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Predicting oxygen in small estuaries of the Baltic Sea: a comparative approach

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Abstract

Coastal eutrophication, manifested as hypoxia and anoxia, is a global problem. Only a few empirical models, however, exist to predict bottom oxygen concentration and percentage saturation from nutrient load or morphometry in coastal waters, which are successfully used to predict phytoplankton biomass both in lakes and in estuaries. Furthermore, hardly any empirical models exist to predict bottom oxygen from land-use. A data set was compiled for 19 estuaries in the northern Baltic Sea, which included oxygen concentration and percentage saturation, water chemistry, estuary morphometry, and land-use characteristics. In regression analyses, bottom oxygen was predicted both as a function of the percentage of watershed under agriculture and of mean depth. These models accounted for ca. 55% of the variation in oxygen. Additionally, oxygen was linked to fetch (diameter of the area in the direction of the prevailing wind), which accounted for 30% of the variation in oxygen. This suggests that shallow Finnish estuaries are wind-sensitive. In 'pits' (sub-thermocline waters of deep basins), near-bottom total nitrogen strongly correlated with oxygen percentage saturation ($R^2 = 0.81$). Neither chlorophyll *a*, total phosphorus nor nutrient loading explained oxygen variation in entire estuaries or in 'pits', probably mainly due to annual sedimentation/sediment–water flux dynamics. On the basis of the results of cross-validation, the models have general applicability among Finnish estuaries.

Keywords: oxygen; land-use; morphometry; nitrogen; chlorophyll *a*; regression; estuary; Baltic Sea

1. Introduction

Anoxia and hypoxia in marine waters are the two manifestations of eutrophication, and are problematic both globally (Diaz, 2000; Nixon, 1995; NOAA, 1997; Vollenweider, Marchetti, & Viviani, 1992) and in the open and coastal Baltic Sea (Bonsdorff, Blomqvist, Mattila, & Norkko, 1997; Cederwall & Elmgren, 1990; Fonselius, 1969; Larsson, Elmgren, & Wulff, 1985; Pitkänen, Lehtoranta, & Räike, 2001). Oxygen depletion in bottom waters is generally associated with an increase in the production of organic matter owing to nutrient enrichment. The decay of organic material consumes oxygen, causing oxygen concentrations to decrease, especially during the stratified season when the thermocline deepens and becomes stronger. Anoxia and hypoxia are problematic because they can lead to increased mortality of macro-benthic fauna and fish (Baden, Loo, Pihl, & Rosenberg, 1990; Norkko & Bonsdorff, 1996), and to the production of hydrogen sulfide on the sea bottom (Fonselius, 1969; Jørgensen, Bang, & Blackburn, 1990). Moreover, the release of phosphorus from sediment to the water column during oxygen deficit at the sediment/water interface can first lead to increased phytoplankton biomass in the productive surface water layer, followed by increased sedimentation, and ultimately to an internal loading of the system (Jensen, Mortensen, Andersen, Rasmussen, & Jensen, 1995; Jørgensen, 1996). Management tools to predict oxygen levels are, therefore, needed for controlling water pollution.

An empirical approach has provided a successful basis for predicting oxygen conditions in lakes (Vollenweider &

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Janus, 1982). Methods used for predicting oxygen changes in the hypolimnion include areal and volumetric oxygen-deficit rates (Cornett, 1989; Cornett & Rigler, 1979, 1980), oxygen consumption (Charlton, 1980), oxygen concentration (Molot, Dillon, Clark, & Neary, 1992), and oxygen percentage saturation (Crisman, Chapman, & Chapman, 1998). Oxygen in lakes is linked to several variables including morphometry (Charlton, 1980; Cornett & Rigler, 1980), total phosphorus (TP) loading (Welch & Perkins, 1979), TP concentration (Molot et al., 1992), TP retention (Cornett, 1989), temperature (Cornett & Rigler, 1980), productivity (Charlton, 1980), Secchi depth (Cornett & Rigler, 1980), maximum oxygen storage (Trimbee & Prepas, 1988), and initial oxygen concentration at spring turnover (Molot et al., 1992).

Research on oxygen in coastal waters has generally taken the approach of dynamic simulation modelling. Studies on this topic include models simulating spatial and temporal variations of nutrients and oxygen (Kuo, Park, & Moustafa, 1991; Malmgren-Hanse, Mortensen, & Møller, 1984; Stigebrandt & Wulff, 1987), and models predicting oxygen in various loading conditions (Hansen, Ærtebjerg, & Richardson, 1995). The wider use of these models is, however, limited because they are complex, site-specific, and expensive to develop.

Comparative approaches have been used in estuaries. These studies include the models of Boyton, Kemp, and Keefe (1982) and Boyton, Murray, Hagy, Stokes, and Kemp (1996) predicting the phytoplankton chlorophyll *a* (Chl) as a function of total nitrogen (TN) and TP load, the model of Gowen, Tett, and Jones (1992) predicting Chl as a function of TN concentration, models of Meeuwig's (1999) and Meeuwig, Kauppila, and Pitkänen (2000) predicting phytoplankton biomass as a function of land-use, and model of Nixon (1981) predicting benthic remineralization. There are, however, only a few empirical approaches to predict oxygen concentration in coastal waters. Empirical data analyses used for simulation studies include the model of Aure and Stigebrandt (1989), which predicted the volumetric mean rate of oxygen consumption (CONS) in fjords and the model of Wallin and Håkanson (1991), which predicted the mean summer deep water oxygen in coastal inlets, both the models were developed as a function of morphometry. Pure empirical studies include Moehlenberg's (1999) model predicting oxygen as a function of meteorology and nutrient load.

In this study, a comparative approach is applied in predicting oxygen concentration and percentage saturation in the near-bottom waters of Finnish estuaries. A data set was compiled for 19 estuaries to test the hypotheses that: (1) bottom oxygen concentration and percentage saturation, similar to phytoplankton biomass (Meeuwig, 1999; Meeuwig et al., 2000), can be predicted from land-use; and (2) oxygen levels in estuaries are coupled with freshwater discharge, water circulation, and wind effect, as suggested by Kuo et al. (1991) and Stanley and Nixon (1992).

2. Methods

2.1. Data set

Data were compiled for 19 small estuaries located on the Baltic coast of Finland, from the northeastern Bothnian Bay to the eastern Gulf of Finland (Fig. 1). The data originated from Finnish national water quality monitoring surveys, the database being maintained by the Finnish Environment Institute (SYKE). The data sets include information on bottom oxygen concentrations (DO, mg l^{-1}) and percentage saturation, phytoplankton biomass (as Chl), water chemistry, coastal morphometry, land-use, and total nutrient loads (Tables 1 and 2). Seasonal means were estimated for the period July to September during 1989–1993 (Table 3). Large estuaries ($n = 1$) with only one deep sampling station and large estuaries sampled only in deep stations ($n = 2$) were excluded from the data set. One estuary receiving organic load (COD_{MN}) from the pulp and paper industry was also excluded, because its sampling stations were near the outflow pipe. Excluding season, some extra variables, and small changes in the selection of estuaries, the data set is similar to that described by Meeuwig et al. (2000).

Estuarial means from near-bottom waters (1 m above the bottom) ranging from the depth 2 to 47 m were calculated following Meeuwig et al. (2000) in order to incorporate interannual variability. In this approach, growing season averages were calculated, using oxygen DO as an example, as follows:

1. Oxygen $DO_{x,y}$ is the mean oxygen DO for year x and station y, using the values of oxygen DO from July to September of year x.
2. Oxygen $DO_{x,z}$ is the mean oxygen DO for year x, using the values of oxygen $DO_{x,y}$ for all the sampling stations located in estuary z.
3. Oxygen DO_z is the mean oxygen DO for estuary z, using the values of oxygen $DO_{x,z}$ for all the sampling years for estuary z.

