g., from mesotrophy to eutrophy, Vollenweider 1976). One type of trophic gradient that is widely used is the trophic state index (TSI) (Carlson 1977), which was modified for use in Florida lakes (Kratzer and Brezonik 1981; Huber et al. 1982). Models of Vollenweider (1975, 1976) were also modified for Florida lakes to help determine "permissible" or critical (lower to upper mesotrophic) and "dangerous" (eutrophic) P loads (Baker et al. 1985).

Early work (Schindler and Nighswander 1970; Lerman 1974; Dillon 1975) indicated that the effects of nutrient loading were strongly influenced by water residence time ($\tau_w = V/Q$, where V is mean lake volume and Q is mean water discharge rate). Vollenweider (1976) and the Organization of Economic Cooperation and Development (OECD 1982), in their studies of North American temperate, nordic, and alpine lakes and reservoirs, showed that water renewal rate ($1/\tau_w$) or τ_w significantly improved correlations between loading rates and primary production or trophic parameters. Baker et al. (1981, 1985) modified the critical loading models of Vollenweider (1975) and Dillon (1975) for Florida lakes by increasing the P concentrations for the upper and lower bounds of mesotrophy. This modification accounted for the naturally higher productivity of Florida's comparatively shallower and more highly colored subtropical lakes. Huber et al. (1982), in their major effort to classify Florida lakes, observed that the nutrient load did not predict water

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quality or trophic status unless it was combined with volume and τ_w , and then compared with a critical threshold (e.g., upper mesotrophy).

The significance of this lake research on nutrient loading and τ_w was not lost on those interested in estuaries. The importance of τ_w in an estuary's assimilative capacity, as in lakes, seems intuitive; but the magnitude of its significance became clearer over time. There are important differences between freshwater and estuarine systems, but their similarities are sufficient for successful application of limnological modeling concepts to estuaries (Boynton et al. 1996; Meeuwig et al. 2000; Kelly 2008). Biggs et al. (1989) utilized the Vollenweider approach to classify estuaries as to their sensitivity to nutrient loading and susceptibility to eutrophication. According to their classification scheme, an estuary's assimilative capacity is a function of its hydraulic loading ($q_s = z/\tau_w$, in which z is mean depth). Swaney et al. (2008) found that nutrient loading rates and τ_w were the most critical controlling factors determining nutrient concentrations and the steady-state response of an estuary. Related to that are findings that nutrient loss rates (burial and denitrification) in estuaries are largely driven by q_s (Nixon et al. 1996) or τ_w (Dettmann 2001).

These findings indicate that τ_w is a major driver of processes that regulate nutrient availability and trophic responses in both freshwater and estuarine systems. The potential significance of τ_w across aquatic ecosystems prompts the following questions: (1) Can variation in the established nutrient load limits (e.g., TMDLs or PLRGs) be largely explained by general empirical models using q_s or τ_w as independent variables? (2) If so, will the models differ for freshwater and estuarine systems? (3) Do the established load limits represent a common trophic state? (4) Do the models developed for Florida differ significantly from the τ_w -dependent critical loading (L_c) model of the OECD (1982)? The OECD model predicts the critical nutrient loading rate (L_c) from three parameters: the critical steady-state nutrient concentration ($[TP]_c$ or $[TN]_c$), z , and τ_w :

$$L_c(TP \text{ or } TN) = [TP]_c \text{ or } [TN]_c \times z / \tau_w (1 + \tau_w^{0.5}) \quad (1a)$$

Or, given that $z/\tau_w = q_s$, then

$$L_c(TP \text{ or } TN) = [TP]_c \text{ or } [TN]_c \times q_s (1 + \tau_w^{0.5}) \quad (1b)$$

To answer the questions above, we examined the relationships between load limits (TMDLs or PLRGs) and τ_w and q_s for Florida freshwaters and estuarine waters. We found that simple regression models using q_s or τ_w as independent variables explain a very large portion of the variation in load limits across systems. Separate models for freshwater and estuarine systems were not required. Further, the model for TP is so similar to the OECD model developed for temperate lakes as to suggest that one relationship holds for both temperate and subtropical estuaries and inland waters.

Methods

To address the four questions above, we reviewed the literature to extract essential load-limit data sets for Florida

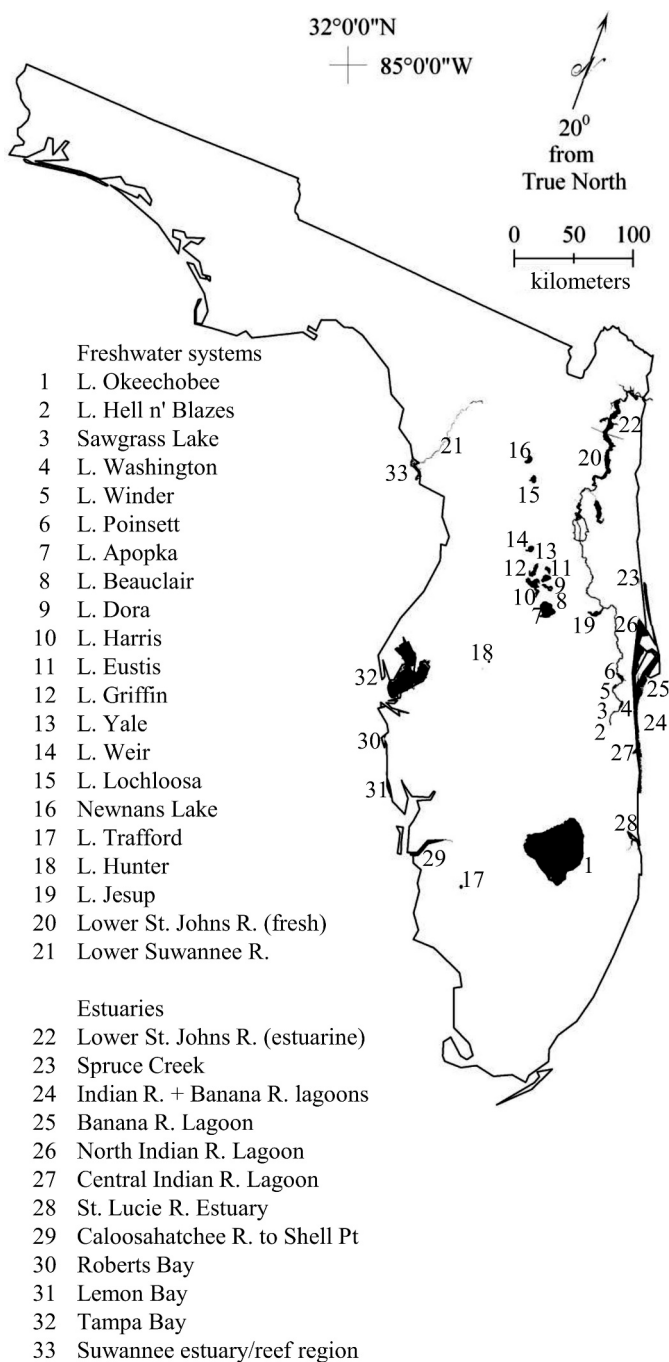


Fig. 1. List and map of Florida freshwater and estuarine systems evaluated in this study.

systems where TMDLs or PLRGs were established. From agency reports and journal articles, we obtained complete data sets on 33 systems located throughout much of the Florida peninsula (Fig. 1; Tables 1 and 2), comprising freshwater lakes and river segments, and estuarine systems (riverine, bay, lagoonal, and nearshore). The agency reports consisted of TMDL reports from FDEP and U.S. EPA and reports on allowable loads and PLRGs from Florida's water management districts. Although some of these reports were not published in peer-reviewed journals,

the TMDL and PLRG development process demands a rigorous scientific peer review. For many systems, not all essential data (e.g., τ_w , trophic status) were found in the literature; in which case we contacted the principal authors or investigators to obtain the data. Less than one-third of the TMDL or PLRG studies (listed in Table 1) reported τ_w or applied τ_w in the determination of load limits.

