Predicting coastal eutrophication in the Baltic: a limnological approach

Jessica J. Meeuwig, Pirkko Kauppila, and Heikki Pitkänen

Abstract: Coastal eutrophication is a key environmental concern in Finland. A highly indented, well-settled coastline with a myriad of small estuaries means that eutrophication occurs at numerous localities. There is a clear need for general models that predict eutrophication across estuaries. Lake eutrophication has been successfully predicted using a combination of chlorophyll *a* (Chl) – total phosphorus (TP) regression models and TP mass-balance models. We applied this limnological approach to 19 Finnish estuaries. The Chl–TP regression was highly significant, accounting for 67% of the variation in Chl. When combined with a TP mass-balance equation, log observed and predicted Chl differed by 28% on average. Accuracy was improved by dividing the estuaries into those dominated by non-point-source (NPS) loading (n = 11) and those dominated by point-source (PS) loading (n = 7). A land-use regression model based on percentage of the catchment forested and estuarine mean depth then best predicted Chl in the NPS-dominated estuaries. The mass-balance approach remained the most accurate model for the PS estuaries. The land-use model and mass-balance approach are complementary tools in that their use maximizes accuracy for both NPS- and PS-dominated estuaries. This high level of accuracy demonstrates the relevance of limnological approaches to Finnish estuaries.

Résumé : L'eutrophisation du littoral est un problème environnemental grave en Finlande. Étant donné que la ligne de côte est extrêmement découpée et fortement colonisée, et compte une myriade de petits estuaires, l'eutrophisation se manifeste à de nombreux endroits. Il est clairement nécessaire d'établir les modèles généraux permettant de prédire l'eutrophisation d'un estuaire à l'autre. Dans le cas des lacs, on a réussi à prédire l'eutrophisation grâce à une combinaison de modèles de régression chlorophylle (Chl) / phosphore total (PT) et de modèles du bilan massique du PT. Nous avons appliqué cette approche limnologique à 19 estuaires finnois. La régression Chl:PT était fortement significative, puisqu'elle rendait compte de 67 % de la variation de Chl. Si on y combinait une équation du bilan massique du PT, la différence des logarithmes des valeurs observées et prédites de Chl était de 28 % en moyenne. Nous avons amélioré la précision en séparant les estuaires entre ceux qui étaient dominés par une charge de pollution diffuse (PD) (n = 11) et ceux qui étaient dominés par une charge de pollution ponctuelle (PP) (n = 7). Un modèle de régression de l'utilisation des terres basé sur le pourcentage boisé du bassin et la profondeur moyenne de l'estuaire donnait alors les meilleures prédictions pour Chl dans les estuaires dominés par la pollution diffuse. L'approche du bilan massique demeurait le modèle le plus exact pour les estuaires à pollution ponctuelle. Le modèle de l'tilisation des terres et l'approche du bilan massique sont des outils complémentaires du fait que leur utilisation maximise la précision pour les deux types d'estuaires. Ce haut niveau de précision montre la pertinence des approches limnologiques pour les estuaires finnois.

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Introduction

Coastal eutrophication has been identified as a global problem from temperate estuaries (Rosenberg 1990) to tropical waters (Lapointe and Clark 1992). It manifests as an increase in phytoplankton and macroalgal biomass, increased incidences of toxic and noxious blooms, hypoxia and anoxia, and fish and benthos kills (Fisher et al. 1995). The Baltic Sea is perhaps one of the first coastal systems in which eutrophication was identified (Rosenberg 1990), and it continues to be a key environmental concern (HELCOM 1997). Finland's coastal waters are particularly sensitive to coastal eutrophication: they are generally shallow, and water exchange with the open Baltic is restricted due to the complex coastal morphometry (Bonsdorff et al. 1997). Nutrient loads are derived from agriculture (Rekolainen et al. 1995), municipal and industrial wastes (Pitkänen 1994), and fish farms (Bonsdorff et al. 1997). Finland has a long and highly indented coastline (39 000 km; Pitkänen 1994) that includes approximately 50 river-fed estuaries as well as numerous embayments. Human settlement, while concentrated in the south and southwest, occurs along the entire coastline. Such a coastline and settlement pattern mean that eutrophication is not a localized problem; rather, it occurs in a number of estuaries from the eastern Gulf of Finland to the northeast Bothnian Bay.

The Baltic Sea has received much attention in terms of modeling eutrophication (Wulff et al. 1990). However, much of Finland's coastline comprises a myriad of small estuaries for which eutrophication models have not been developed.

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Given constraints on human and financial resources, it is also unlikely that site-specific models will be developed for the majority of small estuaries. Thus, there is a need to develop general models that predict eutrophication for a variety of systems. To be relevant to environmental managers, these models must (*i*) be accurate, (*ii*) link the eutrophication response with variables that can be managed, and (*iii*) quantify the error associated with predictions so that decision risk can be evaluated.

Such models have been developed to address lake eutrophication. OECD (1982) used a mass-balance approach to estimate total phosphorus (TP) concentrations in lakes as a function of TP loading (estimated from land-use activities), water residence time, and P sedimentation. The TP estimate from a mass-balance equation is then used in a regression model (e.g., Dillon and Rigler 1974; OECD 1982) that predicts response variables of concern, typically phytoplankton biomass (as chlorophyll *a* (Chl)). This approach has formed the basis of successful lake eutrophication management both in Europe and North America (OECD 1982). Meeuwig and Peters (1996) demonstrated that regression models based on land use also accurately predict Chl and are an alternative to the P-based mass-balance approach.

In Finland, riverborne nutrients have been identified as the major source of excess nutrients both within estuaries and in the open coastal waters (e.g., Pitkänen 1994). These nutrients are derived primarily from agricultural activities and to a lesser extent from point-source waste waters (Rekolainen et al. 1995). To date, strong relationships have been developed to predict total nitrogen (TN) and TP loads as a function of land use (r^2 = 0.80 and 0.73, respectively; Pitkänen 1994). However, no analogous empirical models have yet been developed to predict phytoplankton biomass in Finnish estuaries as a function of either ambient nutrient concentrations, nutrient loads, or land use. Nor has the mass-balance approach, so successful in lakes, been applied to these systems despite the fact that these Baltic estuaries are thought to be P limited (Pitkänen and Tamminen 1994). We have compiled a data set for 19 estuaries and their watersheds to test the hypothesis that the mass-balance approach accurately predicts Chl in Finnish estuaries. To test this hypothesis, we must demonstrate that (*i*) total nutrients (TN and TP) accurately estimate estuarine Chl, (*ii*) mass-balance models accurately estimate TP, and (*iii*) TP estimated from mass-balances accurately estimates estuarine Chl. We also tested the hypothesis that land use, which integrates a number of factors affecting phytoplankton biomass, better estimates Chl than TP or the mass-balance approach.