Mean oxygen concentration and percentage saturation in near-bottom waters were also estimated in the shallow parts of the estuaries (2–15 m deep). In the case of 'pits', the sub-thermocline waters of the deep basins permanently stratified during summer (total depths from 20 to 47 m), oxygen values in depths 20, 40, and 1 m above the bottom were averaged.

Oxygen concentration and oxygen percentage saturation were determined by the Winkler method (Grasshoff, Erhardt, & Kremling 1983). TP and TN were

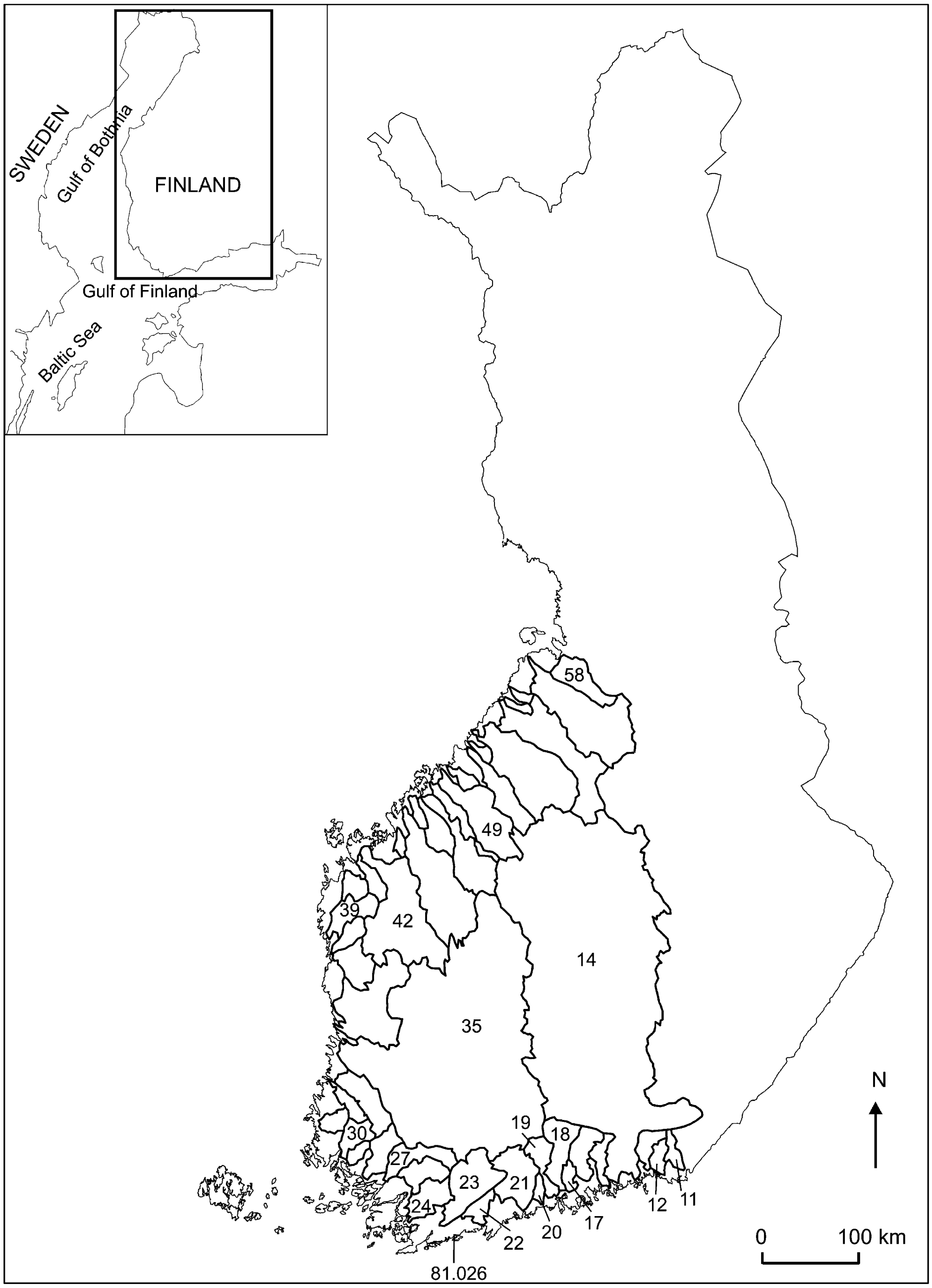

Fig. 1. Map of Finland indicating the locations (by code) of the estuaries included in this study: 11, Virojoki; 12, Vehkajoki; 14, Kymijoki; 17, Ilolanjoki; 18, Porvoonjoki; 19, Mustijoki; 20, Sipoonjoki; 21, Vantaanjoki; 22, Siuntionjoki; 23, Karjaanjoki; 24, Kiskonjoki; 27, Paimionjoki; 30, Laajoki; 35, Kokemäenjoki; 39, Närpiönjoki; 42, Kyrönjoki; 49, Perhonjoki; 58, Temmesjoki; and 81.026, Fagerviken.

analysed from unfiltered samples according to Finnish standard methods (Koroleff, 1976). Chl was analysed after filtering (Whatman GF/C) from integrated water samples (surface to 5 m) according to Lorenzen (1967). Salinity was calculated using the Practical Salinity Scale (UNESCO, 1985). Annual material inputs from rivers were calculated by multiplying the monthly flows by the mean monthly concentrations (Pitkänen, 1994).

Estuarial morphometry was characterized by mean depth, surface area, volume, residence time, openness, and fetch. Theoretical residence time was calculated by Knudsen's equation (e.g. Bowden, 1980). Openness in the mouth of the estuary was calculated by the relationship between the diameter of water surface area and that of the whole area including islands. Fetch is the measure of the longest diameter of the water area in the direction of the prevailing wind, and it has previously been used by e.g. Cattaneo (1990) and Oldham and Lavery (1999). Prevailing wind directions, calculated by the Finnish Meteorological Institute (1990–1995), were

Table 1
Descriptive data for the estuaries

Estuary number	Oxy ($mg\,l^{-1}$)	Ox_{sat} (%)	Chl ($mg\,l^{-1}$)	TP ($mg\,m^{-3}$)	TN ($mg\,m^{-3}$)	Sec (m)	Temp (°C)	Sal	Z_{mn} (m)	Z_{max} (m)	A_o (km^2)
11	9.3	92.1	16.6	43.1	546	2	–	3.7	4.4	9.1	32.6
12	8.2	81.3	6.9	43.9	468	1.2	17.9	3.6	6	14.7	10
14	6.7	72	11.8	56.1	482	1.8	14.5	3.8	4.9	14	51.8
17	6	73.3	7	–	–	2.4	15.9	4.9	–	18	–
18	7.8	78.8	27.2	64.7	576	1.4	12.9	4.7	12.3	35	48.8
19	8.4	79.9	9.7	38.1	388	2	11.9	5.3	11.8	42	35.9
20	7.4	76.5	14.8	55.8	653	0.9	14.3	4.3	3.8	12	2
21	8.8	95	31.3	90	1338	0.6	16.6	2.8	4	6.3	5.5
22	7.7	76.9	6.4	35.3	327	2.6	11.1	5.7	6.2	22	19.9
23	6.5	65.7	7.7	25.4	498	2.8	13.5	3	12.2	37	45.1
24	5.9	52.3	6.2	35.6	450	1.5	7.4	5.8	17.2	53	84.4
27	6	59	4.5	35.7	441	2.2	12.4	6.2	17.9	49	102
30	8.8	92	7.7	29.3	460	1	16.4	5.6	4.6	11	82.6
35	7.7	80	13.1	27.4	403	1.6	16.4	3.4	3.1	6.5	31.4
39	9	92	2.9	19.4	297	2.4	13.3	5.7	6.4	18	34.2
42	8.1	82	20.5	90.9	1403	0.7	17.8	2.1	–	7.5	–
49	10	94.3	5.6	18.8	447	1.9	13.3	3.2	4.4	9.2	6.2
58	9.6	96.7	10	27.6	431	1	15.5	1.9	3.1	4.7	86.2
81.026	7.7	74.2	5.6	36	380	2.2	11.1	5.7	8.6	22	5.3
Mean	7.9	80.1	11.3	42.2	559	1.7	14.0	4.3	7.9	20.9	39.3
SD	1.2	12.5	8.0	21.6	318	0.7	2.8	1.4	4.9	15.5	33.1
Min	5.9	52.3	2.9	18.8	297	0.6	7.4	1.9	3.1	4.7	2
Max	10	96.7	31.3	90.9	1403	2.8	17.9	6.2	17.9	53	144.7