For lake and riverine systems, τ_w is defined as the average time for complete replacement of the system's original water volume. These values were available in the literature or obtained from investigators upon request (Table 2). For bay and lagoonal estuaries, where tidal and wind diffusion strongly affects transport, residence times were reported for varying fractions of complete water replacement. In an attempt to scale their residence times to match complete water replacement times, we adjusted the reported residence times for bays and lagoons to R_{99} values (time to flush 99% of the initial mass or water parcel) by applying a first-order exponential decay rate as described by Chapra (1997, p. 60–61) and Monsen et al. (2002). In the first-order model, the time required for any fraction of the water or mass to be replaced is given by the natural log of the fraction. We multiplied the reported times for differing fractions of water replacement by the appropriate factor to obtain R_{99} . For example, freshwater residence times ($e^{-1} = 0.37$, which translates to a R_{63}) were reported for Tampa Bay (Meyers and Luther 2008), Roberts Bay, Lemon Bay (Sheng and Peene 1992), and Suwannee River reef region (Bledsoe et al. 2004). Because R_{99} is based on a reduction of mass to $e^{-4.6}$ of its original value, a multiplier of 4.6 was applied to the R_{63} estimates to obtain R_{99} . For the Indian River and Banana River lagoon segments where R_{50} was reported (Sheng and Davis 2003), we multiplied the estimated times by 6.64 to obtain R_{99} .

For many systems, the current and target trophic status (Table 2) were not found in the literature, but were made available upon request of the system's principal investigators. We define trophic status according to two trophic state models: one for Florida lakes (Kratzer and Brezonik 1981) and one for warm-water, tropical lakes (Salas and Martino 1991).

The TMDL or PLRG load limits (hereinafter referred to as L_M) were developed independently for each waterbody by U.S. EPA, FDEP, or a Florida water management district by using either empirical or mechanistic models that quantified the requisite load–response relationships. Most of the methods consisted of linked, empirical regression models (Table 1). Target concentrations of nutrients, critical in setting the L_M for most systems, were determined by correspondence analysis, reference-site comparisons, mass-balance nutrient models, or by a weight of evidence using any combination of those methods (Table 1). A potential source of spurious correlation lies in estimates of L_M on the basis of τ_w . For most systems (23 of 33), however, L_M was developed with no accounting for τ_w ; thus independence between the parameters in the empirical $L_M - \tau_w$ models was largely satisfied.

All methods accounted for watershed point and non-point surface-water sources, groundwater contributions, and atmospheric deposition. Oceanic loading to estuaries

was not reported, except in the case of the Lower St. Johns River where offshore loads of organic seston were considered important in that system's DO dynamics (Sucusy and Hendrickson 2003). Overall, for Florida's microtidal estuaries, there is large uncertainty in quantifying the fraction of oceanic nutrient loading that is “new,” which is considered to be quite small and inconsequential on the basis of offshore concentration data (St. Johns River Water Management District unpubl.; Hoyer et al. 2002) and a perception that tidal delivery of loads to Florida estuaries is weak relative to freshwater delivery as described by Kelly (2001). That perception is supported by the fact that variation in primary production in the estuaries of this study is strongly regulated by watershed loadings.

We addressed the first two questions posed above by regressing the L_M values for TP ($L_M[P]$) and TN ($L_M[N]$) separately against the independent variables q_s and τ_w .

We addressed the third and fourth questions by comparing L_M values and the log-linear regression equations to the theoretical upper and lower bounds of mesotrophy and to the L_c predictions of the OECD equation (Eq. 1a or 1b).

We applied the OECD equation to predict L_c by using the target nutrient concentrations set by the agencies ($[TP]_M$ and $[TN]_M$ in Table 2) as the critical concentration values. We regressed L_c against τ_w and compared that $L_c - \tau_w$ relationship to our $L_M - \tau_w$ model by examining the plotted positions of their lines of best fit. Finally, those lines were compared with lines representing the upper and lower limits of mesotrophy in freshwater and estuarine systems. The bounds of mesotrophy were derived by using TP and TN concentration thresholds, determined previously to be associated with the upper and lower limits of mesotrophy for lakes in Florida and for estuaries in general (Baker et al. 1985; Smith 1998), as critical concentrations in the OECD equation.

Results

We obtained data on freshwater and estuarine systems that are widely distributed in Florida (Fig. 1) and exhibit a wide range of characteristics (Table 2). The smallest system is Lake Hunter (376,776 m²), and the largest is Lake Okeechobee, Florida's largest lake (1.8 billion m²). Average annual water color ranges from 5 platinum cobalt units (PCU) (Lemon Bay) to 200 PCU (Upper St. Johns River lakes), and τ_w ranges from about 1 d (or 0.0027 yr in the Lower Suwannee River) to nearly 17 yr (Lake Weir). However, there is one physical characteristic common among all these systems—shallow water depth. Mean depths (z) range from 1.2 m (Newnans Lake) to 5.2 m (Lake Weir), and the median z of all systems is 2.0 m.

$L_M - q_s$ models— $L_M[P]$ and $L_M[N]$ varied log-linearly with q_s ($R^2 = 0.97$ and 0.93 , respectively) and for all systems combined (Figs. 2, 3). These regressions indicate that q_s is a good predictor of L_M for both TP and TN. However, among these shallow systems, the following results will show that the τ_w component of q_s largely influences the load limit.

Table 1. Sources of data evaluated for this study and a summary of methods used to determine nutrient TMDL or PLRG load limits (L_M). Data sources are a mix of unpublished agency reports and published papers. References in this table are fully cited in a Web Appendix at http://www.aslo.org/lo/toc/vol_55/issue_1/0433a.pdf.