Materials and methods

Description of data set

Data were compiled for 19 estuaries located on the Baltic coast of Finland from the northeast Bothnian Bay to the eastern Gulf of Finland, near Russia (Fig. 1). Each estuary is hence referred to by the name of the river with which it is associated (Table 1). The data set includes information on: phytoplankton biomass (as Chl), water chemistry, coastal morphometry, land use, and total nutrient loads (Table 1). Chl and water chemistry data are from monitoring surveys between 1989 and 1993 conducted by the Regional Environmental Centers of Finland and coordinated by the Finnish Environment Institute. TP, TN, and conductivity were measured along vertical profiles, while Chl was measured from integrated water samples from the surface to 5 m. Chl, TP, and TN were analyzed from unfiltered samples using Finnish standard methods (Pitkänen 1994). Conductivity (*C*, millisiemens per metre) was also measured and used to estimate salinity (S, parts per thousand) (National Board of Waters 1981):

$$(1) \qquad S = (C - 60.91) \times 163^{-1}.$$

There are altogether approximately 500 sampling stations in the coastal waters monitoring program. Of these 500 stations, we included 176 stations that were located “within” the estuaries. Estuarine boundaries are typically difficult to define. We used a geographical approach, drawing the outer limit of the estuary across the narrowest part of the outermost headlands. Growing season averages were calculated for Chl, TP, TN, and salinity using all measurements taken at depths less than 5 m between June and August.

Most estuaries included in the study were well sampled (Table 2). Of the 19 estuaries, 12 included data for five years. Spatially, there was an average of 4.2 stations per estuary with five estuaries represented by a single station. These five estuaries were generally the smaller systems (mean surface area = 23 km^2 versus 48 km^2 for all estuaries). Although represented by a single station, three of these five estuaries were sampled three to six times during the growing season. Temporally, the sampling effort in a given area varies depending on the source and amounts of loading and local hydrographical conditions. The sampling frequency ranges in general from two to 20 times per year, being most typically from four to six times per year. In our data set, the mean number of samples per growing season was 3.4 and the mode was 2.

Phytoplankton biomass and nutrients are strongly affected by variation in hydrology (Schaub and Gieskes 1991), which varies annually. We incorporated this interannual variability by calculating mean values X_{mean} for each estuary as

$$(2) \qquad X_{\mathrm{mean}} = \left\langle \sum_{y=1}^{y=i_y} \left\{ \sum_{s=1}^{s=i_s} \left[\left(\sum_{m=1}^{m=i_m} \right) n_m^{-1} \right] \right\} n_s^{-1} \right\rangle n_y^{-1}$$

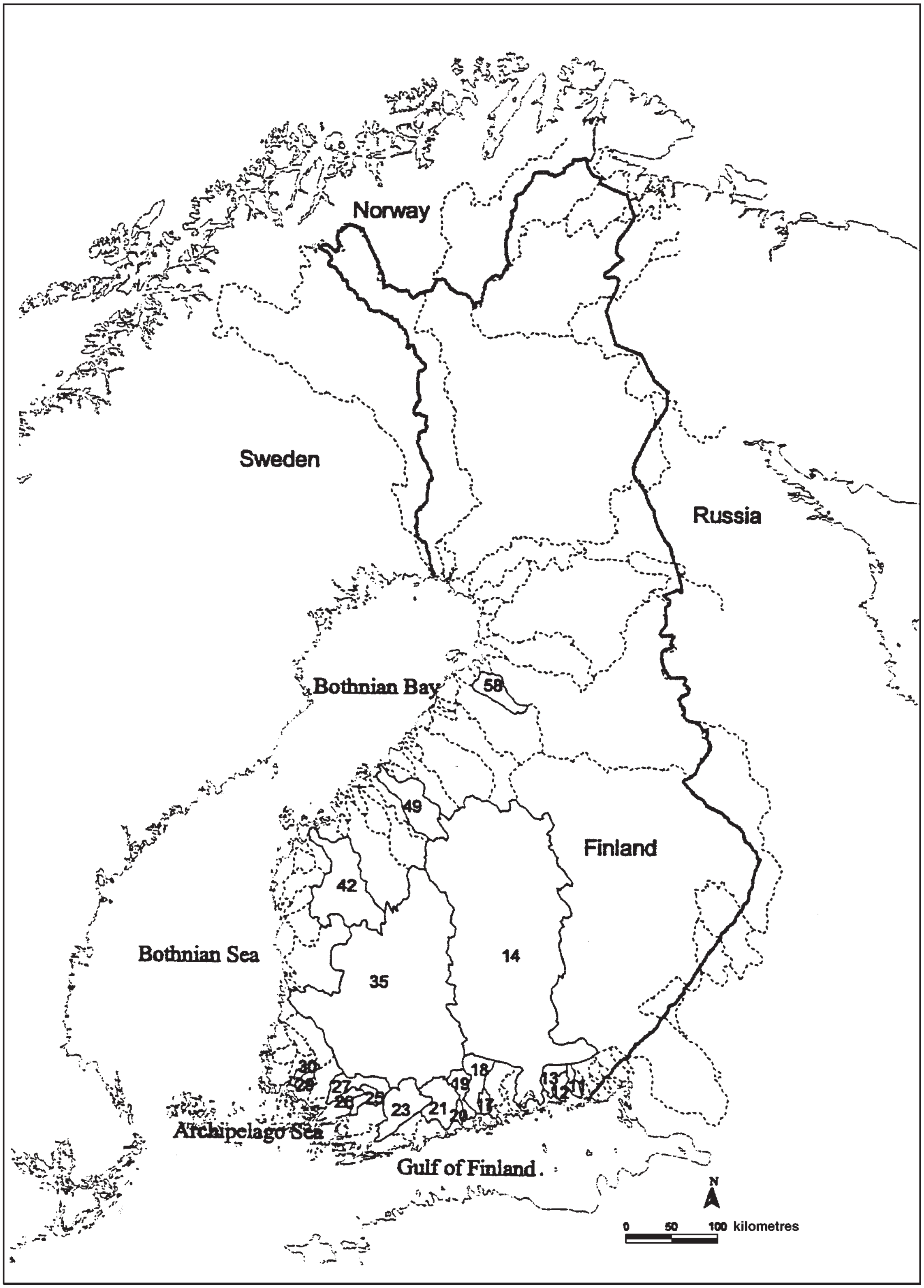
where *X* is the value for the variable of interest in a given year (*y*) at a given station (*s*) in a given month (*m*) and *n* indicates the number of years (y), stations (s), or months (m). This averaging approach gives equal weight to each year even if the year is represented by a single station. By emphasizing years, we were also able to better match the water chemistry data to the hydrologic flow and nutrient loading data that were calculated on a yearly basis.

Estuarine surface area and mean depth were calculated from 1 : 50 000 bathymetric charts (Finnish Institute of Navigation 1996–1998). Surface area was measured by planimetry. Mean depth was estimated using a grid technique whereby the depth under each square of the grid covering the estuary was recorded and the average of these values taken. The mean depths estimated using this approach may overestimate the true mean depths, as the available bathymetric charts did not indicate depths for the shallower fringing areas at the mouths of the rivers. These areas were, however, generally small relative to the total surface area.