Estuary number	Vol ($10^6\,m^3$)	Q_r ($m^3\,s^{-2}$)	Res year	Urb (%)	Ag (%)	For (%)	Wshed (km^2)	P_{den} (%)	P_{load} ($t\,year^{-1}$)	N_{load} ($t\,year^{-1}$)	Fetch (km)	Open
11	144.1	3.7	1.23	0.5	13.5	82.9	357	9.6	8.7	181	3.5	0.41
12	59.8	3.6	0.51	0.8	14	80.7	380	28.3	9.4	232	2.4	1
14	251.5	172	0.02	3.8	22.8	70.1	37 159	80.4	145	3685	7.5	–
17	–	2.9	–	0.7	23.1	73.4	309	36	7.4	184	4.6	0.6
18	600.5	13.6	1.4	2.4	28.5	67.9	1273	64.7	63.4	1683	4.3	1
19	425.2	6.1	2.22	1.3	26.8	70.7	783	30.5	22.2	559	5.7	0.76
20	7.5	2.1	0.11	2.5	32.3	64.8	220	46.9	5.3	131	1.8	0.55
21	20	16.4	0.06	6.7	23.6	67.7	1686	265.1	53.4	1453	3.3	0.16
22	104.5	4.8	0.07	2.5	28	67	487	–	12.9	343	7.2	0.33
23	550.2	20.1	0.24	2.8	29.9	65.3	2046	37.8	28.1	728	6.3	0.3
24	1451.7	10.3	0.8	2.4	32	64	1047	–	18.6	393	17.8	0.22
27	1826	10.1	5.62	1.2	43	54.3	1088	17.7	80.8	952	18.8	0.63
30	376.3	6.6	4.21	1.2	19	78.6	685	14	36	537	5.7	0.67
35	97.7	255.6	0.01	1.4	27	67.7	27 046	26.4	466.6	10 770	11.6	0.21
39	218.8	10.1	0.66	0.7	20.6	78	992	–	21.1	369	6.8	0.9
42	–	53.2	–	1	23.3	74.3	4923	20	163.2	3215	–	1
49	27	21.3	0.04	0.3	9.5	87.2	2524	11.7	66	828	1.9	1
58	265.5	12.1	0.7	0.4	15.4	83.6	1181	9	28.1	431	4.2	1
81.026	45.6	0.7	0.18	–	–	–	70	–	1.8	49	1.4	0.85
Mean	388.8	25.2	1.1	1.7	24.1	72.2	2616.5	44.1	60.7	1280	6.3	0.6
SD	527.6	58.7	1.6	1.5	8.3	8.7	6202	65.5	108.5	2488	5.2	0.3
Min	7.5	0.7	0.01	0.3	9.5	54.3	70	9	1.8	49	1.4	0.16
Max	1826	255.6	5.62	6.7	43	87.2	37 159	265.1	466.6	10 770	18.8	1

Note: Oxy, oxygen concentration; Ox_{sat}, oxygen percentage saturation; TP and TN, TP and TN concentrations in bottom water layers; Chl, chlorophyll *a*; Sec, Secchi depth; Temp, bottom water temperature; Sal, bottom water salinity; Z_{mn}, mean depth; Z_{max}, maximum depth; A_o, surface area; Vol, volume; Q_r, river water loading; Res, water residence time; Urb, Ag, For, percentage of watershed that is urban, agricultural, or forested, respectively; Wshed, watershed area; P_{den}, human population density; P_{load} and N_{load}, P and N loads, respectively; Fetch, the longest diameter of the estuary in the direction of main wind; Open, openness.

based on measurements at 11 meteorological stations close to the estuaries. Descriptions of the other morphometric variables can be found in Meeuwig et al. (2000).

2.2. Statistical analyses

Multiple regression techniques were used to consider relationships between oxygen DO and percentage

Table 2
Mean values of chemical and physical variables

Estuary number	Oxy ($mg\,l^{-1}$)	Ox_{sat} (%)	Chl ($mg\,l^{-1}$)	TP ($mg\,m^{-3}$)	TN ($mg\,m^{-3}$)	Sal
(a) In 'pits', the sub-thermocline waters of the deep basins permanently stratified during summer						
18	6.6	61	10.4	54.7	404	5.5
19	8	73.9	10.5	39.2	324	5.6
22	7.5	71	5.5	28.8	316	6
23	3.9	32.4	7.7	19.3	542	4.5
24	5.6	46	4	35.1	450	5.7
27	5.7	52.6	3.8	32	410	6.1
81.026	7.5	66.9	–	35.6	421	5.9
Mean	6.4	57.7	7.0	35.0	410	5.6
SD	1.4	14.9	3.0	10.8	77	0.5
Min	3.9	32.4	3.8	19.3	316	4.5
Max	8	73.9	10.5	54.7	542	6.1
(b) In the shallow parts of the estuaries in near-bottom waters						
11	9.3	92.1	16.6	43.1	546	1.8
12	8.2	81.3	6.9	43.9	468	0.1
14	9.4	77.3	9.6	47.1	488	2.1
18	8.5	90	30.4	65.5	614	2.8
19	8.6	89.2	8.4	36.8	424	3.7
20	7.4	76.5	14.8	55.8	653	1.3
21	8.8	95	31.3	90	1338	0.1
22	7.8	78.9	6.5	36.8	329	6.5
23	7.6	79.4	7.5	26.8	483	5.1
27	7.7	85	6	40	500	5.7
30	9	94	10	29.3	460	0.1
35	7.7	80.9	10.6	29.5	432	1.0
39	9.2	101	–	19	320	1.1
42	8	82.2	–	90.9	1402	0
49	10	94.3	5.6	18.8	448	1.3
58	9.6	96.7	10	27.6	431	0
81.026	7.7	81.5	5.6	35.7	373	2.2
Mean	8.5	86.8	12.0	43.3	571	2.0
SD	0.8	7.7	8.3	21.4	313	2.1
Min	6.7	68.3	5.6	18.8	320	0
Max	10	101	30.4	90.9	1402	6.5

saturation, and Chl, nutrients, land-use, and morphometry. All the variables were log-transformed to stabilize variance (Zar, 1996). Land-use variables that were calculated as percentages of the watershed were transformed as log $10(x+1)$ in order to avoid problems due to the presence of zeros in some of the land-use categories. The set of preliminary models for regression was identified using the R-square option in the SAS procedure 'Proc Reg'. The 'best' model was chosen on the basis of the highest coefficient of determination (R^2) and the smallest standard error of the estimate (SEE). Correlation analyses were performed in order to avoid collinearity between independent variables.