Waterbody	Source of data	Method of L_M determination
1. Lake Okeechobee	Havens and James 1997; Havens and Schelske 2001; FDEP 2001	WOE* set $[TP]_{M\ddagger}$, modified Vollenweider model (Kratzer 1979)
2. Lake Hell n' Blazes	Keenan et al. 2002; Gao 2006a	WOE set $[TP]_M$, Vollenwieder and Kerekes (1980) model; DO-TP regression
3. Lake Sawgrass	Keenan et al. 2002; Gao 2006a	WOE set $[TP]_M$, Vollenwieder and Kerekes (1980) model; DO-TP regression
4. Lake Washington	Keenan et al. 2002; Gao 2006a	WOE set $[TP]_M$, Vollenwieder and Kerekes (1980) model; DO-TP regression
5. Lake Winder	Keenan et al. 2002; Gao 2006a	WOE set $[TP]_M$, Vollenwieder and Kerekes (1980) model; DO-TP regression
6. Lake Poinsett	Keenan et al. 2002; Gao 2006a	WOE set $[TP]_M$, Vollenwieder and Kerekes (1980) model; DO-TP regression
7. Lake Apopka	Coveney et al. 2005; Lowe et al. 1999	WOE set $[TP]_M$, Vollenweider (1969) model
8. Lake Beauclair	Fulton and Smith 2008	WOE set $[TP]_M$, adjusted to 90% background transparency; [P] proportional to P load
9. Lake Dora	Fulton and Smith 2008	WOE set $[TP]_M$, adjusted to 90% background transparency; [P] proportional to P load
10. Lake Harris	Fulton and Smith 2008	WOE set $[TP]_M$, adjusted to 90% background transparency; [P] proportional to P load
11. Lake Eustis	Fulton and Smith 2008	WOE set $[TP]_M$, adjusted to 90% background transparency; [P] proportional to P load
12. Lake Griffin	Fulton and Smith 2008	WOE set $[TP]_M$, adjusted to 90% background transparency; [P] proportional to P load
13. Lake Yale	Fulton and Smith 2008	WOE set $[TP]_M$, adjusted to 90% background transparency; [P] proportional to P load
14. Lake Weir	Fulton and Smith 2008	WOE set $[TP]_M$, adjusted to 90% background transparency; [P] proportional to P load
15. Lake Lochloosa	Di and Marzolf 2008 $\ddagger$	Reference lake [P] and TSI-[P] regression; 1999 Walker model
16. Newnans Lake	Di et al. 2009	Reference lake [P] and TSI-[P] regression; 1999 Walker model
17. Lake Trafford	Kang and Gilbert 2008; Huber et al. 1982	HSPF model run iteratively with different reductions of current load
18. Lake Hunter	Baniukiewicz and Gilbert 2004	1996 Walker model Chl <i>a</i> prediction, correspondence with TSI and loads
19. Lake Jesup	Gao 2006b	WOE set $[TN, TP]_M$, Chl <i>a</i> , TSI targets; 1999 Walker model
20. Lower St. Johns River (fresh)	Hendrickson et al. 2003; Magley and Joyner 2004; U.S. EPA 2008	Three-dimensional (3-D) mechanistic model: CE-QUAL-ICM (Sucsy and Hendrickson 2003)
21. Lower Suwannee River	Hallas and Magley 2008; R. Mattson pers. comm. 22 Jan 2009	WOE set $[NO_3]_M$; $L_M(N)$ derived from NO_3 target and flow
22. Lower St. Johns River (estuarine)	Hendrickson et al. 2003; Magley and Joyner 2004; U.S. EPA 2008	3-D mechanistic model: CE-QUAL-ICM (Sucsy and Hendrickson 2003)
23. Spruce Creek	Magley 2008; F. Marshall pers. comm. 12 Nov 2008	DO-[TP] correlation and correspondence with P load time series
24. Indian River+Banana River lagoons	Steward and Green 2007; Gao 2009	Load-seagrass depth limit regression; $L_M(P,N)$ @ 90% background transparency
25. Banana River lagoon	Steward and Green 2007; Gao 2009	Load-seagrass depth limit regression; $L_M(P,N)$ @ 90% background transparency
26. North Indian River lagoon	Steward and Green 2007; Gao 2009	Load-seagrass depth limit regression; $L_M(P,N)$ @ 90% background transparency
27. Central Indian River lagoon	Steward and Green 2007; Gao 2009	Load-seagrass depth limit regression; $L_M(P,N)$ @ 90% background transparency
28. St. Lucie River estuary	Chamberlain and Hayward 1996; Doering 1996; Parmer et al. 2008	Correspondence analysis: [N, P] and loads, Chl <i>a</i> , light, color, target flows
29. Caloosahatchee River to Shell Point	Doering and Chamberlain 1999; Chamberlain 2007	WOE set $[TN]_M$, then flow $\times$ concentration
30. Roberts Bay	Tomasko et al. 2001; U.S. EPA 2005; Petrus and Lassiter 2005	Correspondence analysis: 1992 to 1996 load when annual Chl <i>a</i> averaged $4.8 \mu g L^{-1}$

Table 1. Continued.

Waterbody	Source of data	Method of L_M determination
31. Lemon Bay	Tomasko et al. 2001; D. Tomasko pers. comm. 17 Feb 2009	Sequential analyses (regressions): TN load, Chl <i>a</i> , light, seagrass
32. Tampa Bay	Greening and Janicki 2006; Meyers and Luther 2008	Sequential analyses (regressions): TN load, Chl <i>a</i> , light, seagrass
33. Suwannee estuary reef region	Bledsoe et al. 2004; Hallas and Magley 2008; Steward 2008§	WOE set [TN] _M ; TN load vs. Chl <i>a</i> regression

* WOE, weight of evidence.

† [$]_M$, target concentration.

‡ Information for Lochloosa Lake TMDL development: Setting achievement targets. 02 April 2008 draft. St. Johns River Water Management District. Palatka, FL.

§ TN load limit for the Suwannee River estuary or reef region. Draft white paper sent by E-mail, 04 Nov 2008, to Florida Department of Environmental Protection and to U.S. EPA.

$L_M - \tau_w$ models and comparison to the OECD model— $L_M(P)$ and $L_M(N)$ were inversely correlated with τ_w for all systems combined (Figs. 4, 5). For both $L_M(P)$ and $L_M(N)$, the determinant coefficients were only slightly lower for the models using τ_w ($R^2 = 0.94$ and 0.92) than for those using q_s . The strength of the $L_M - \tau_w$ models, which do not explicitly include z , indicates that the depth differences (1.2–5.2 m) among these Florida systems is sufficiently small to have little effect on differences in predicted load limits among the systems.

The correlation between $L_c(P)$ values and empirical $L_M(P)$ values is strong and close to unity in slope ($R^2 = 0.96$, $y = 0.97x - 0.128$). The log-linear $L_c(P) - \tau_w$ equation is quite similar to the log-linear $L_M(P) - \tau_w$ equation (Fig. 6). The striking similarity between the two equations and the strong correlation between L_c and L_M values indicate that either the $L_M(P) - \tau_w$ model or the OECD model could be used to predict TP load limits for Florida systems and, more generally, for both subtropical and temperate systems.

Even though a correlation exists between the $L_c(N)$ and $L_M(N)$ values ($R^2 = 0.87$ after the extreme point for the lower Suwannee River segments is removed, $y = 0.52x + 2.14$), the OECD model overestimates TN load limits as indicated by the low slope value (0.52). Overestimation by the OECD model is also shown by the plotted position of the log-linear $L_c(N) - \tau_w$ equation relative to the $L_M(N) - \tau_w$ model equation (Fig. 7). This overestimation indicates that the trophic state of N-limited Florida systems is more responsive to nitrogen than were the systems used to develop the OECD model. The $L_M(N) - \tau_w$ model appears to be more appropriate than the OECD model for determining TN load limits in Florida systems.

Evaluation of system classifications for model improvement—The data points in the L_M vs. τ_w or q_s plots (Figs. 2–5) are generally well distributed throughout the plots with respect to the two types of system classes: estuarine vs. freshwater systems and low-color vs. high-color systems. It should be noted that in the $L_M(P)$ plots (Figs. 2, 4), the most extreme points of high q_s or low τ_w belong to systems of high color (187 to 200 PCU), but their tight fit with the model favors hydrology, not color, as the primary determinant of load limits. The mixed distribution of

points combined with their good fit to the models suggests that the models can be used across those system classes. Development of separate intraclass models does not appear necessary, although data on more systems may prove otherwise.

L_M represents mesotrophy for Florida systems—The L_c determined by the OECD equation is intended to set an upper mesotrophic threshold for nutrient loading to lakes, above which lakes would tend toward eutrophy. This threshold is derived from nutrient concentrations at the upper limit of mesotrophy. Baker et al. (1981, 1985), analyzing data from 101 Florida lakes, set the upper and lower mesotrophic limits (UML and LML) for those subtropical systems above those set by Vollenweider (1976) or OECD (1982) for the less productive, temperate lakes. Using the Baker bounds (0.05 mg L⁻¹ TP and 1.0 mg L⁻¹ TN for the UML, and 0.025 mg L⁻¹ TP and 0.5 mg L⁻¹ TN for LML) in Eq. 1a or 1b, along with z and τ_w data in Table 2, we constructed log-linear $L_c - \tau_w$ lines of best fit for the UML and LML (Figs. 8, 9). In relation to those lines, the empirical L_M and the theoretical L_c for the Florida systems (from Fig. 6, 7) can be positioned within the trophic continuum (Figs. 8, 9).

Both the $L_M(P)$ and $L_c(P)$ lines for Florida systems lie on or very close to the UML line at intermediate τ_w , above the UML at low τ_w , and below at high τ_w (Fig. 8), indicating that Florida's TP TMDLs and PLRGs are being set generally within the limits of mesotrophy. It is interesting to note that for systems with $\tau_w < 3.5$ months, the TP TMDLs or PLRGs may yield conditions that are at best mesoeutrophic. However, there is a possibility that the Baker-based mesotrophic limits do not adequately address systems with short residence times given that only 18 of the 101 lakes evaluated by Baker et al. (1981, 1985) had $\tau_w < 3.5$ months (Huber et al. 1982).