Water residence time (years) was calculated using two different approaches: (*i*) Bowden’s (1980) saltwater fraction method and (*ii*) freshwater replacement (Vollenweider 1975). Mean salinity for Bowden’s (1980) method was calculated for the entire estuary as were the Chl and nutrient mean values. Freshwater inflow was calculated for each year from 1989 to 1993. Mean values for each estuary were then calculated by averaging values for those years for which Chl and nutrient data were available.

Catchment size and land-use data from the early 1990s were obtained from the databases of the Finnish Environment Institute fol-

Fig. 1. Map of Finland indicating the locations (by code, Table 1) of the estuaries included in this study.



lowing Pitkänen (1994). Human population density was estimated using Arc Info by overlaying municipal and watershed boundaries (Finnish Environment Institute 1998).

TP and TN loads for the watershed were calculated from the annual river loads (L_R) by multiplying the monthly concentration (C_m) by the monthly water flow (Q_m) and summing the monthly loads:

$$L_R = \Sigma(Q_m C_m). \tag{3}$$

The frequency of flow proportional sampling was typically 12 times per year, ranging from four to 20 times per year. With low sampling frequencies, the estimates of river fluxes are expected to underestimate the true loads (Walling and Webb 1985). However, this effect is mitigated by using multiple years of data. The loads for nonmonitored river catchments (Table 1) were extrapolated on the basis of small coastal rivers within the four main catchment areas of the Finnish coastal waters (Pitkänen 1994). Six of the estuaries also receive point-source loading from municipalities and industry. These loads, obtained from the Finnish Environment Institute databases, were calculated as annual means. As with the river inflows, river and point-source nutrient loads were calculated for each year from 1989 to 1993. Mean values for each estuary were then calculated by averaging the values for years for which Chl and nutrient data exist.

Mass-balance approach

The mass-balance approach to predicting eutrophication involves two steps. First, TP is estimated from a mass-balance equation as a function of TP load, sedimentation, and flushing (Vollenweider 1975). The estimated TP value is then used to predict Chl via an empirical regression model such as that of Dillon and Rigler (1974) or the OECD (1982).

We chose to use an existing mass-balance equation for the Finnish estuaries, as we lacked information on general patterns in P sedimentation rates in these systems. There are a number of mass-balance equations in the literature, but with few guidelines as to their use. The equations are similar in structure, differing principally in their estimates of P sedimentation. We chose Canfield and Bachmann's (1981) mass-balance equation because, in a review of 17 mass-balance equations, Meeuwig and Peters (1996) demonstrated that this equation estimated TP values that most accurately predicted lake Chl. Their equation is

$$(4)\qquad \mathrm{TP} = \mathrm{APL}(1 - (\sigma/(\sigma + \rho)))/Z_{\mathrm{m}}\rho \quad \text{in which } \sigma = 0.129(\mathrm{APL}/Z_{\mathrm{m}})^{0.549}$$

where APL is areal P loading (milligrams per square metre per year), Z_{m} is mean depth (metres), ρ is hydraulic flushing rate (per year), and σ is the P sedimentation coefficient (per year). The TP values estimated from the Canfield and Bachmann (1981) equation were then used in a Chl–TP regression model developed in this study specific to the Finnish estuaries.

Statistical analyses and goodness-of-fit criteria

Standard least squares regression techniques were used to develop regression models estimating Chl as a function of total nutrients and land use (Zar 1984). All variables were log transformed to stabilize variation and render the data linear (Zar 1984). Land-use variables that were calculated as a percentage of the watershed were transformed as $\log_{10}(X + 1)$ due to the presence of zeros in some of the land-use categories.

Whereas the Chl – total nutrient relations are univariate, the land-use models include two independent variables. Aquatic systems respond differently to disturbance as a function of their sensitivity, usually determined by morphometry. This can be conceptualized in terms of a load – sensitivity effect relationship. Thus the land-use models include one variable indicating the load or disturbance (e.g., the amount of agriculture, forested land, or human population density) and one variable indicating their sensitivity (e.g., mean depth or water residence time). Together, these two variables estimate the response variable, Chl. A set of preliminary models was identified using exploratory regression techniques; the "best" model was then chosen based on model standard errors and the significance levels of partial regression coefficients.

To quantitatively compare the accuracy, precision, and bias of Chl values estimated from the mass-balance approach and the land-use model, we used criteria based on the least squares goodness-of-fit criterion: the mean squared residual (MSR). Following Meeuwig and Peters (1996), accuracy was estimated as

$$(5)\qquad \mathrm{MSR} = (\Sigma(\mathrm{lChl_o} - \mathrm{lChl_p})^2)n^{-1}$$

where $(\mathrm{lChl_o} - \mathrm{lChl_p})$ is the difference between the log values of observed and predicted Chl and n is the number of observations. This MSR differs from the model MSR calculated in regression models in that the denominator is the sample size (n) rather than n adjusted for the number of estimated parametres. The adjustment was made because (*i*) it is not possible to calculate the degrees of freedom associated with the mass-balance approach and (*ii*) we wished simply to know how great is the difference between observed and predicted. The variance of the squared residuals (vSR) and the mean error (ME) were used as criteria of precision and bias (Meeuwig and Peters 1996):

$$(6)\qquad \mathrm{vSR} = (\Sigma((\mathrm{lChl_o} - \mathrm{lChl_p})^2 - \mathrm{MSR}))^2(n - 1)^{-1}$$

$$(7)\qquad \mathrm{ME} = (\Sigma(\mathrm{lChl_o} - \mathrm{lChl_p}))n^{-1}.$$

The MSR, vSR, and ME were calculated for each estuary for log Chl estimated via the mass-balance equation and log Chl estimated from the land-use model. These individual estuary values were then averaged to evaluate the overall accuracy, precision, and bias of the estimates of the two types of models.

Accuracy, precision, and bias assess the goodness-of-fit of these models to the data. The models are predictive in the sense that they estimate a Chl value for each of the estuaries included in the analysis. The small sample size restricts our ability to utilize techniques such as cross-validation or bootstrapping to assess their more general predictive capacity. Thus, we are assuming those models found to best fit these data will also best predict Chl in similar estuaries not included in the analysis.