In model validation, adjusted R-square (Eqs. (1) and (2)) and the predicted residual of the sums of squares (PRESS) statistics (Eq. (3)) were used to estimate general applicability of the model because there was insufficient data for random splitting (Stevens, 1996). In order to estimate the amount of shrinkage found in the coefficient of determination (R^2), two kinds of formulae, measuring different attributes, were used. The formula of Wherry estimates that how much variance in y would be accounted for the prediction equation that had been derived in the population from which the sample was drawn. This is the adjusted R^2 printed out in the SAS procedure, and it is given by

Table 3
Distribution of sampling effort for water chemistry data, where years is the number of years sampled, station 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 from July to September, averaged for all stations in the estuary

Estuary number	Years	Station					Season		
		1989	1990	1991	1992	1993	Mean	Min	Max
11	5	7	8	7	8	9	1.7	1	2
12	5	4	4	4	4	5	1.1	1	3
14	5	1	2	2	2	2	2.1	1	4
17	4	1	1	1	1	1	1	1	1
18	5	9	10	9	9	7	1.4	1	4
19	5	10	11	11	11	8	2.5	1	5
20	5	4	3	3	3	4	2.3	1	3
21	5	1	1	1	1	1	1	1	1
22	5	9	9	9	9	9	1.2	1	2
23	5	9	9	9	9	9	1.4	1	3
24	5	3	2	3	2	3	1.6	1	3
27	5	4	4	4	4	4	2.1	1	3
30	5	3	3	3	3	3	1.6	1	3
35	5	6	6	6	6	6	1.2	1	2
39	5	2	3	3	3	3	1.8	1	3
42	5	1	1	1	1	1	1	1	1
49	5	1	2	2	2	2	1.6	1	2
58	5	5	5	5	5	4	2.9	1	4
81.026	5	5	5	7	6	4	1.2	1	2

$$\hat{\rho}^2 = 1 - (n-1)/(n-k-1)(1-R^2) \qquad (1)$$

where $\hat{\rho}$ is the estimate of ρ, the population multiple correlation coefficient, n is sample size, and k is the number of predictors. The formula according to Stein (1960) estimates average cross-validation predictive power by indicating how well the derived equation will predict using other samples from the same population. It is given by

$$\hat{\rho}_c^2 = 1 - ((n-1)/(n-k-1))((n-2)/(n-k-2)) \times ((n+1)/n)(1-R^2). \qquad (2)$$

The PRESS statistics involve an estimate of average predictive power using other samples. In this approach, the y value for each subject is set aside, and the prediction equation is derived using the remaining data (Stevens, 1996). Thus, n prediction equations are derived and n honest prediction errors are found. We used the SAS REG program to print out PRESS. The PRESS residual for each subject is given by

$$\hat{e}_{(-i)} = y_i - \hat{y}_{(-i)} \qquad (3)$$

where y_i is the individual sample and $\hat{y}_{(-i)}$ is the predicted value for subject i, where that subject was not used in developing the prediction equation. Then the PRESS sum of squared residuals is given by

$$\mathrm{PRESS} = \sum \hat{e}^2_{(-i)} \quad (4)$$

The PRESS value was then used to calculate an R^2-like statistics that reflect the general applicability of the model. It is given by

$$R^2_{\mathrm{PRESS}} = 1 - (\mathrm{PRESS}) \Big/ \sum (y_i - \bar{y})^2 \quad (5)$$

where y_i is the individual sample and $\bar{y}$ is the mean of the samples.

3. Results

Estuaries were highly variable with regard to a number of characteristics (Table 1). Oxygen DO and percentage saturation in near-bottom waters ranged from 5.9 to 10 mg l^{-1} and from 52 to 97%, respectively, with bottom temperature ranging from 7.4 to 17.9 °C. Phytoplankton biomass (measured as Chl) ranged from 2.6 to 31.3 mg m^{-3}, and nutrient concentrations near the bottom varied from 18.8 to 91 mg P m^{-3} and from 297 to 1403 mg N m^{-3}. The nutrient loads received by estuaries ranged from 1.8 to 467 t TP year^{-1} and from 49 to 10 770 t TN year^{-1}. Land-use varied between watersheds characterized by high agricultural activity (field percentage up to 43%), high percentage of forestry (up to 87%), and high population density (up to 265 km^{-2}). Coastal morphometry varied between well-mixed and stratified estuaries, the mean depth ranging from 3.1 to 17.9 m. The estuaries were mainly more or less enclosed by islands and shallows or by sills. The wind effect on mixing conditions, measured as fetch, spanned one order of magnitude. Theoretical residence time ranged from 0.01 to 5.62 years.

3.1. Oxygen models

Bottom oxygen DO and percentage saturation in Finnish estuaries could be explained by both land-use and morphometric variables (Eqs. (6)–(11) in Table 4). The best predictor for oxygen DO was the percentage of watershed under agriculture, which accounted for 54% of the variation in oxygen (Eq. (6), Fig. 2). Alternatively, oxygen DO could also be explained by morphometric variables, mean depth accounting for 36% (Eq. (7)) and fetch for 29% (Eq. (8)) of the variation in oxygen. With regard to oxygen percentage saturation, land-use and morphometry were comparable: mean depth accounted for 58% (Eq. (9), Fig. 3) and the percentage of watershed under agriculture for 50% (Eq. (10)) of the variation in oxygen. Furthermore, fetch accounted for 30% (Eq. (11)) of the variation in oxygen. Neither Chl, TP nor TN explained variation in oxygen DO or percentage saturation. No multiple regressions combining land-use and morphometric variables predicted DO or oxygen percentage saturation.

The regression equations for predicting oxygen DO and percentage saturation were significant or highly significant (Eqs. (6), (7), (9), and (10) in Table 4). Some curvilinearity was, however, identified between oxygen and the percentage of watershed under agriculture (Eqs. (6) and (10), Fig. 2). Square root transformation did not improve the predictions, but resulted in increased instability in variance. Neither did a quadratic term improve the distribution. As curvilinearity was mainly due to three semi-enclosed estuaries with local deep basins (estuaries 23, 24, and 27), regression was also run for the

Table 4
Summary statistics for models of oxygen concentration, percentage saturation, and bottom nitrogen concentrations for the entire estuaries, the 'pits', and the shallow areas of estuaries (2–15 m)

Model (equation number)		n	R^2	p	SEE	B_0	B_1
Entire estuaries							
$\log \mathrm{Oxy} = B_0 + B_1 \log \mathrm{Ag}$	(6)	18	0.54	0.0005	0.040	1.37***	−0.35***
$\log \mathrm{Oxy} = B_0 + B_1 \log Z_{\mathrm{mn}}$	(7)	17	0.36	0.0105	0.056	1.04***	−0.18**
$\log \mathrm{Oxy} = B_0 + B_1 \log \mathrm{Fetch}$	(8)	18	0.29	0.0243	0.064	0.97***	−0.12*
$\log \mathrm{Ox}_{\mathrm{sat}} = B_0 + B_1 \log Z_{\mathrm{mn}}$	(9)	17	0.58	0.0004	0.039	2.08***	−0.22***
$\log \mathrm{Ox}_{\mathrm{sat}} = B_0 + B_1 \log \mathrm{Ag}$	(10)	18	0.51	0.0009	0.045	2.37***	−0.34**
$\log \mathrm{Ox}_{\mathrm{sat}} = B_0 + B_1 \log \mathrm{Fetch}$	(11)	18	0.30	0.0170	0.064	1.98***	−0.13*
Pits: hypolimnion							
$\log \mathrm{TN}_{\mathrm{B}} = B_0 + B_1 \log \mathrm{Oxy}$	(12)	7	0.76	0.0105	0.057	7.19***	−0.65**
$\log \mathrm{TN}_{\mathrm{B}} = B_0 + B_1 \log \mathrm{Ox}_{\mathrm{sat}}$	(13)	7	0.81	0.0057	0.040	8.31***	−0.57**
Shallow parts of the estuaries							
$\log \mathrm{Oxy} = B_0 + B_1 \log \mathrm{Ag}$	(14)	16	0.76	0.0001	0.013	1.27***	−0.25***

Symbols for variables are as described in Table 1. $\mathrm{TN_B}$, TN concentration in bottom water layer; n, sample size; R^2, coefficient of determination; p, probability of the model; SEE, standard error of the estimate; B_0, intercept; $B_1 - B_2$, partial regression coefficients. The asterisks are the levels of significance ($0.01 < p_{\mathrm{obs}} < 0.05$, $0.001 < p_{\mathrm{obs}} < 0.01$, $p_{\mathrm{obs}} < 0.001$).