The greater divergence between the $L_M(N)$ and $L_c(N)$ lines causes them to be nearly superimposed on the LML and UML for TN, respectively (Fig. 9). That suggests that Florida's TN TMDLs and PLRGs correspond to a lower mesotrophic condition tending toward oligotrophy. But, that may not be the case given the fact that the $L_M(N)$ line is heavily influenced by estuaries (11 of the 18 systems used in the $L_M[N]$ analysis are estuaries).

Table 2. Empirical data* from the source studies listed in Table 1: area (*A*), mean depth (*z*), color, average time for nearly complete replacement of water volume (τ_w), current and target trophic status (TS),† TMDL or PLRG TP load limit ($L_M(P)$) and concentration target ($[TP]_M$), TMDL or PLRG TN load limit ($L_M(N)$) and concentration target ($[TN]_M$), and the response end point on the basis of load limit.

Waterbody	<i>A</i> (m ²)	<i>z</i> (m)	Color (PCU)	τ_w (yr)	Current TS	Target TS	$L_M(P)$ (g m ⁻² yr ⁻¹)	$[TP]_M$ (mg L ⁻¹)	$L_M(N)$ (g m ⁻² yr ⁻¹)	$[TN]_M$ (mg L ⁻¹)	Targeted or predicted response end point
1. Lake Okeechobee	1,800,000,000	3	40	3.60	E	M	0.08	0.040	n.a.r.‡	n.a.r.	>Macrophytes; diatoms>cyanobacteria
2. Lake Hell n' Blazes	959,139	1.76	200	0.0045§	ME	M	35.3	0.090	n.a.r.	NO _x <0.1	<Freq. of blue-green algae increases; DO state std.
3. Lake Sawgrass	1,991,124	1.73	200	0.0074§	ME	M	27.4	0.090	n.a.r.	NO _x <0.1	<Freq. of blue-green algae increases; DO state std.
4. Lake Washington	10,408,884	2.33	200	0.0531§	OM	OM	5.4	0.090	n.a.r.	NO _x <0.1	<Freq. of blue-green algae increases; DO state std.
5. Lake Winder	6,171,675	1.32	200	0.0103§	ME	M	12.2	0.090	n.a.r.	NO _x <0.1	<Freq. of blue-green algae increases; DO state std.
6. Lake Pointsett	15,920,898	1.61	200	0.0263§	ME	M	4.87	0.090	n.a.r.	NO _x <0.1	<Freq. of blue-green algae increases; DO state std.
7. Lake Apopka	125,003,736	1.6	25	2.50	E	M	0.13	0.055	n.a.r.	n.a.r.	>Transparency and rooted SAV; 29 µg L ⁻¹ Chl <i>a</i>
8. Lake Beauclair	4,300,000	1.95	56	0.12	E	OM	0.74	0.032	n.a.r.	n.a.r.	90% background transparency; 30 µg L ⁻¹ Chl <i>a</i>
9. Lake Dora	17,600,000	2.89	40	0.65	E	OM	0.36	0.031	n.a.r.	n.a.r.	90% background transparency; 29 µg L ⁻¹ Chl <i>a</i>
10. Lake Harris	75,100,000	3.51	31	2.89	ME	OM	0.12	0.026	n.a.r.	n.a.r.	90% background transparency; 22 µg L ⁻¹ Chl <i>a</i>
11. Lake Eustis	31,100,000	3.3	31	0.60	ME	OM	0.34	0.026	n.a.r.	n.a.r.	90% background transparency; 24 µg L ⁻¹ Chl <i>a</i>
12. Lake Griffin	35,800,000	2.3	38	0.49	E	OM	0.24	0.032	n.a.r.	n.a.r.	90% background transparency; 29 µg L ⁻¹ Chl <i>a</i>
13. Lake Yale	19,600,000	3.27	20	10.36	ME	OM	0.07	0.020	n.a.r.	n.a.r.	90% background transparency; 14 µg L ⁻¹ Chl <i>a</i>
14. Lake Weir	21,900,000	5.22	16	16.85	M	OM	0.06	0.014	n.a.r.	n.a.r.	90% background transparency; 8 µg L ⁻¹ Chl <i>a</i>
15. Lake Lochloosa	22,663,200	2.3	110	2.30§	E	M	0.15	0.051	1.0	1.40	TSI 60; 21 µg L ⁻¹ Chl <i>a</i>
16. Newnans Lake	25,293,750	1.2	176	0.68§	E	M	0.13	0.068	1.59	1.125	TSI 60; 40 µg L ⁻¹ Chl <i>a</i>
17. Lake Trafford	6,159,316	2.1	121	0.71	E	M	0.25	0.025	4.2	1.09	TSI 56; 19 µg L ⁻¹ Chl <i>a</i>
18. Lake Hunter	376,776	1.8	25	0.26§	E	M	0.59	0.045	7.9	0.52	TSI 63
19. Lake Jesup	45,000,000	1.27	83	0.25	E	M	0.44	0.094	5.73	1.32	TSI 65; 30.5 µg L ⁻¹ Chl <i>a</i>
20. Lower St. Johns River (fresh)	180,000,000	2.9	156	0.10§	E	ME	2.78	0.062	47.6	1.21	Algal blooms >40 µg L ⁻¹ @≤40 d
21. Lower Suwannee River	16,940,000	4.5	130	0.0027§	ME	M	n.a.r.	n.a.r.	413	0.98	<Freq. macroalgal blooms
22. Lower St. Johns River (estuarine)	219,700,000	3.8	88	0.11§	M	M	n.a.r.	n.a.r.	6.7	0.90	Site-specific DO state criteria
23. Spruce Creek	516,600	1.45	187	0.025§	M	M	6.17	0.16	n.a.r.	n.a.r.	DO state std.; <11 µg L ⁻¹ Chl <i>a</i>
24. Indian River, Banana River lagoons	619,450,000	1.7	20	1.45	M	OM	0.18	0.055	1.4	<0.9	90% background seagrass depth limit; <7 µg L ⁻¹ Chl <i>a</i>
25. Banana River lagoon	185,630,000	1.7	20	2.83	M	OM	0.06	0.055	0.7	<0.9	90% background seagrass depth limit; <5 µg L ⁻¹ Chl <i>a</i>
26. North Indian River lagoon	294,370,000	1.9	20	1.97	M	OM	0.10	0.055	1.0	<0.9	90% background seagrass depth limit; <7 µg L ⁻¹ Chl <i>a</i>

Table 2. Continued.

Waterbody	A (m ²)	z (m)	Color (PCU)	τ_w (yr)	Current TS	Target TS	$L_M(P)$ (g m ⁻² yr ⁻¹)	$[TP]_M$ (mg L ⁻¹)	$L_M(N)$ (g m ⁻² yr ⁻¹)	$[TN]_M$ (mg L ⁻¹)	Targeted or predicted response end point
27. Central Indian River lagoon	139,450,000	1.7	25	0.22	ME	M	0.60	0.055	3.6	<0.9	90% background seagrass depth limit; <5 μ g L ⁻¹ Chl <i>a</i>
28. St. Lucie River estuary	29,000,000	2.4	73	0.075§	ME	M	2.15	0.081	19	0.72	Chl <i>a</i> <10 μ g L ⁻¹ ; >SAV coverage
29. Caloosahatchee River to Shell Point	64,752,000	1.5	58	0.05§	M	M	n.a.r.	~0.09	35	0.7–0.9	>SAV; 4.2 μ g L ⁻¹ Chl <i>a</i> in lower bay
30. Roberts Bay	11,736,300	2.2	<20	0.06	ME	M	n.a.r.	n.a.r.	26	n.a.r.	4.4 μ g L ⁻¹ Chl <i>a</i> >50% time or <7.2 μ g L ⁻¹
31. Lemon Bay	31,000,000	2	5	0.30	ME	M	n.a.r.	n.a.r.	2.6	n.a.r.	Seagrass coverage to c. 1950s level
32. Tampa Bay	954,000,000	4	<15	1.28	ME	M	n.a.r.	n.a.r.	2.9	n.a.r.	Seagrass coverage to c. 1950s level
33. Suwannee River estuary reef	70,000,000	2	85	0.01	ME	M	n.a.r.	n.a.r.	90	0.35 NO ₃	<Freq. macroalgal blooms; <9 μ g L ⁻¹ Chl <i>a</i>

* Data are reported or calculated on the basis of values as reported in the literature; thus, not all data for certain variables will have equivalent number of significant figures.