Results

The 19 estuaries encompass a wide range of conditions (Table 1). Size varies from mean depths of 3 to 18 m and surface areas of from 2 to 145 km^2. Growing season Chl ranges from 3.9 to 45.9 mg·m^{-3} and TP and TN range from 20 to 92.2 mg·m^{-3} and from 320 to 2133 mg·m^{-3}, respectively. Annual P and N loads vary across two orders of magnitude from 5 to 447 t·year^{-1} and from 130 to 10433 t·year^{-1}, respectively. Salinity ranges from 0 ppt in Kyrönjoki in the oligohaline Bothnian Bay to 6.1 ppt in Paimionjoki in the westernmost Finnish archipelago. Lunar tides are nonexistent, but occasional wind-driven water level changes can be as high as 50 cm. The hydrological water balance in the Baltic also affects water levels, with fluctuations of approximately 7 cm in the Bothnian Bay and 16 cm in the Gulf of Finland (Pitkänen 1994). Coastal morphometry is also highly variable and includes relatively enclosed systems such as Virojoki, the winding, island-rich systems of the Finnish archipelago such as Paimionjoki, and relatively simple pocket estuaries such as Temmesjoki. Land use is also highly variable: the percentage of the watershed under agriculture ranges from 9.5 to 42.9% with a mean value of 23.9%, and the percentage of forest ranges from 54.3 to 87.2% (Table 1).

Estimating Chl from total nutrients

Regression models estimating log Chl as a function of log TP and log TN were highly significant (Fig. 2). The relationship between log Chl and log TP was stronger than that based on log TN, explaining 67% of the variation in log Chl as compared with 53%. The TP relationship is similar to those generated in lakes: the coefficients are intermediate between those of the Dillon and Rigler (1974) and OECD (1982) equations (Table 3) and indeed are intermediate for a

Table 1. Descriptive data for the estuaries.

Estuary name (Finnish code)	Chl ($mg \cdot m^{-3}$)	TP ($mg \cdot m^{-3}$)	TN ($mg \cdot m^{-3}$)	Sec (m)	Sal (ppt)	Zm (m)	Ao (km^2)	Vol (10^6 m^3)	QR ($m^3 \cdot s^{-1}$)	Res (years)
Virojoki (11)	17.3	47.8	558	1.97	3.7	4.4	32.6	144	3.7	1.23
Vehkajoki (12)*	5.6	46.8	509	1.47	3.9	6.0	10.0	59.8	3.6	0.51
Summanjoki (13)*	3.9	35.3	475	2.43	4.0	4.5	2.1	9.6	5.4	0.05
Kymijoki (14)**	9.8	39.3	510	1.72	2.5	4.9	51.8	252	328	0.02
Ilolanjoki (17)	5.8	28.0	450	2.05					2.9	
Porvoonjoki (18)	30.5	66.6	697	1.28	4.4	12.3	48.8	601	13.6	1.40
Mustijoki (19)	9.4	36.0	411	2.12	5.1	11.8	35.9	425	6.1	2.22
Sipoonjoki (20)*	12.4	59.9	633	1.00	4.3	3.8	2.0	7.5	2.1	0.11
Vantaanjoki (21)	45.9	91.2	1214	0.55	2.8	7.2	20.4	147	16.4	0.28
Karjaanjoki (23)**	6.3	23.0	410	2.85	2.4	12.2	79.2	965	20.1	1.52
Halikonjoki (25)	15.7	65.9	573	1.09	4.3	8.1	93.2	755	6.7	3.56
Paimionjoki (27)	4.5	28.2	420	2.19	6.1	17.9	144.7	2593	10.1	8.18
Hirvijoki (29)*	5.5	33.7	460	1.13	5.8	6.6	35.3	233	2.8	2.65
Laajoki (30)*	7.9	34.9	403	0.86	5.2	4.6	82.6	376	2.8	4.21
Kokemäenjoki (35)**	13.4	39.1	451	1.23	2.0	3.1	31.4	97.7	256	0.01
Närpiönjoki (39)*	4.2	20.0	320	2.00	5.6	6.4	34.2	219	10.1	0.66
Kyrönjoki (42)	23.2	92.2	2133	0.47	0.0				53.2	
Perhonjoki (49)	7.5	20.2	335	1.82	3.0	4.4	6.2	27.0	21.3	0.04
Temmesjoki (58)*	9.9	37.8	696	0.79	1.8	3.1	86.2	266	12.1	0.70
Mean	12.6	44.5	613	1.53	3.7	7.1	46.9	422	40.9	1.61
SD	10.7	21.5	417	0.67	1.7	4.11	38.90	621	90.0	2.12
Minimum	3.9	20.0	320	0.47	0.0	3.08	1.96	7.49	2.14	0.01
Maximum	45.9	92.2	2133	2.85	6.1	17.92	144.70	2593	327.76	8.18

Note: Sec, Secchi depth; Sal, salinity; Zm, mean depth; Ao, surface area; Vol, volume; QR, river water loading; Res, water residence time; Urb-P, Ag-P, and For-P, percentage of the watershed that is urban, agricultural, or forested, respectively; Wshed, watershed area; Pden, human population density; TPL-R and TNL-R, non-point-source TP and TN loads, respectively; TPL-D and TNL-D, point-source TP and TN loads, respectively. The last column is the ratio of direct to total TP load. Missing values for the direct loads indicate estuaries where the direct load is less than 0.1. *, unmonitored rivers; **, watersheds where only lower catchment was used due to the high proportion of lakes.

Table 2. Distribution of sampling effort for Chl where years is the number of years sampled, stations is the number of stations sampled in each year, and season indicates the mean, minimum, and maximum number of samples taken at a given station during the growing season, averaged for all stations in the estuary.

		Stations					Season		
Estuary	Years	1989	1990	1991	1992	1993	Mean	Minimum	Maximum
11	1	7	0	0	0	0	2	2	2
12	1	2	0	0	0	0	3	3	3
13	1	1	0	0	0	0	3	3	3
14	5	2	2	2	2	2	2.5	1	7
17	2	0	0	0	1	1	1.5	1	2
18	5	15	15	15	15	13	4.7	4	6
19	5	2	2	2	2	1	4.7	4	5
20	4	3	3	0	3	3	2.8	1	4
21	5	1	1	1	1	1	6.6	6	7
23	5	3	3	3	3	3	4.3	3	5
25	5	15	16	17	16	17	2.1	1	5
27	5	4	4	4	4	4	2.2	2	3
29	5	1	1	1	1	1	3	3	3
30	5	2	2	2	2	2	1.9	1	3
35	5	6	6	6	6	1	3.4	1	9
39	2	1	1	0	0	0	1	1	1
42	5	1	1	1	1	1	1.8	1	2
49	5	2	2	2	2	2	2.6	1	5
58	5	5	5	5	5	4	2.8	2	3