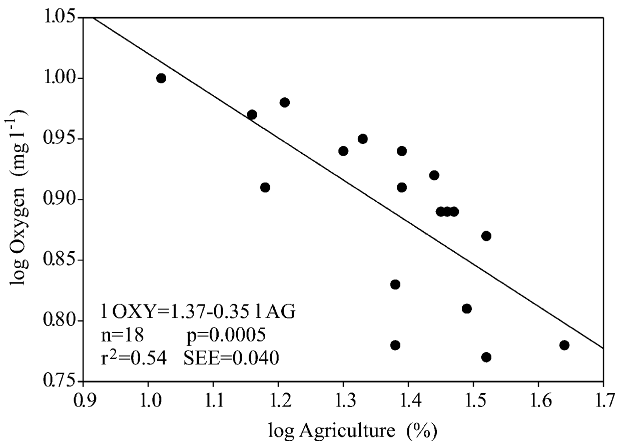

Fig. 2. Regression equation for bottom oxygen ($mg\,l^{-1}$) as a function of the watershed under agriculture for 18 estuaries where R^2 is the coefficient of determination and SEE is the model standard error of the estimate.

entire estuaries having these 'pits' (seven estuaries). No relationships were found between oxygen and the percentage of watershed under agriculture in the entire estuaries having 'pits'.

Within individual estuaries, well-mixed to weakly stratified, the difference in oxygen between top and bottom varied from 0.1 to $3.3\,mg\,DO\,l^{-1}$ and from 0.1 to 49% saturation, respectively. However, in the 'pits' ($n = 10$) the difference between top and bottom was greater, ranging from 1.9 to $5.4\,mg\,DO\,l^{-1}$ and from 30 to 69% saturation. Oxygen relationships were also considered separately in sub-thermocline waters of the deep basins and at the shallow parts of the estuaries (near-bottom areas above 15 m depth).

In 'pits', the sub-thermocline waters of deep basins, bottom oxygen DO, and percentage saturation were associated with bottom TN concentrations (Table 2a). Only one predicting variable could, however, be used for the model, owing to the small sample size (seven estuaries). Among the estuaries, bottom TN correlated both with bottom oxygen DO and percentage saturation. The negative relationship was very strong, bottom oxygen DO and percentage saturation accounting for 76 and 81% of the variation in bottom TN, respectively (Eqs. (12) and (13) in Table 2, Fig. 4). Moreover, the equation was highly significant. Surprisingly, bottom TP did not statistically significantly link to bottom oxygen DO or percentage saturation, although these variables correlated ($R = 0.48$) with each other. No morphometric or land-use variables were associated with bottom oxygen conditions. This study could not demonstrate a relationship of oxygen with phytoplankton biomass or morphometry through bottom TN concentration.

In shallow parts of the estuaries, bottom oxygen DO could only be explained by land-use, the percentage of watershed under agriculture accounting for 57% of the variation in oxygen (Table 2b, Eq. (14) in Table 4). Morphometric variables, such as mean and maximum depth could not be associated with the equations, which is understandable because variation in these variables were small due to weak stratification of these water areas. In other words, there were no differences between the sensitivities of shallow parts of the estuaries determined by morphometry.

As the set of estuaries and the season (July to September) in this study were slightly different from those in Meeuwig et al. (2000), relationships between Chl, nutrients, and land-use were also considered. In the present study, Chl was a function of land-use and morphometry, but not of land-use alone. For example, the percentage of watershed under urban population and fetch accounted for 44% of the variation in Chl (Eq. (15) in Table 5). TP and TN accounted for 74 and 75% of the variations in Chl (Eqs. (16) and (17)). The pattern explaining the dynamics of TP and TN was similar to

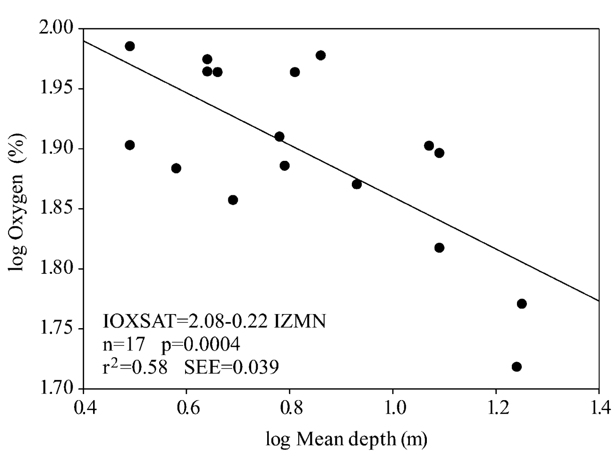

Fig. 3. Regression equation for bottom oxygen percentage saturation as a function of mean depth for 17 estuaries where R^2 is the coefficient of determination and SEE is the model standard error of the estimate.

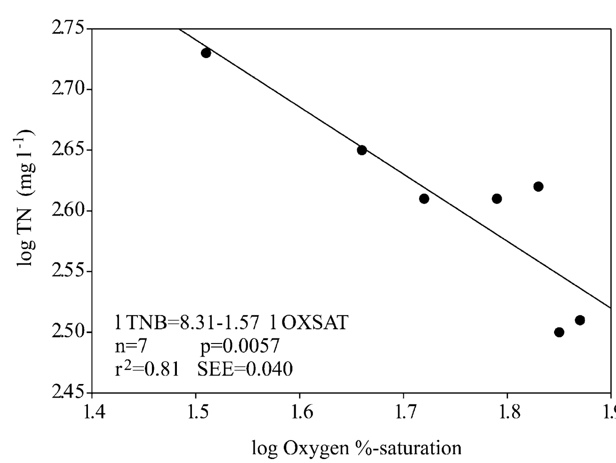

Fig. 4. Regression equation for bottom TN as a function of bottom oxygen percentage saturation for seven 'pits', where R^2 is the coefficient of determination and SEE is the model standard error of the estimate.

Table 5
Models for chlorophyll *a* and bottom nitrogen concentrations for entire estuaries

Model (equation number)		n	R^2	p	SEE	B_0	B_1	B_2
$\log \text{Chl} = B_0 + B_1 \log \text{Urb} + B_2 \log \text{Fetch}$	(15)	17	0.44	0.0167	0.635	0.95***	0.74***	−0.39*
$\log \text{Chl} = B_0 + B_1 \log \text{TN}$	(16)	19	0.75	0.0001	0.323	−2.60***	1.32***	
$\log \text{Chl} = B_0 + B_1 \log \text{TP}$	(17)	19	0.74	0.0001	0.345	−0.82***	1.16***	
$\log \text{TN} = B_0 + B_1 \log \text{Urb} + B_2 \log \text{Fetch}$	(18)	17	0.62	0.0011	0.136	2.71***	0.47**	−0.28*
$\log \text{TP} = B_0 + B_1 \log \text{Urb} + B_2 \log \text{Fetch}$	(19)	17	0.67	0.0007	0.179	1.56***	0.59***	−0.37**

Symbols for variables and statistics are as described in Tables 1 and 4.

that of Chl: equations that combined land-use and morphometric variables (Eqs. (18) and (19)) were stronger than equations with land-use or morphometry alone.

3.2. Results of model validation

Using the adjusted *R*-square estimation according to Wherry (Eq. (1)), the population multiple correlation coefficients in the models were rather similar to the coefficients of determination (R^2), the adjusted R^2 ranging from 0.47 to 0.77 for the model using mean depth as an independent variable and for the model in ‘pits’, respectively (Eqs. (9) and (13), Table 6). The estimated amounts of shrinkage remained usually below 5%, being, however, 11% for the Eq. (9).