† Indicated as eutrophic (E), mesotrophic (M), oligotrophic (O), mesoeutrophic (ME), or oligomesotrophic (OM). Information obtained from principal investigator.

‡ n.a.r., not available or required by the specific case study.

§ Water residence time data obtained from principal investigator (PT) or author via E-mail (see Table 1 for name of PT or primary author).

|| Mesotrophy at its upper limit (D. Tomasko pers. comm., 17 Feb 2009; H. Greening, Tampa Bay National Estuary Program pers. comm., 06 Jun 2008).

Estuarine systems generally should have lower critical N concentrations compared with freshwater systems (Smith 1998). In the absence of P limitation, estuaries cannot be expected to maintain their mesotrophic conditions if TN concentrations greatly exceed ~ 0.4 mg L⁻¹, or for oligotrophic conditions, 0.26–0.30 mg L⁻¹ (table 2.3 in Smith 1998). These levels seem rather low for Florida estuaries considering the TN concentration targets set for them ($[TN]_M$ in Table 2); nonetheless, some downward adjustment in the Baker UML and LML may be appropriate for estuaries.

Although we have not conducted the comprehensive analysis required to establish an estuary UML and LML for TN, we provide a reasonable first approximation on the basis of TN:TP concentration ratios. The TN:TP ratio by weight of the Baker mesotrophic thresholds is 20 (e.g., UML 1.0 mg L⁻¹ TN:0.05 mg L⁻¹ TP = 20), and Florida lakes with ratios > 20 were considered P-limited by Baker et al. (1981 and 1985), whereas Huber et al. (1982) considered $10 \leq \text{TN:TP} \leq 30$ a nutrient-balanced lake in Florida. The TN:TP ratio by weight for a nutrient-balanced marine system is the Redfield ratio of 7.2 (Redfield et al. 1963; Hecky et al. 1993). Interestingly, 7.9 is the ratio of the coefficients in the untransformed $L_M(N)$ and $L_M(P)$ power functions (see respective L_M power functions in Figs. 5, 4), in which the $[TN]_c$ or $[TP]_c$ are implicit. At present, with no evidence to the contrary, we assume that the Baker TP concentration levels for the UML (0.05 mg L⁻¹) and LML (0.025 mg L⁻¹) are applicable to Florida estuaries. However, we believe that there should be a downward adjustment in TN concentration levels and we did so by using the Redfield ratio of 7.2. This adjustment reduced the TN boundaries of mesotrophy to 0.4 mg L⁻¹ for the UML and to 0.2 mg L⁻¹ for the LML. We used these boundaries to develop new estuary UML and LML lines for TN (Fig. 10). With this adjustment, the $L_M(N)$ equation for estuaries only (after removing the seven freshwater systems from the TN data set) is practically superimposed on the estuary UML, the two lines being nearly identical (Fig. 10). Also, because of the downward adjustment in the mesotrophic TN concentrations for estuaries, the ratio of the coefficients in the $L_M(N)$ and $L_M(P)$ power functions (see respective equations in Figs. 10, 4) is 7.3, nearly equivalent to the Redfield ratio. Regardless of which set of mesotrophic boundaries is used, the Baker or estuary UML and LML, Florida has set load limits for TN, as for TP, that sit generally within the limits of mesotrophy.

As a second line of evidence, survey responses from primary investigators generally point to mesotrophy as the target trophic status for all systems (see Target TS in Table 2).

Discussion

We were surprised to find that τ_w could explain so much of the variation in L_M for such a high diversity of aquatic systems with varying response end points. It appears that a simple model, on the basis of τ_w (or q_s), can predict nutrient load limits associated with mesotrophy in fresh and estuarine waters despite their differences in biology, hydrology, and water chemistry. Moreover, the consistency of the empirical $L_M(P) - \tau_w$ model with the OECD model

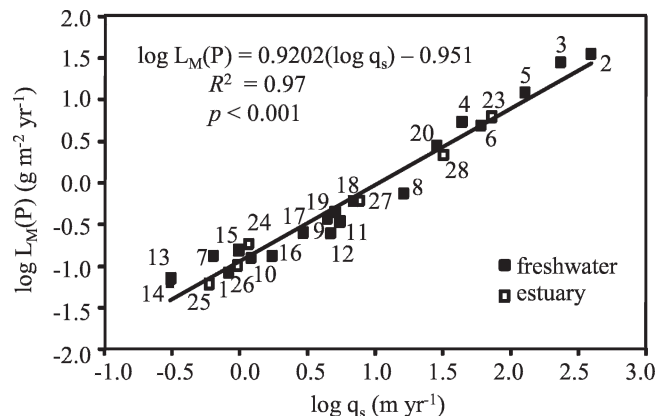


Fig. 2. Agency-established TP load limits, $L_M(P)$, regressed against hydraulic loading, q_s , for all systems using log-transformed data. Points are labeled with waterbody identification numbers as shown in Fig. 1 and the tables.

indicates that either one could be used to predict TP L_M , which would help maintain mesotrophy in subtropical systems (freshwater and estuarine) and temperate freshwater systems. The application of either model to predict TP L_M in temperate estuaries remains to be tested.

The surprising strength and generality of the models reasonably raises concerns about self-correlation. Spurious self-correlation occurs when a variable is plotted or regressed against a function of itself (Peters 1991). Areal load is not a simple manipulation of τ_w and can be determined with complete independence of τ_w , so the $L_M - \tau_w$ models are not self-correlations. It is clear, however, that the forms of the functions are largely mathematical. Although the L_M values we used were in most cases developed without reference to τ_w , trophic state is related to the areal loading rate through the steady-state nutrient concentration. For example, in a simple, mass-balance model of phosphorus, the areal load is a function of four variables: the steady-state TP concentration, z , the sedimentation coefficient, and $1/\tau_w$ (Reckhow and Chapra 1983). Of these variables, τ_w shows the widest variation. In

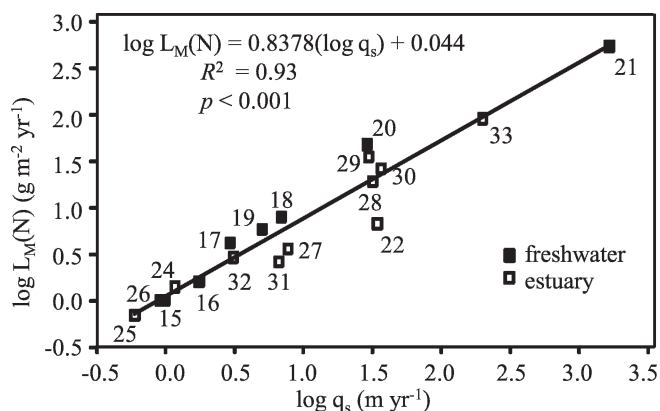


Fig. 3. Agency-established TN load limits, $L_M(N)$, regressed against hydraulic loading, q_s , for all systems using log-transformed data. Points are labeled with waterbody identification numbers as shown in Fig. 1 and the tables.