Urb-P	Ag-P	For-P	Wshed (km^2)	Pden (km^{-2})	TPL-R ($t \cdot year^{-1}$)	TPL-D ($t \cdot year^{-1}$)	TNL-R ($t \cdot year^{-1}$)	TNL-D ($t \cdot year^{-1}$)	Ratio
0.5	13.5	82.9	357	9.6	7.0	3.0	242	23.6	30
0.8	14.0	80.7	380	28.3	9.4		232		
0.4	15.3	82.4	569	24.3	14.1		348		
3.8	22.8	70.1	989	80.4	259.8		6 858		
0.7	23.1	73.4	309	36.0					
2.4	28.5	67.9	1273	64.7	50.8	12.8	1609	73.6	20
1.3	26.8	70.7	783	30.5	20.4	1.8	468	91.0	8
2.5	32.3	64.8	220	46.9	5.3		131		
6.7	23.6	67.7	1686	265.1	53.4		1 453		
2.8	29.9	65.3	116	37.8	26.8	1.3	683	45.4	5
2.0	34.0	63.6	764	33.4	43.2	0.9	501	84.4	2
1.2	43.0	54.3	1088	17.7	80.8		952		
1.5	33.0	65.4	284	28.3	15.3		224		
1.2	19.0	78.6	685	14.0	15.5		227		
1.4	27.0	67.7	6817	26.4	446.6	20.0	10 433	337.4	4
0.7	20.6	78.0	992		21.1		369		
1.0	23.3	74.3	4923	20.0	163.2		3 215		
0.3	9.5	87.2	2524	11.7	66.0		828		
0.4	15.4	83.6	1181	9.0	28.1		431		
1.7	24.5	72.6	1365	43.6	73.7	6.6	1 623	109.2	12
1.54	8.36	8.59	1719	58.3	113.0	7.9	2 730	114.6	11.1
0.35	9.50	54.34	116	9.0	5.3	0.9	131	23.6	2.0
6.73	42.97	87.16	6817	265.1	446.6	20.0	10 433	337.4	30.0

range of Chl–TP equations (Fig. 3). The estuarine Chl–TN relationship is less similar to those in lakes: both the intercept and slope are shallower (Table 3). It is unclear, however, whether this reflects a true difference between lakes and Finnish estuaries, a difference in range, or the relatively weaker fit of the model.

Estimating Chl from the mass-balance

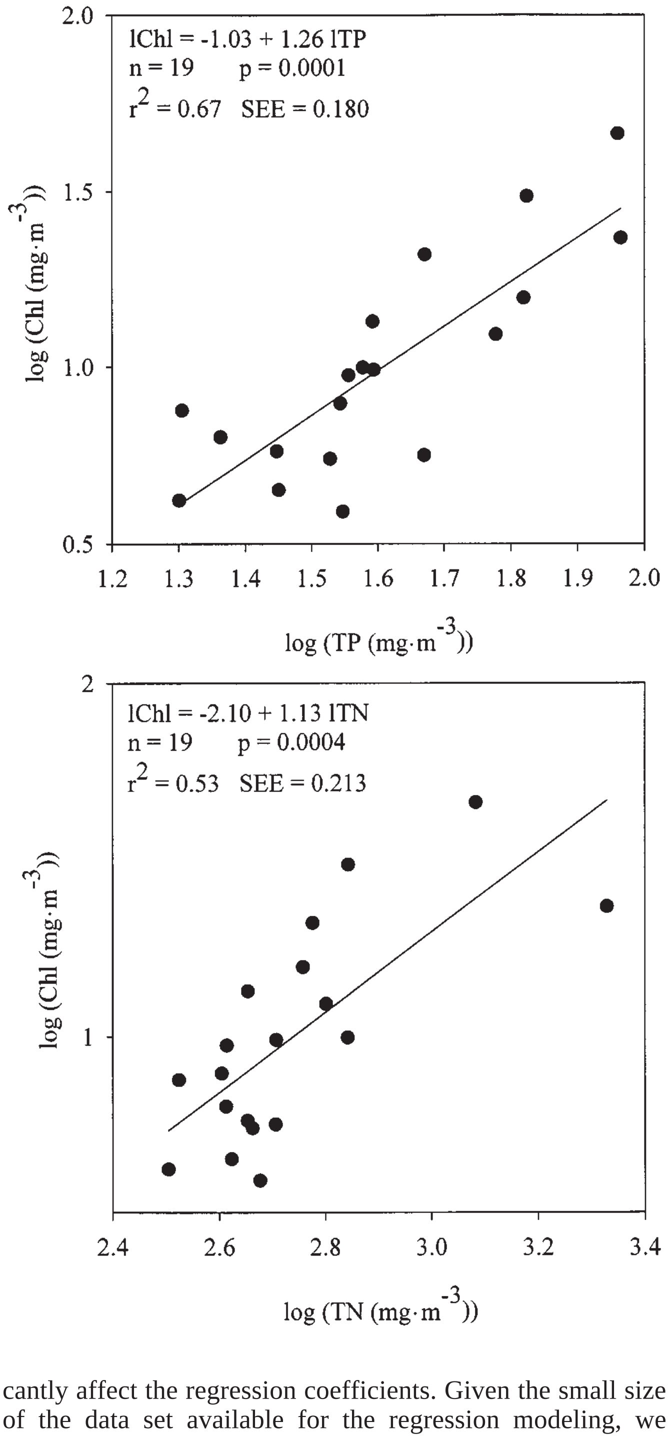
We first used the mass-balance to estimate TP using the two different estimates of water residence time (Table 4). Freshwater replacement time generated the best estimates of TP when compared with observed values of TP, while Bowden's (1980) saltwater fraction method generally underestimated TP by an order of magnitude (Table 4). This underestimate suggests that either the TP loads are underestimated, there are large internal loads, or the water residence times are underestimated. Because we are confident in the estimates of TP loading and because we have little information on internal loading for most estuaries, we have assumed that it is the water residence time that is underestimated. We have thus used the water residence times calculated from freshwater replacement times. The only problematic estuary was Perhonjoki in which TP was overestimated by threefold. Perhonjoki is a relatively small, open system and TP was best predicted by Bowden's (1980) saltwater fraction method.

Given the relatively close agreement between observed and estimated values of TP in general, we estimated Chl as a function of TP estimated from the mass-balance, using the Finnish Chl–TP equation (Table 3). The MSR for Chl estimated from the mass-balance approach was 0.095 (Table 5; Fig. 4).

Estimating Chl from land use

No significant relationships existed between Chl and land-use variables for 17 estuaries (two of the 19 estuaries were missing morphometric data and thus could not be included in the land-use models). However, land-use models primarily capture non-point-source human influences, and of the 17 estuaries, seven had significant municipal or industrial point sources immediately on the shore of the estuary that were not included in the riverine nutrient load estimates. We thus temporarily removed these estuaries, hence referred to as point-source (PS) estuaries, from the data set. The remaining 10 estuaries, hence referred to as non-point-source (NPS) estuaries, then formed the core data set for the land-use regression modeling. A number of significant regression models were then generated. Of these models, the model estimating log Chl (lChl) as a function of log mean depth (lZm) and log percentage of the catchment forested (lFor-P) had the highest coefficient of determination and lowest model standard error. Both partial regression coefficients were also significant ($p < 0.05$). We then added the PS estuaries to the model to see which fit the same pattern. Of the seven PS estuaries, only Kokemäenjoki did not substantially decrease the model coefficient of determination or increase the model standard error when included. Moreover, its inclusion did not signifi-

Fig. 2. Regression equations for Chl as a function of TP and Chl as a function of TN for the 19 estuaries where r^2 is the coefficient of determination and SEE is the model standard error of the estimate.



cantly affect the regression coefficients. Given the small size of the data set available for the regression modeling, we included Kokemäenjoki in the regression model. The best land-use model is thus

(8) lChl = 5.44 – 0.96 lZm – 2.09 lFor-P

with n = 11, p = 0.005, r^2 = 0.74, and model standard error = 0.108. (Fig. 5). One must be cautious in estimating three parameters from a sample size of 11. However, both p values for the partial regression coefficients were significant (p = 0.0014 for lZm and p = 0.01 for lFor-P), and the coefficient of determination sums to more than the individual coefficients of determination (r^2 = 0.44 for lZm and r^2 = 0.01 for lFor-P). Thus, the model coefficients are likely robust.