Being interested in cross-validity predictive power, the Stein formula (Eq. (2)) was used to compare the fact that how well the equations fitted not only to a specific set of data, but also to entirely different sets of data. In this case, one expects to be able to account for 39 and 45% of the variance in oxygen for entire estuaries (Eqs. (6) and (9)) and for 67% of the variance in TN for ‘pits’ (Eq. (13), Table 6). The shrinkage varied from 8 to 19% (Eqs. (9) and (14)).

The results of the PRESS statistics, were quite similar to those obtained using the Stein formula. For the models of land-use and morphometry (Eqs. (6), (9), and (14), Table 6), one expects to be able to account for 54 to 58% of the variance in oxygen. The best cross-validity predictive power was obtained for the model in ‘pits’, where $R^2_{(\text{PRESS})}$ was 0.81 (Eq. (13)).

4. Discussion

Bottom oxygen DO and percentage saturation can be explained in estuaries both as a function of land-use and morphometry ($R^2 = 0.54$ and 0.58 in Eqs. (6) and (9), respectively). Oxygen models in lakes are even better: Cornett (1989) predicted the change in hypolimnion oxygen concentration (VOD) from phosphorus retention, temperature, and morphometry ($R^2 = 0.91$). Molot et al. (1992) predicted hypolimnion oxygen concentration in lakes by the function of the stratum volume/sediment surface area ratio (VSA), initial oxygen concentration, and both epilimnion and spring turnover TP, the model accounting for 88% of the variation of oxygen. In coastal inlets, Wallin and Håkanson (1991) also presented a strong relationship between bottom oxygen and morphometry (R^2 up to 0.85, $n = 15$, $k = 3$), but the n/k ratio (sample size/predictors) was low for a reliable regression equation, that is, an equation that will cross-validate well (see Stevens, 1996).

In the present set of estuaries, oxygen DO and percentage saturation were explained by land-use or morphometry alone. Meeuwig and Peters (1996) and Meeuwig et al. (2000), however, found that Chl in

Table 6
Statistics for the validation of the models of oxygen concentration and percentage saturation for the entire estuaries, the ‘pits’, and the shallow areas of estuaries (2–15 m)

Model (equation number)	n	k	R^2	Adjusted R^2 Eq. (1)	Adjusted R^2 Eq. (2)	PRESS	$R^2_{(\text{PRESS})}$
Entire estuaries							
$\log \text{Oxy} = 1.27 - 0.35 \log \text{Ag}$ (Eq. (6))	18	1	0.54	0.51	0.45	0.049	0.44
$\log \text{Ox}_{\text{sat}} = 2.08 - 0.22 \log Z_{\text{mn}}$ (Eq. (9))	17	1	0.58	0.47	0.39	0.052	0.43
Pits: hypolimnion							
$\log \text{TN}_{\text{B}} = 8.31 - 0.57 \log \text{Ox}_{\text{sat}}$ (Eq. (13))	7	1	0.81	0.77	0.67	0.069	0.81
Shallow parts of the estuaries							
$\log \text{Oxy} = 1.27 - 0.25 \log \text{Ag}$ (Eq. (14))	17	1	0.57	0.55	0.49	0.014	0.45

Symbols for variables are as described in Table 1. n, sample size; k, the number of predictors; R^2, the coefficient of determination; the adjusted R^2 (Eqs. (1) and (2)), the estimates of the amount of shrinkage found in R^2; PRESS, the predicted residual of the sums of squares (Eq. (4)); $R^2_{(\text{PRESS})}$, R^2-like statistics more accurately reflecting the general applicability of the model (Eq. (5)). See Section 2 for Eqs. (1) and (2) in the adjusted R^2.

estuaries was a function of land-use and morphometry, of which land-use indicates dose or general disturbance in the catchment, whereas morphometry describes inherent properties of the water body. Similar loading and sensitivity effects for e.g. Chl and oxygen concentrations have earlier been presented, for example, by Wallin and Håkanson (1991), and Håkanson (1994). The present result could be a consequence of: (i) correlation between land-use and morphometry; (ii) lack of power; or (iii) different response of oxygen to Chl in lakes and estuaries. The percentage of watershed under agriculture and morphometry did not correlate with each other. Neither one could argue the model by the lack of power, because the coefficient of determination was high and the model was highly significant. It may, therefore, be questioned whether Chl in this set of estuaries still link with land-use, since oxygen did, and oxygen and Chl concentrations were not related to each other. This would be unexpected, as these are the same estuaries as those in Meeuwig et al. (2000), so the model should work because only a few estuaries were replaced, and the summer season was shifted slightly. In fact, the same pattern as in Meeuwig et al. (2000) was found: Chl was a function of land-use and morphometry, but not of land-use alone (Table 5). Chl was also a function of nutrients. This suggests that oxygen concentration and percentage saturation in near-bottom water layer also might respond to external disturbance independent of morphometry, probably mainly due to the responses of oxygen to internal processes.

The pattern in which oxygen DO and percentage saturation are not related to Chl or nutrients is somewhat unexpected, oxygen usually decreases in an enriched system (OECD, 1982). In the deep coastal basins of the eutrophied Gulf of Finland, the relationship between near-bottom oxygen concentration and the state of the surface sediment was not straightforward: the physical and biogeochemical behaviour of the deep basins was strongly affected by local morphometric conditions (Pitkänen et al., 2001). The fact that oxygen DO and percentage saturation in Finnish estuaries did not correlate with Chl or nutrients may result from the annual sedimentation/sediment–water flux dynamics. Most of the P and N sediment out of the mixed surface layer in spring (April–May), whereas only a very small portion of annual external nutrients reaches the estuaries in summer (Heiskanen & Tallberg, 1999; Pitkänen, 1994); in fact, most of the nutrients are coming from the sea at this time. The summer concentration of P in the near-bottom waters of the estuaries is a function of numerous processes (Nixon & Pilson, 1984) that have time to change the actual concentrations after the peak sedimentation. Moreover, in the present study, the coefficient of determination in regression analyses may be weakened by the absence of real extreme values of bottom oxygen DO and percentage saturation in entire estuaries, and high P values in some estuaries (e.g. estuaries 21 and 42) due to resuspension, not due to poor oxygen conditions.

Hypoxia and anoxia are also partly natural features of an aquatic system. In some North American and Baltic estuaries, wind and river flow may strongly influence stratification and bottom oxygen condition (Malve et al., 2000; Nixon, 1988; Stanley & Nixon, 1992). For example, in the Pamlico River Estuary, no cause-and-effect relationship between nutrients or algal abundance and bottom water DO was found (Stanley & Nixon, 1992). The simulation studies by Malve et al. (2000) show that bottom oxygen DO in the Pojo Bay (estuary 23) in Finland was not affected by the loading of organic matter and temperature variations in hypolimnion. This may be explained by the low organic carbon content of the settled material in the Pojo Bay, which according to Heiskanen and Tallberg (1999) indicates that the input of planktonic material is diluted by substantial amounts of resuspended or allochthonous material. By contrast, bottom oxygen DO in the Pojo Bay was related to catchment outflow and sea level variations, saline water inflow into the estuary being determined by favourable wind direction (Malve et al., 2000).

In the present study, bottom oxygen DO and percentage saturation were not related to catchment outflow, but the results suggest that Finnish estuaries are to some extent wind-sensitive, as fetch accounted for about 30% of the variation in oxygen. Furthermore, the coastal morphometry characterized by enclosed deep basins with restricted mixing conditions supports the hypothesis that hypoxia in deep estuaries (e.g. estuaries 23, 24, and 27) is partly contributed by natural features.