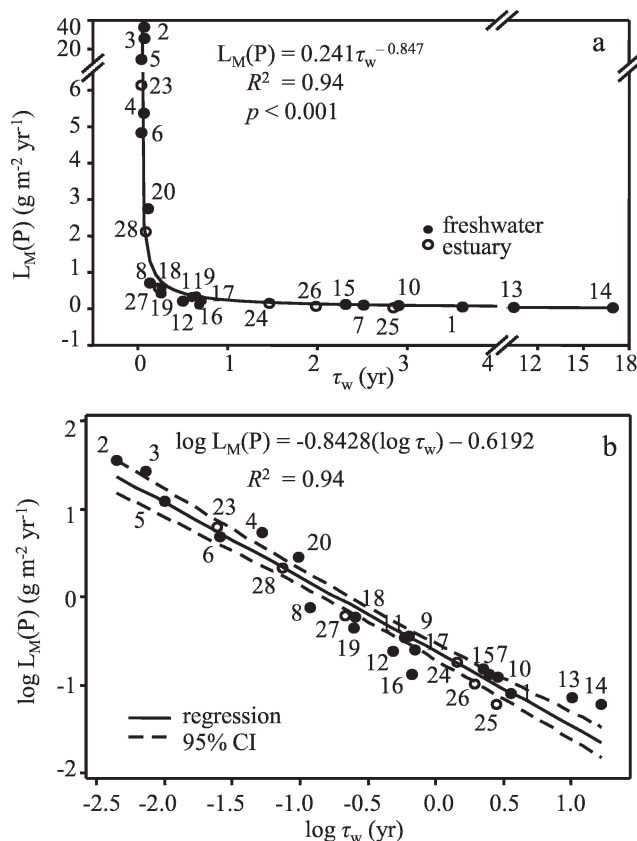


Fig. 4. Agency-established TP load limits, $L_M(P)$, regressed against water residence time, τ_w , for all systems using (a) untransformed data and (b) log-transformed data. Points are labeled with the waterbody identification numbers as shown in Fig. 1 and the tables.

our data set (Table 2), depth varies by a factor of 4.35 and the targeted steady-state P concentration by a factor of 11.4. Sedimentation coefficients vary roughly as the square root of areal load divided by depth (Reckhow and Chapra 1983) and thus will vary within a similar range. Residence time, however, varies by a factor of nearly 4000 among just the $L_M(P)$ systems, and over 6000 from the highest to lowest τ_w values in the entire data set. Thus, $1/\tau_w$ separates the systems to a much greater degree than do any of the other variables and, since L_M is positively correlated with $1/\tau_w$, the $L_M - \tau_w$ relationship approaches the form of the function $1/\tau_w$ vs. τ_w .

Although there is a mathematical explanation for the form of the $L_M - \tau_w$ relationship, there is also a biological one. The high variation in τ_w shows that, among the abiotic factors regulating trophic state, hydrology most strongly distinguishes the range of aquatic ecosystems, and it does so in a biologically relevant fashion. Residence time strongly regulates the utilization of nutrients through its effects on the extent of nutrient recycling and on the time for complete utilization of nutrients in population growth (Lucas et al. 2009). Because both effects increase with τ_w , load limits must decrease with increasing residence time. Thus, the mathematical relationship reveals a fundamental driver for variation in the trophic state of aquatic ecosystems.

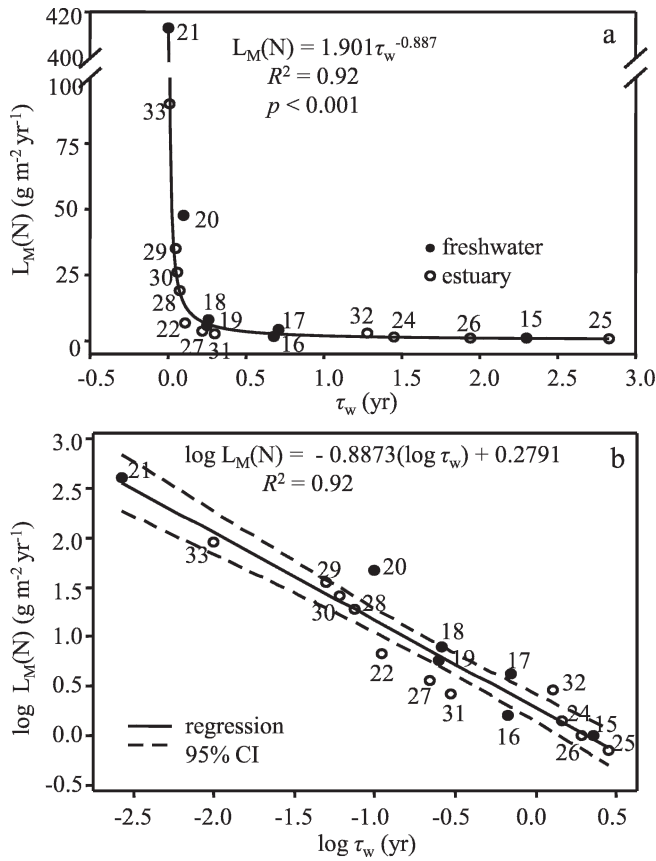


Fig. 5. Agency-established TN load limits, $L_M(N)$, regressed against water residence time, τ_w , for all systems using (a) untransformed data and (b) log-transformed data. Points are labeled with the waterbody numbers as shown in Fig. 1 and the tables.

The adequacy of our models must also stem from two other factors. First, there must have been a common trophic-state goal across the systems. Second, there must be a fundamental similarity between lotic and lentic, colored and clear, and freshwater and estuarine systems in the trophic response to nutrient concentrations.

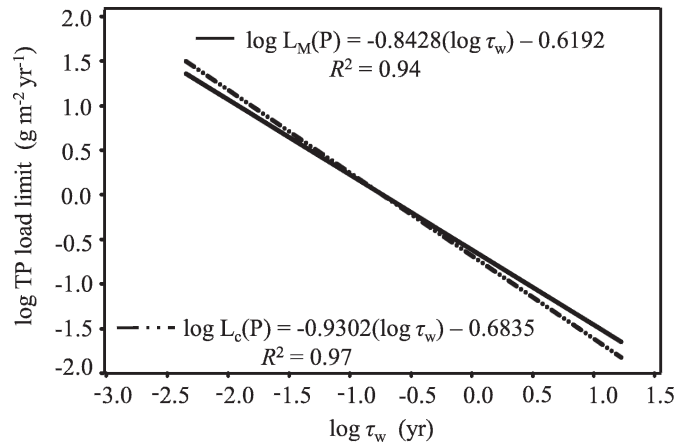


Fig. 6. Comparison between the log-linear plots of the empirical $L_M(P) - \tau_w$ model and the theoretical $L_c(P)$ vs. τ_w relationship. The theoretical $L_c(P)$ was calculated using the OECD equation (see Eq. 1a or 1b).

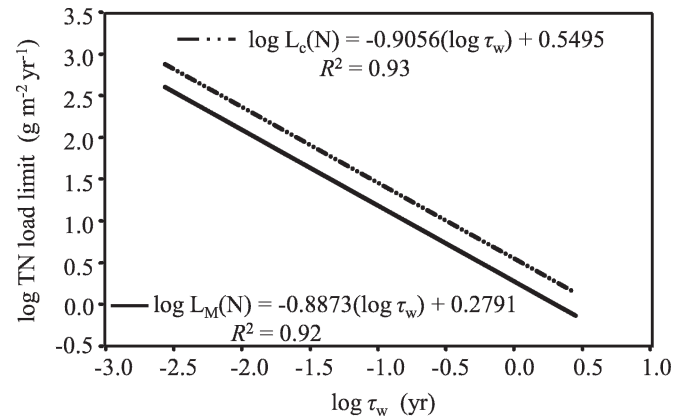


Fig. 7. Comparison between the log-linear plots of the empirical $L_M(N) - \tau_w$ model and the theoretical $L_c(N)$ vs. τ_w relationship. The theoretical $L_c(N)$ was calculated using the OECD equation (see Eq. 1a or 1b).