Comparing the accuracy of Chl estimates from the mass-balance and land-use models

We compared the accuracy, precision, and bias of the Chl estimates generated via the mass-balance model and the land-use model for all the estuaries, the NPS estuaries, and the PS estuaries (with Kokemäenjoki included in both groups) (Table 5). When considering all the estuaries, the estimates of Chl via the mass-balance model were more accurate and precise and less biased than the estimates from the land-use model. Accuracy (MSR) was 0.095 when the mass-balance model was used as opposed to 0.143 with the land-use model. Precision (vSR) was 0.011 and 0.073, respectively, and bias (ME) was 0.023 and 0.203, respectively. The positive bias value for the land-use model suggests that it tends to underestimate observed Chl.

The most accurate, precise, and unbiased estimates are to be achieved by separating the estuaries into NPS and PS groups (Table 5). The land-use model most accurately and precisely estimates Chl for the NPS estuaries, with an MSR and vSR of 0.011 and 0, respectively. Bias is only 0.022. If the mass-balance is used to estimate Chl in the NPS estuaries, the MSR and vSR increase to 0.079 and 0.011, respectively, and bias increases to –0.11. This can only partially be attributed to the poor estimate of TP for Perhonjoki (for which it was difficult to estimate TP), as its removal only reduced the MSR to 0.68. The mosta ccurate, precise, and unbiased estimates of Chl for the PS estuaries were generated by the mass-balance model. The MSR is 0.107, which is substantially smaller than the MSR of 0.334 for the estimates based on the land-use model.

Discussion

Predicting coastal eutrophication in Finnish estuaries

The above analyses demonstrate the applicability of regression and P mass-balances to Finnish estuaries. We have shown that regression models based on single nutrients or land use can be fit to data from Finnish estuaries and that existing P mass-balance models such as that of Canfield and Bachmann (1981) accurately estimate TP. The approach did, however, require the division of the estuaries into two groups: those dominated by point-source loading and those dominated by non-point-source loading. Based on the relative goodness-of-fit, the analysis suggests the following guidelines for model choice in predicting Chl: (*i*) for estuaries dominated by non-point-source loading (ratio of PS load to total load <0.01), Chl is best estimated by the land-use regression model and (*ii*) for estuaries receiving point-source loading greater than 1% of the total load, Chl is best estimated by the mass-balance model. These guidelines and the land-use model itself must still be tested. Chl should be predicted for estuaries not included in this analysis using both the land-use regression and the mass-balance. The relative accuracy of the predictions can then be compared along with their correspondence to the above guidelines. To this end,

Table 3. Comparison of the Chl–TP and Chl–TN regression equations in Finnish estuaries and lakes.

	Equation	r^2	n
Finnish estuaries	lChl = –1.03 + 1.26 lTP	0.67	19
Lakes (OECD 1982)	lChl = –0.55 + 0.96 lTP	0.88	77
Lakes (Dillon and Rigler 1974)	lChl = –1.14 + 1.45 lTP	0.96	77
Finnish estuaries	lChl = –2.10 + 1.13 lTN	0.53	19
Lakes (Sakamoto 1966)	lChl = –2.5 + 1.4 lTN	nr	21
Lakes (Prairie et al. 1989)	lChl = –3.13 + 1.45 lTN	0.69	133

Note: All variables transformed to $\log_{10}$ (l). r^2, coefficient of determination; n, sample size; nr, not reported.

we have made the guidelines quantitative to reduce ambiguity as to which model should perform best.

It may be that the above guidelines reflect quirks in the present set of data rather than any general patterns in the applicability of the land-use model, particularly as it was developed on a small number of observations. In this case, the most conservative position would predict Chl via the mass-balance. Such predictions should still be within 10% of observed values and thus represent more predictive power than yet exists for Finnish estuaries. The mass-balance estimates are also true predictions in that an existing mass-balance model was applied to these data rather than fitting a new one.

It is admittedly dangerous to subdivide one's data until "accurate" predictions are attained. A division between PS and NPS estuaries does, however, seem reasonable. This division likely reflects the different sources and thus composition of TP in PS and NPS estuaries. In the PS estuaries, there are generally two sources of TP: point-source TP derived from municipal and industrial activities and, in one case, fish farms, and non-point-source TP derived primarily from agricultural activities in the catchment. In the NPS estuaries, the majority of TP is derived from agricultural activity (Rekolainen et al. 1995). TP from municipal wastes contains a greater proportion of bioavailable P than TP derived from agricultural sources (Ekholm 1994). In addition, the point sources are proximate to the estuaries compared with non-point-source TP, which is derived from the entire catchment. The combination of a greater proportion of bioavailable P and its rapid delivery to the estuary may explain why the mass-balance model performs better than the land-use model in the PS estuaries. The land-use model gives equal weight to all disturbance, regardless of proximity to the estuary, and thus emphasizes diffuse disturbance. The effect is also likely exaggerated in these estuaries, which have a relatively large catchment to surface area ratio (>70). This analysis indicates the importance of considering both the nature of the sources and their spatial distribution.

Coastal limnology: Finnish estuaries as salty lakes

Eutrophication tradition emphasizes the differences between lakes and estuaries (Richardson and Jørgensen 1996). Lakes and estuaries differ in water residence time, water chemistry, turbidity, grazing, morphometry, physical energy, and limiting nutrients. Whether these differences translate into differential eutrophication response to nutrients and anthropogenic disturbance is unclear, however. Finnish estuaries share characteristics of both lakes and estuaries: like lakes, they are nontidal, and like estuaries, they are open systems. They are, however, intermediate in terms of salinity. They thus provide an interesting test case for both limnological approaches and a test case of whether the eutrophication responses are quantitatively similar.

Our results suggest that mass-balance and regression approaches can in fact be applied to estuaries such as those in Finland. Our results also show that eutrophication responses in lakes and Finnish estuaries are quantitatively similar, at least with respect to Chl–TP equations (Table 3; Fig. 3) and the mass-balance equation. This is perhaps not surprising. In the case of the Chl–TP equation, we have to consider that phytoplankton have similar elemental composition and requirements in both systems (Hecky and Kilham 1988); thus, the Chl–TP yield should be similar. Water residence time and morphometry should have little effect on Chl–TP yields, as one would not expect differential dilution of these components. For instance, Basu and Pick (1996) developed regression models predicting Chl as a function of TP in rivers with residence times as short as 3 days. Water chemistry could affect the Chl–TP yield if the bioavailability of TP changed. However, both removal of P via flocculation at low salinity (2–3 ppt) (Howarth et al. 1995) and release of P through competition for sorption sites when river and seawater meet still allow for corresponding decreases or increases in Chl.