Both oxygen concentration and percentage saturation models were comparable when predicted from agriculture. The models were curvilinear, overestimating bottom oxygen values in the deepest and most enclosed estuaries. However, the scatter and residual plots of the shallow parts of estuaries indicated no curvilinearity, which confirms the assumption that the deep estuaries with weakened water exchange are responsible for the curvilinearity in the models for the whole estuaries. In the case of morphometry, oxygen percentage saturation was more strongly linked to mean depth than oxygen DO. This is understandable because temperature, incorporated in oxygen percentage saturation, is affected by changing depth.

The model in the 'pits' was also very good: oxygen DO and percentage saturation were strongly correlated with ambient (near-bottom) TN (Table 2). This is explained by weak circulation in these 'pits', contributing to the accumulation of nitrogen in sub-thermocline waters. In Danish micro-tidal estuaries, oxygen percentage saturation was shown to be related to TN load rather than to TP load (Moehlenberg, 1999). However, no link was found between ambient TN and TN loadings. The lack of the relationship between ambient TN and Chl may be due to the fact that ca. 50% of nitrogen in Finnish rivers is bound to DOM (Pitkänen, 1994), most of which releases bioavailable N only slowly to be used by phytoplankton (Burton, 1988). Additionally, deep water N is affected by denitrification, the main process removing bioavailable N from the coastal marine systems (Seitzinger, 1988).

The model explaining oxygen DO in the shallow parts of the estuaries was similar to that of the entire estuaries. It is understandable as to why oxygen in shallow parts of the estuaries was not a function of morphometry (mean depth or fetch). Despite the general estuarial circulation with outflowing fresh-water in surface and inflowing more saline water in bottom, water column stratification in the shallow parts of the estuaries was usually weak. Moreover, wind mixing in shallow estuaries tends to decrease water column stratification more frequently, so that bottom water hypoxia is generally of short duration and limited in spatial extent (Stanley & Nixon, 1992). This explains as to why oxygen did not correlate with morphometry.

Due to the small sampling number, cross-validation of the oxygen models in Finnish estuaries was estimated by means of the adjusted *R*-square according to the Stein formula (Eq. (2)) and by the PRESS statistics (Eq. (3)). Taking into account the fact that the population represented different kinds of estuaries, both of these approaches supported the view that the oxygen models are applicable in Finland (Table 6). On the basis of the Stein formula, one could expect the predictive power of the best models (Eqs. (6), (13), and (14) in Table 6) to be reduced by about 8–14%, due both to the small population number and also to variation that remained unexplained. As a result of the low values of the sums of squared residuals, the PRESS statistics also gave reasonably good estimates of the general applicability of models based on the same sample size as in this study. For the models predicting oxygen as a function of land-use or morphometry, the $R^2_{(\mathrm{PRESS})}$ was reasonably good ranging from 0.43 to 0.45 (Eqs. (6), (9), and (14)), whereas for the model between near-bottom TN and oxygen percentage saturation, it was really good, 0.81 (Eq. (13)).

In conclusion, oxygen DO and percentage saturation can be predicted empirically in Finnish estuaries as a function of both land-use (the percentage of watershed under agriculture) and of morphometry (mean depth). This is useful for management, as the Finnish coastline is very complex and includes numerous small estuaries. It is not possible to arrange and fund all the different kind of sampling efforts and analyses presumed by the classification of EC's Water Framework Directive (WFD) to cover the whole 1200 km long coastal water zone. Evaluation of the effects of land-use development on bottom oxygen conditions is important for the action of water policy, both at national and international levels (e.g. WFD, Nitrates Directive), because agriculture is a major source of nutrients transported to Finland's coastal waters (Kauppila et al., 2001; Pitkänen, 1994; Vuorenmaa, Rekolainen, Lepistö, Kenttämies, & Kauppila, 2002). Information on the sensitivity of an estuary to eutrophication, given by coastal morphometry, such as mean depth and fetch, must also be taken into account in management plans. More studies on the internal processes are needed in order to understand the critical factors affecting coastal eutrophiction and to produce more reliable modelling applications.

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References

Aure, J., & Stigebrandt, A. (1989). On the influence of topographic factors upon the oxygen consumption rate in sill basins of fjords. *Estuarine, Coastal and Shelf Science 28*, 59–69.

Baden, S. P., Loo, L.-O., Pihl, L., & Rosenberg, R. (1990). Effects of eutrophication on benthic communities including fish: Swedish west coast. *Ambio 19*, 113–122.

Bonsdorff, E., Blomqvist, E. M., Mattila, J., & Norkko, A. (1997). Long-term changes and coastal eutrophication. Examples from the Åland Islands and the Archipelago sea, northern Baltic Sea. *Oceanologica Acta 20*, 319–328.

Bowden, K. F. (1980). Physical factors: salinity, temperature, circulation, and mixing processes. In E. Olausson, & I. Cato (Eds.), *Chemistry and biogeochemistry of estuaries* (pp. 37–70). Chichester: Wiley.

Boyton, W. R., Kemp, W. M., & Keefe, C. W. (1982). A comparative analysis of nutrients and other factors influencing estuarine productivity. In V. S. Kennedy (Ed.), *Estuarine comparisons* (pp. 60–90). New York: Academic Press.

Boyton, W. R., Murray, L., Hagy, J. D., Stokes, C., & Kemp, W. M. (1996). A comparative analysis of eutrophication patterns in a temperate coastal lagoon. *Estuaries 19*, 408–421.

Burton J. P. (Ed.). (1988). *River inputs to ocean systems: Status and recommendations for research*, UNESCO technical papers in marine research, no. 46 (25 pp.).

Cattaneo, A. (1990). The effect of fetch on periphyton spatial variation. *Hydrobiologia 206*, 1–10.

Cederwall, H., & Elmgren, R. (1990). Biological effects of eutrophication in the Baltic sea, particularly the coastal zone. *Ambio 19*, 109–112.

Charlton, M. N. (1980). Hypolimnion oxygen consumption in lakes: discussion of productivity and morphometry effects. *Canadian Journal of Fisheries and Aquatic Science 37*, 1531–1539.

Cornett, R. J. (1989). Predicting changes in hypolimnetic oxygen concentrations with phosphorus retention, temperature, and morphometry. *Limnology and Oceanography 34*, 1359–1366.

Cornett, R. J., & Rigler, F. H. (1979). Hypolimnion oxygen deficits: their prediction and interpretation. *Science 205*, 508–581.

Cornett, R. J., & Rigler, F. H. (1980). The areal hypolimnetic oxygen deficit: an empirical test of the model. *Limnology and Oceanography 25*, 672–679.

Crisman, T. L., Chapman, L. J., & Chapman, C. A. (1998). Predictors of seasonal oxygen levels in small Florida lakes: the importance of color. *Hydrobiologia 368*, 149–155.

Diaz, R. J. (2000). Overview of hypoxia around the world. *Journal of Environmental Quality 30*(2), 275–281.

Finnish Meteorological Institute (1990–1995). *Meteorological yearbook of Finland 1989, 1990, 1991, 1992, 1993*. Helsinki, Finland.

Fonselius, S. H. (1969). *Hydrography of the Baltic deep basins III. Fishery Board of Sweden, Series hydrography* (97 pp.). Report 23.

Gowen, R. J., Tett, P., & Jones, K. J. (1992). Pedicting marine eutrophication: the yield of chlorophyll from nitrogen in Scottish coastal waters. *Marine Ecology Progress Series 85*, 153–161.

Grasshoff, K., Erhardt, M., & Kremling, K. (Eds.). (1983). *Methods of seawater analysis*. (419 pp.). Weinheim: Verlag Chemie.

Håkanson, L. (1994). A review on effect–dose–sensitivity models for aquatic ecosystems. *International Review of Hydrobiology 79*, 621–667.

Hansen, I. S., Ærtebjerg, G., & Richardson, K. (1995). A scenario analysis of effects of reduced nitrogen input on oxygen conditions in the Kattegat and the Belt Sea. *Ophelia 42*, 75–93.