Regarding trophic state, it appears that the various systems' end points—Chl a , TSI, DO, water transparency, etc. (Table 2)—converged upon a shared trophic state that society views as acceptable, probably near the UML as described by the $L_M - \tau_w$ lines for TP and TN (Figs. 8, 10). This is supported by the high consistency in the state's trophic goals among the systems: mesotrophy, and perhaps at the upper limit of mesotrophy. Most degraded systems are eutrophic or tending toward eutrophy but were mesotrophic or oligomesotrophic in their native state. In most cases, mesotrophy can satisfy state-designated uses as indicated by any of the end points chosen for a system and for which a numeric criterion exists (Table 2). TMDLs and PLRGs appear to be adequate to restore mesotrophy, but they appear only just adequate for systems with short residence times. Because TMDLs and PLRGs are intended to be maxima, a UML target is congruent with policy.

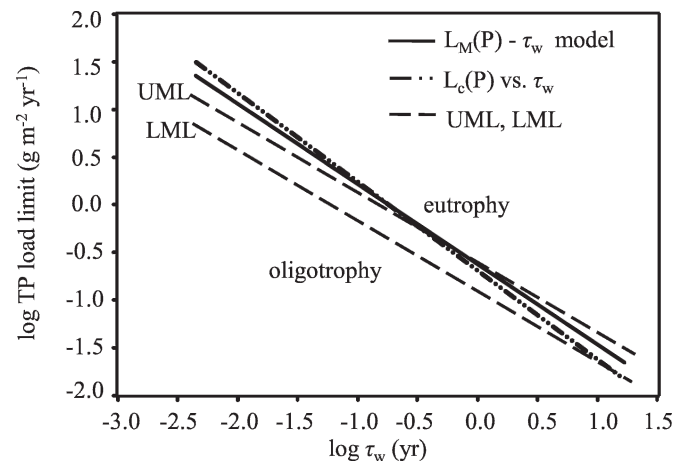


Fig. 8. The positions of the empirical $L_M(P) - \tau_w$ model and the OECD $L_c(P)$ vs. τ_w lines relative to the theoretical upper mesotrophic limit (UML) and lower mesotrophic limit (LML) for TP and for all systems. The OECD equation (Eq. 1a or 1b) was used with the Baker et al. (1985) bounds for [TP] (0.05 and 0.025 mg L⁻¹) to develop the UML and LML. The log-linear equation for the UML line is $y = -0.7389x - 0.6029$ ($R^2 = 0.95$), and for the LML line, it is $y = -0.7389x - 0.9039$ ($R^2 = 0.95$).

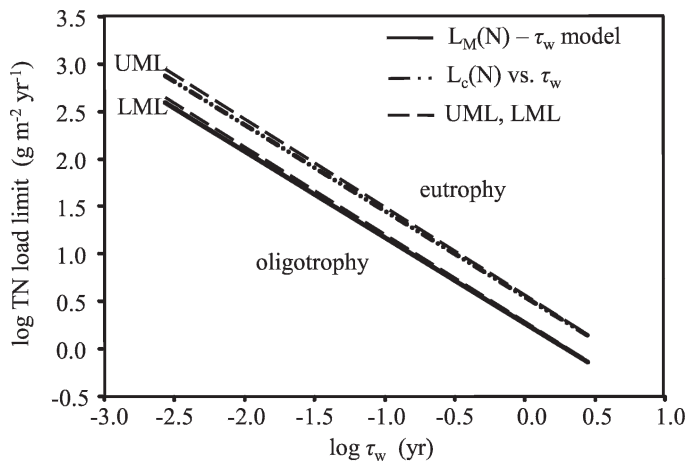


Fig. 9. The positions of the empirical $L_M(N) - \tau_w$ model and the OECD $L_c(N)$ vs. τ_w lines relative to the theoretical upper mesotrophic limit (UML) and lower mesotrophic limit (LML) for TN and for all systems. The OECD equation (Eq. 1a or 1b) was used with the Baker et al. (1985) bounds for [TN] (1.0 and 0.5 mg L⁻¹) to develop the UML and LML. The log-linear equation for the UML line is $y = -0.9238x + 0.581$ ($R^2 = 0.95$), and for the LML line, it is $y = -0.9238x + 0.28$ ($R^2 = 0.95$). In this case, the line representing the all-systems $L_M(N) - \tau_w$ model $\approx$ LML line.

TN load limits predicted by the $L_M(N) - \tau_w$ model should maintain mesotrophy in estuaries and certainly so in freshwater systems (probably tending more toward oligotrophy). In contrast, the OECD equation (Eq. 1a or 1b) appears to be inadequate for predicting mesotrophic TN load limits for Florida's aquatic systems. Additional TN L_M data on freshwater systems may reveal that those systems could be classified separately and be represented by a separate $L_M(N) - \tau_w$ model, and that model may possibly be more consistent with the N analog of the OECD model. However, it is possible that the OECD model fails to adequately capture the internal processes regarding N burial or denitrification in the subtropics. Attempts to modify the OECD model for TN in the subtropics, or even in temperate areas, could yield different equation terms between freshwater systems and estuaries. For example, Dettmann (2001) found that the rate of N losses in estuaries is dependent upon τ_w , like the OECD model, but that rate is unaffected by z , unlike the OECD model. Additionally, estuaries generally have faster rates of denitrification than freshwaters (Hecky and Kilham 1988; Seitzinger 1988).

A fundamental similarity between freshwater and estuarine systems in terms of their primary production or trophic response to nutrient loading is revealed in a general model normalized by τ_w , an observation reported by others who have conducted cross-system analyses. Marine and freshwater phytoplankton share common responses to N and P enrichment and to TN:TP ratios (Hecky and Kilham 1988; Guilford and Hecky 2000; Smith 2006). Others who have applied limnological models, particularly Vollenweider (1976) or the OECD (1982), to estuaries individually (Lee and Jones 1981) or as a group (Boynton et al. 1996; Meeuwig et al. 2000; Dettmann 2001) have shown the significance of τ_w as a fundamental underpinning of an estuary's response to

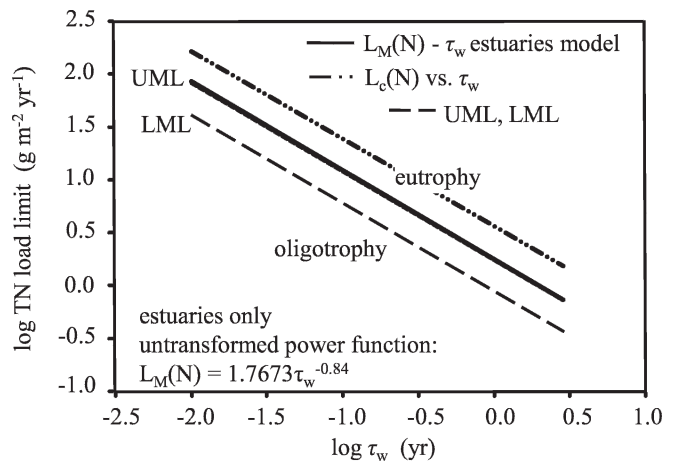


Fig. 10. The positions of the $L_M(N) - \tau_w$ model and the OECD $L_c(N)$ vs. τ_w lines relative to the theoretical upper mesotrophic limit (UML) and lower mesotrophic limit (LML) for TN and for estuaries only. The estuary UML and LML were developed on the basis of the OECD equation (Eq. 1a or 1b) with a [TN] of 0.4 and 0.2 mg L⁻¹, respectively (refer to text in Results for explanation). The log-linear equation for the estuary UML line is $y = -0.8363x + 0.2411$ ($R^2 = 0.96$), and for the estuary LML line, it is $y = -0.8363x - 0.0599$ ($R^2 = 0.96$). In this case, the line representing the estuaries $L_M(N) - \tau_w$ model $\approx$ UML line.

nutrient loading, as it is in limnology. For further discussion on this topic, we defer to Kelly (2008), who provides an excellent synthesis of these and other studies on cross-system comparisons. It is worth noting that these previous comparative studies largely focused on temperate systems. This study serves to expand the discussion to subtropical systems.