Of the cited differences between freshwater and coastal systems, turbidity, herbivory, and P versus N limitation may change the Chl–TP yield. Light limitation reduces the yield of Chl as phytoplankton are unable to take advantage of available P (Fisher et al. 1995). However, light limitation occurs primarily in permanent turbidity maxima that, to our knowledge, have not been documented in Finnish estuaries. Herbivory has also been shown to decrease Chl yields (Mellina et al. 1995; Meeuwig et al. 1998). The zoobenthos of Finnish estuaries includes phytoplanktivores such as *Macoma balthica* and *Mytilus edulis* (Lax et al. 1993). However, although abundance of bivalves has increased in general in the Baltic (Cederwall and Elmgren 1980), inshore areas show declines due to pollution and hypoxia (Mattila 1993). The Chl–TP yield also changes as a function of P versus N limitation. For instance, Prairie et al. (1989) demonstrated systematic changes in the regression coefficients with changing N:P ratios that they interpreted as indicating changes in P versus N limitation. However, although the outer waters of the Baltic are thought to be primarily N limited (Wulff et al. 1990), the estuaries are considered to be primarily P limited (Pitkänen and Tamminen 1994), thus suggesting that the yields should be similar to those seen in P-limited lakes.

It is not surprising that a lake-derived, P-based mass-balance model can be applied to these estuaries. First, we do not really know whether estuaries are N or P limited; N versus P limitation in estuaries remains an issue of some contention, despite the dogma of N limitation. Hecky and Kilham (1988) reviewed the evidence for N versus P limitation in coastal waters; they argued that the evidence for N limitation at an ecosystem level was weak, given the lack of large-scale experiments and comparative data. These gaps in empirical or experimental evidence remain. Further, Elser et

Fig. 3. Frequency histogram of slopes and intercepts for Chl–TP equations in the literature (J.J. Meeuwig, unpublished data).

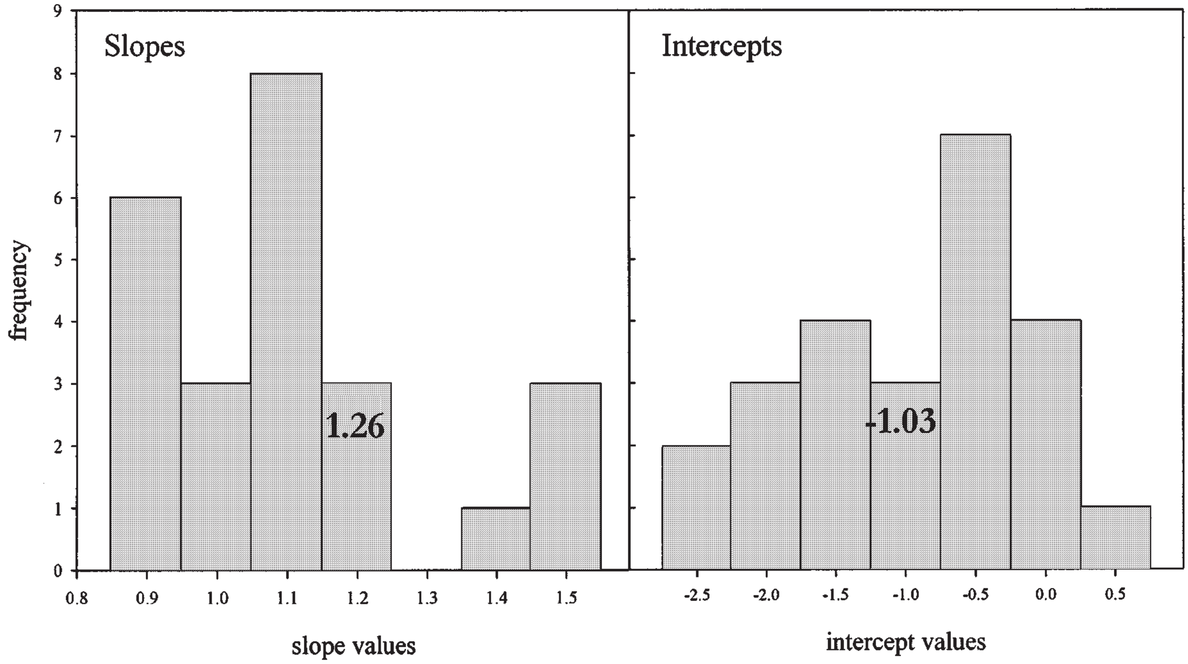


Table 4. Comparison of observed TP to TP calculated using water residence times calculated as freshwater replacement time (TP-FW, Rt-FW) and via Bowden's (1980) saltwater fraction method (TP-B, Rt-B).

Estuary code	Observed TP (mg·m^{-3})	TP-FW (mg·m^{-3})	TP-B (mg·m^{-3})	Rt-FW (months)	Rt-B (months)
11	46.9	32.4	9.7	14.71	2.05
12	46.8	39.1	6.8	6.16	0.57
13	35.3	58.1	5.5	0.66	0.05
14	39.3	22.0	9.9	0.29	0.12
17	28.0				
18	66.6	44.5	18.0	16.81	2.85
19	36.0	33.0	9.7	26.65	2.81
20	59.9	51.1	17.0	1.33	0.33
21	91.2	53.3		3.40	
23	23.0	19.7	15.8	18.23	11.70
26	65.9	39.4	28.7	42.73	14.46
27	28.2	32.0	11.0	98.10	6.04
29	33.7	39.5	9.2	31.76	2.05
30	34.9	33.5	16.5	50.51	7.98
35	39.1	49.7	34.5*	0.15	0.10
39	20.0	31.1	3.1	7.89	0.40
42	92.2				
49	20.2	71.5	17.4*	0.48	0.09
58	37.8	34.1	15.6	8.36	2.34

Note: Estuaries where TP is better estimated via Bowden's equation are indicated with an asterisk.

al. (1990) reviewed the evidence for single nutrient limitation in whole-lake experiments and argued that colimitation was more likely the norm at the scale of whole ecosystems. Indeed, we have argued that this is the case in lakes (Meeuwig and Peters 1996) and eastern Canadian estuaries (Meeuwig 1999) and where land-use models account for a greater proportion of the variance. Finally, as N and P covary, P can be viewed as a surrogate for N if the idea of P or colimitation remains too uncomfortable.