Heiskanen, A.-S., & Tallberg, P. (1999). Sedimentation and particulate nutrient dynamics along a coastal gradient from a fjord-like bay to the open sea. *Hydrobiologia 393*, 127–140.

Jensen, H. S., Mortensen, P. B., Andersen, F. Ø., Rasmussen, E., & Jensen, A. (1995). Phosphorus cycling in a coastal marine sediment, Aarhus Bay, Denmark. *Limnology and Oceanography 40*, 908–917.

Jørgensen, B. B. (1996). Case study—Aarhus Bay. In B. B. Jørgensen, & K. Richardson (Eds.), *Eutrophication in coastal marine ecosystems. Coastal and Estuarine Studies 52*, 137–154.

Jørgensen, B. B., Bang, M., & Blackburn, T. H. (1990). Anaerobic mineralization in marine sediments from the Baltic Sea–North Sea transition. *Marine Ecology Progress Series 59*, 39–54.

Kauppila, P., Korhonen, M., Pitkänen, H., Kenttämies, K., Rekolainen, S., & Kotilainen, P. (2001). Loading of pollutants. In P. Kauppila, & S. Bäck. (Eds.), *The state of Finnish coastal waters in the 1990s. The Finnish Environment 427* (134 pp.).

Koroleff, F. (1976). Determination of nutrients. In K. Grasshoff (Ed.), *Methods of seawater analysis* (pp. 117–133). Weinheim/New York: Verlag Chemie.

Kuo, A. Y., Park, K., & Moustafa, M. Z. (1991). Spatial and temporal variabilities of hypoxia in the Rappahannock River, Virginia. *Estuaries 14*, 113–121.

Larsson, U., Elmgren, R., & Wulff, F. (1985). Eutrophication and the Baltic Sea: causes and consequences. *Ambio 14*, 9–14.

Lorenzen, C. J. (1967). Determination of chlorophyll and pheopigments: spectrophotometric equations. *Limnology and Oceanography 12*, 343–346.

Malmgren-Hanse, A., Mortensen, P., & Møller, B. (1984). Modelling of oxygen depletion in coastal waters. *Water Science and Technology 17*, 967–978.

Malve, O., Virtanen, M., Villa, L., Karonen, M., Åkerla, H., Heiskanen, A.-S., Lappalainen, K.-M., & Holmberg, R. (2000). Artificial oxygenation experiment in hypolimnion of Pojo Bay Estuary in 1995 and 1996: factors regulating estuary circulation and oxygen and salt balances. *The Finnish Environment 377*, 1–163 (In Finnish with English summary).

Meeuwig, J. J. (1999). Predicting coastal eurtrophication from land-use: an empirical approach to small non-stratified estuaries. *Marine Ecology Progress Series 176*, 231–241.

Meeuwig, J. J., Kauppila, P., & Pitkänen, H. (2000). Predicting coastal eutrophication in the Baltic: a limnological approach. *Canadian Journal of Fisheries and Aquatic Science 57*, 844–855.

Meeuwig, J. J., & Peters, R. H. (1996). Circumventing phosphorus in lake management: a comparison of chlorophyll *a* predictions from land-use and phosphorus-loading models. *Canadian Journal of Fisheries and Aquatic Science 53*, 1795–1806.

Moehlenberg, F. (1999). Effects of meteorology and nutrient load on oxygen depletion in a Danish micro-tidal estuary. *Aquatic Ecology 33*, 55–64.

Molot, L. A., Dillon, P. J., Clark, B. J., & Neary, B. P. (1992). Predicting end-of-summer oxygen profiles in stratified lakes. *Canadian Journal of Fisheries and Aquatic Science 49*, 2363–2372.

Nixon, S. W. (1981). Remineralization and nutrient cycling in coastal marine ecosystems. In B. M. Neilson, & L. E. Cronin (Eds.), *Estuaries and nutrients* (pp. 111–138). Clifton: Humana Press.

Nixon, S. W. (1988). Physical energy inputs and the comparative ecology of lake and marine ecosystems. *Limnology and Oceanography 33*, 1005–1025.

Nixon, S. W. (1995). Coastal marine eutrophication: a definition, social causes, and future concerns. *Ophelia 41*, 199–219.

Nixon, S. W., & Pilson, M. W. Q. (1984). Estuarine total system metabolism and organic exchange calculated from nutrient ratios: an example from Narragansett Bay. In V. S. Kennedy (Ed.), *The estuary as a filter* (511 pp.). New York: Academic Press.

NOAA (1997). NOAA's *estuarine eutrophication survey, Mid-Atlantic region (Vol. 2)* (50 pp.). National Ocean Service, Office of Ocean Resources Conservation and Assessment: Silver Spring.

Norkko, A., & Bonsdorff, E. (1996). Population responses of coastal zoobenthos to stress induced by drifting algal mats. *Marine Ecology Progress Series 140*, 141–151.

OECD (1982). *Eutrophication of waters: monitoring, assessment and control* (154 pp.). Paris: OECD.

Oldham, C. E., & Lavery, P. S. (1999). Porewater nutrient fluxes in a shallow fetch-limited estuary. *Marine Ecology Progress Series 183*, 39–47.

Pitkänen, H. (1994). *Eutrophication of the Finnish coastal waters: Origin, fate and effects of riverine nutrient fluxes* (45 pp.). Publications of the Water and Environment Research Institute No. 18.

Pitkänen, H., Lehtoranta, J., & Räike, A. (2001). Internal nutrient fluxes counteract decreases in external load: the case of the estuarial eastern Gulf of Finland, Baltic Sea. *Ambio 30*, 195–201.

Seitzinger, S. P. (1988). Denitrification in freshwater and coastal marine ecosystems: ecological and geochemical significance. *Limnological Oceanography 33*, 702–724.

Stanley, D. W., & Nixon, S. W. (1992). Stratification and bottom-water hypoxia in the Pamlico River estuary. *Estuaries 15*, 270–281.

Stein, C. (1960). Multiple regression. In I. Olkin (Ed.), *Contributions to probability and statistics, essays in honor of Harol Hotelling* (511 pp.). Stanford: Standford University Press.

Stevens, J. (1996). *Applied multivariate statistics for the social sciences* (659 pp.). Mahwah: Lawrence Erlbaum.

Stigebrandt, A., & Wulff, F. (1987). A model for the dynamics of nutrient and oxygen in the Baltic proper. *Journal of Marine Research 45*, 729–759.

Trimbee, A. M., & Prepas, E. E. (1988). Dependence of lake oxygen depletion rates on maximum oxygen storage in a partially meromictic lake in Alberta. *Canadian Journal of Fisheries and Aquatic Science 45*, 571–576.

UNESCO (1985). The International System of Units (SI) in oceanography. *UNESCO technical papers in marine science* (Vol. 45) (124 pp.).

Vollenweider, R. A., & Janus, L. L. (1982). Statistical models for predicting hypolimnetic oxygen depletion rates. *Memorie dell'instituto italiano di idrobiologia dott. Marco de Marchi 40* (pp. 1–24).

Vollenweider, R. A., Marchetti, R., & Viviani, R. (1992). *Marine coastal eutrophication* (1310 pp.). New York: Elsevier.

Vuorenmaa, J., Rekolainen, S., Lepistö, A., Kenttämies, K., & Kauppila, P. (2002). Losses of nitrogen and phosphorus from agricultural and forest areas in Finland during 1980s and 1990s. *Environmental Monitoring and Assessment 76*, 213–248.

Wallin, M., & Håkanson, L. (1991). The importance of inherent properties of coastal areas. *Marine Pollution Bulletin 22*, 381–388.

Welch, E. B., & Perkins, M. A. (1979). Oxygen deficit—phosphorus loading relation in lakes. *Journal WPCF 51*, 2823–2828.

Zar, J. H. (1996). *Biostatistical analysis* (662 pp.). Englewood Cliffs: Prentice-Hall.