Nixon (1988) noted that certain differences between estuaries and lakes diminish as one moves from temperate to tropic latitudes; that relevant conditions (e.g., vertical temperature and mixing gradients) are nearly the same between tropical lakes and estuaries, contributing to similar levels of areal phytoplankton production or biomass. It also seems likely that shared shallow depths should further promote this similarity. The adequacy of single models for TN and TP across shallow fresh and estuarine waters in Florida indicates strong similarity in their trophic response. If their trophic responses are similar, why are target levels for phytoplankton production, measured as $\mu\text{g L}^{-1}$ Chl *a*, so dissimilar between Florida's fresh and estuarine waters? Half of the freshwater systems included in this study have mean annual Chl *a* target levels $\geq 19 \mu\text{g L}^{-1}$, whereas nearly all the estuaries have target levels $< 11 \mu\text{g L}^{-1}$ (see end points in Table 2). In their extensive study of Florida coastal systems, Hoyer et al. (2002) provide convincing evidence that the Chl *a*-to-biomass ratio in phytoplankton is much less in nearshore coastal waters than in lakes; that the amount of Chl *a* per unit of TP is much less in coastal waters, even when factoring out high flushing rates. This latter result was confirmed by Smith (2006) using a larger worldwide data set. Through a review of other studies, Hoyer et al. (2002) concluded that their findings can be explained by the lower Chl *a*-to-biovolume ratios in coastal systems as compared with lakes. These results indicate that as Florida's load limits are met and mesotrophy is

achieved, one should expect to see substantially lower Chl *a* levels in estuaries than in freshwaters but approximate equivalence in phytoplankton biomass.

Color-induced light limitation is important in controlling primary production, particularly in blackwater systems (Meyer 1990; Phlips et al. 2000). However, this phenomenon does not appear to be a significant factor affecting the empirical relationship between τ_w and nutrient load limits among systems with color ranging from 5 to 200 PCU. That relationship may be different for blackwater systems > 200 PCU as inferred by Phlips et al. (2000), but Canfield et al. (1984) found that the effect of high color on algal abundance and biomass in Florida lakes, even as high as 416 PCU, is not very strong and that development of separate nutrient-trophic state models on the basis of color is not warranted. It is interesting to note that color and τ_w are often inversely correlated, as observed in the St. Johns River (J. Hendrickson and L. Keenan, St. Johns River Water Management District pers. comm., 29 Jan 2009), Suwannee River estuary (Bledsoe et al. 2004), as well as in a very large and diverse set of north temperate lakes where τ_w strongly controls color-dissolved organic matter levels and nutrient cycling (Webster et al. 2008).

If the consequence of reducing trophic state from eutrophy to mesotrophy is the reduction of (and approximate equivalence in) phytoplankton biomass across systems, estuarine and fresh, clear and colored, then water transparency should respond in kind, leading to an increase in maximum colonization depths (MCD) of submersed macrophytes (notwithstanding other depth-limiting factors). This trophic state transition, then, is associated with a shift in primary producer community structure from phytoplankton dominance toward macrophyte dominance (Scheffer et al. 1993). Under mesotrophic conditions, the potential MCD of macrophytes could reach or exceed the mean water depths (≥ 2 m) of the shallow fresh and estuarine systems of this study (see range of mean depths in Table 2). This assessment is based on an examination of the range of light requirements and MCD for Florida's freshwater macrophytes (Canfield et al. 1985) and seagrasses (Dixon 1999; Steward et al. 2005) and the assumption that mesotrophy will provide water transparencies that approach background levels (e.g., 90% of background water transparency, see response end points in Table 2).

Another reason why the general models presented here work well across systems is the normalization of τ_w to R_{99} . Apparently, the normalization was appropriate and sufficiently accurate. Nixon et al. (1996) and Boynton and Kemp (2000) indicated the need to scale nutrient loads to τ_w or normalize τ_w among estuaries to successfully develop general load-response models. Kelly (2008) demonstrated that τ_w normalization could help produce models across freshwater and estuarine systems as we have done here. Kelly (2008) goes on to state that "Estuarine scientists have been slow to incorporate this [τ_w modeling] concept..." because of the perceived lack of accurate τ_w . Hecky and Kilham (1988) believe that the slower pace is also due to the difficulty in delineating boundaries for coastal systems. Reasonably accurate τ_w are available for many estuaries or

can be generated using mechanistic models or much simpler box models (Hagy et al. 2000). Although defining boundaries for coastal systems can pose a significant challenge, for many estuaries it can be resolved as it was for the estuaries included in this study.

The models and methods presented here should be useful in estimation of nutrient L_M for Florida's aquatic systems and may have broader utility. The empirical $L_M - \tau_w$ models (or the OECD model) can be used to predict TMDLs for Florida's estuaries and inland waters as they are generally represented in the body of literature (Table 1), as annual or long-term averaged external load limits. Apparently, TMDLs on the basis of these empirical models will achieve and maintain mesotrophy.

We believe that these models can complement, but not replace, site-specific analysis where data are available or where time and funding are sufficient to obtain adequate data. TMDLs for some systems deviate significantly from the general models (e.g., Lake Beauclair for TP and Lemon Bay for TN; Figs. 2–5) and these cannot be identified a priori. Therefore, the general models are reasonable first approximations until sufficient site-specific data are available.

These general models may also be useful in the development of numeric nutrient criteria. There are well over 7000 waterbodies in Florida (Huber et al. 1982) and U.S. EPA and FDEP are responsible for developing numeric nutrient criteria applicable to all of them. Our work indicates that τ_w and its relationship to nutrient load limits is an appropriate common factor that could lend itself to the development of numeric criteria applicable to various aquatic systems. According to our general models, a numeric nutrient criterion for any system would be defined as a steady-state areal loading rate derived from τ_w .

Alternatively, a numeric criterion for a steady-state TP concentration target can also be approximated on the basis of our empirical $L_M(P)$ model and its close comparison with the OECD model. The OECD model is a power function and can be expressed as $L_c(P) = ([TP]_c \times z(1 + \tau_w^{0.5}))\tau_w^{-1}$. The empirical $L_M(P) - \tau_w$ model is also a power function: $L_M(P) = 0.24\tau_w^{-0.847}$. A similar function was produced when we calculated the theoretical estimates of $L_c(P)$ (by using Eq. 1a or 1b with TP concentration targets from Table 2) and regressed those values against τ_w : $L_c(P) = 0.21\tau_w^{-0.93}$. It appears that the coefficients of those functions (0.24 and 0.21) represent the coefficient of the OECD function: $[TP]_c \times z(1 + \tau_w^{0.5})$, and that the $L_M(P)$ equation differs from the OECD equation only in the exponent of τ_w ($\tau_w^{-0.847}$ vs. τ_w^{-1}). This suggests that the equation of the $L_M(P)$ model can be represented as follows:

$$L_M(P) = [TP]_M \times z(1 + \tau_w^{0.5}) \times \tau_w^{-0.847}, \quad (2)$$

which is rewritten to solve for $[TP]_M$ as follows:

$$[TP]_M = L_M(P) \times \tau_w^{0.847} / z(1 + \tau_w^{0.5}). \quad (3)$$

Predictions of $[TP]_M$ calculated in this manner correlate well ($r = 0.81$, $n = 26$) with the agency-established $[TP]_M$ targets for the systems in this study (Table 2). A TN concentration criterion complementary to the areal TN loading criterion (or $L_M(N)$ models) cannot be calculated

as described above because of the dissimilarity between the $L_M(N)$ and OECD $L_c(N)$ models (Figs. 9, 10). However, if a target TN:TP ratio can be established for a system, then a TN concentration target could be calculated on the basis of a predicted TP target.

Loading or concentration criteria on the basis of the $L_M - \tau_w$ models would maintain or restore mesotrophy in Florida systems. Oligotrophic systems would not be protected by these models; however, a separate set of criteria could be developed for those systems on the basis of the Baker or estuary LML, or a more empirically derived LML.

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