The mass-balance also likely works because differences between lakes and estuaries are incorporated into the model via variables such as water residence time and P sedimentation. Differences in physical energy and morphometry that result in shorter residence times in coastal systems are addressed by the water residence time. When the estuaries were treated as lakes and residence time calculated only from river freshwater load, estimated log TP values were within 9.7% of observed log TP values on average. This sug-

Table 5. Standardized goodness-of-fit criteria for accuracy (mean squared residual, MSR), precision (variance of the squared residuals, vSR), and bias (mean error, ME) for the mass-balance and land-use regression models applied to all estuaries, non-point-source dominated estuaries (NPS), and estuaries with point-source loads (PS).

Criterion	Model type	All (n = 17)	NPS (n = 11)	PS (n = 7)
Accuracy (MSR)	Mass-balance	0.095	0.079	0.107
	Land use	0.143	0.011	0.334
Precision (vSR)	Mass-balance	0.011	0.011	0.012
	Land use	0.073	0.000	0.123
Bias (ME)	Mass-balance	0.023	–0.110	0.275
	Land use	0.203	0.022	0.486

Note: Kokemäenjoki is included in both the NPS and PS categories. All values are expressed as percentages.

Fig. 4. Observed versus predicted values for Chl estimated by the mass-balance model for the Finnish estuaries dominated by point-source loading.

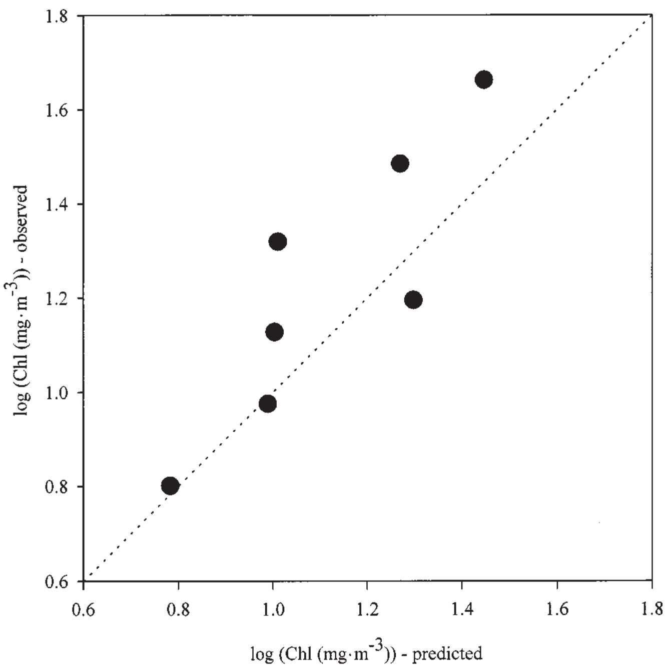
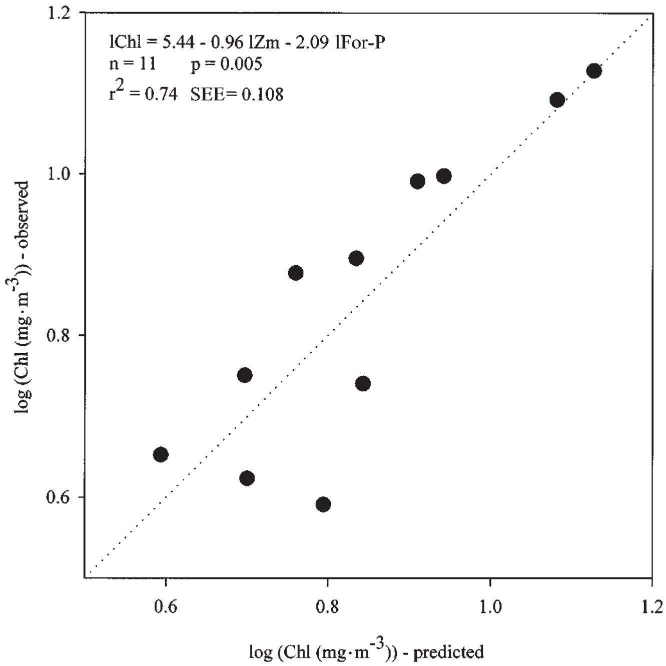


Fig. 5. Observed versus predicted values for Chl as a function of log mean depth (lZm) and log percentage forest (lFor-P) for the Finnish estuaries dominated by non-point-source loading where r^2 is the coefficient of determination and SEE is the model standard error of the estimate.



gests that we can assume zero net inflow of Baltic water. This is perhaps not surprising, as some of the estuaries have very restricted exchange with the open Baltic due to the complex coastal morphometry and the presence of islands. However, even for the relatively open pocket estuaries such as Temmesjoki, estimated TP was within 3% of observed TP (on a log scale). Our results are consistent with those of Nixon et al. (1996) who were also able to effectively use freshwater replacement times as a substitute for water residence times in their mass-balance equations for major estuaries of the North Atlantic.

Differences in salinity are thought to affect P sedimentation: salinity-driven flocculation, as described by Howarth et al. (1995), should increase P sedimentation, while release of P due to competition for sorption sites should decrease sedimentation. The former should be important in Finnish estuaries, as runoff is relatively high in humic substances and iron (S. Rekolainen, Finnish Environment Institute, personal communication). Competition for sorption sites should also be important, as even the low-salinity waters of these estuaries represent a large increase in ionic concentration. It is unclear, however, what the outcome of these two opposing processes is in terms of the sedimentation rate. The slight bias in the mass-balance calculations is positive (0.64%), suggesting that the mass-balance equation slightly underestimates TP. Morphometry may affect P sedimentation, as estuaries tend to be shallower than lakes (Nixon 1988), and, for a given surface area, shallower systems are more vulnerable to resuspension of P sediments. However, the mass-balances include a depth term, so this difference should also be incorporated into the model. The accuracy of the lake-derived, P-based mass-balance model suggests that important differences in P sedimentation rate cannot be identified. This result is consistent with that of Nixon et al. (1996) who, in the absence of estimates for nutrient sedimentation in estuaries, borrowed lentic estimates and demonstrated that lakes and estuaries show similar patterns between net transport of TN and TP and water residence times, consistent with the mass-balance calculations.

Summary

A combination of land-use models and mass-balance models accurately estimates Chl in Finnish estuaries. An average degree of accuracy for the land-use models (e.g., MSR = 0.014) would estimate Chl in Perhonjoki as 9.3 mg·m^{-3} where the observed value is 7.5 mg·m^{-3}. An average degree of accuracy (e.g., MSR = 0.04) for the mass-balance model would estimate Chl in Karjaanjoki as

4 mg·m^{-3} where the observed value is 6 mg·m^{-3}. This degree of accuracy suggests that the mass-balance approach and other limnological models such as the land-use regression can effectively estimate coastal eutrophication in Finnish estuaries. It remains, however, to test the predictive power of these models on estuaries not used in their development.

This analysis also demonstrates that Finnish estuaries and lakes respond similarly to total nutrients and nutrient loads. Finnish estuaries are not typical estuaries, however, as they are essentially nontidal and have lower salinity than most estuaries. It thus remains to be demonstrated whether the mass-balance approach can be effectively applied to the estuaries of North America and Atlantic Europe.

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