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MODELLING THE FLOW OF NITROGEN AND PHOSPHORUS IN EUROPE: FROM LOADS TO COASTAL SEAS

O. Klepper, A.H.W. Beusen and C.R. Meinardi

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This investigation was conducted on behalf and for the account of the Board of Directors of the National Institute of Public Health and Environmental Protection

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80. D. Venema, Hoofdstraat 157A, 9827 PA Lettelbert
81. L. Bijlmakers, Iwaco, Postbus 8520, 3009 AM Rotterdam
82. Dr A.Tonderski, University Dept. of Water and Society, S-581 83 Linkoping, Sweden
83. Dr F.M. Brouwer, LEI-DLO, Postbus 29703, 2502 LS Den Haag

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Abstract

A model is described that aims at predicting surface water quality from N- and P-inputs on a European scale. The model combines a GIS-based approach to estimate loads, geohydrological data to define model structure and statistical techniques to estimate parameter values. The model starts with an inventory of diffuse sources of N and P: agriculture and (for N) atmospheric deposition. Nitrogen flows are assumed to follow both surface- and groundwater flows, while for phosphorus only surface water flow is taken into account. In addition to these non-point sources, sewage is an important source of nutrients, in particular for phosphorus. Using physically realistic parameter values, good agreement is found between observed and calculated riverine loads. Results indicate problem areas in two regions: western Europe (Thames, Scheldt, Elbe, Rhine) with its intensive agriculture and high population density and Southern Europe (some rivers in Spain, Italy), where loads may be locally high and dilution is low due to a low water discharge. For the coastal seas of Europe concentrations were calculated by assuming conservative behaviour of N and P. Existing advection-diffusion models were linked and re-calibrated on salinity data. Results indicate that the main problem areas are the Baltic Sea and the Black Sea, with much lower impacts in the North Sea and the Adriatic Sea; in other coastal waters human impacts are essentially negligible. However, due to seasonal and spatial variability disregarded in this European-scale study, it should be realized that in specific sub-areas, i.c. coastal zone, periods with significant problems still may occur.

Samenvatting

Dit rapport beschrijft een model dat dient om de kwaliteit van het oppervlaktewater te berekenen op Europese schaal. Het model combineert een GIS-gebaseerde aanpak om belastingen te schatten, geohydrologische gegevens om de modelstructuur te bepalen en statistische technieken om parameters te schatten. Het model start met een inventarisatie van diffuse bronnen van N en P: landbouw en atmosferische depositie. Stikstofstromen worden geacht zowel de oppervlakkige afstroming als de grondwaterstroming te volgen, fosfor volgt alleen de oppervlakkige afvoer. Naast diffuse bronnen vormen afvalwaterlozingen een belangrijke bron van nutriënten, in het bijzonder voor fosfor. Met realistische waarden voor de modelparameters blijkt het mogelijk een goede overeenstemming te verkrijgen tussen geschatte riviervrachten en metingen. De resultaten geven twee probleemgebieden aan: West-Europa (Theems, Schelde, Elbe, Rijn) met zijn intensieve landbouw en hoge bevolkingsdichtheid, en Zuid-Europa, waar lokaal hoge belastingen voorkomen, die in combinatie met lage afvoeren van water voor hoge concentraties zorgen. Voor de kustzeeën van Europa werden concentraties berekend op grond van conservatief gedrag van N en P. Bestaande advectie-diffusiemodellen werden gehercalibreerd op basis van saliniteitsgegevens. De resultaten wijzen uit dat de belangrijkste probleemgebieden liggen in de Oostzee en Zwarte Zee, met veel kleinere effecten in de Noordzee en de Adriatische Zee. In andere kustwateren zijn de menselijke invloeden gering. Hierbij moet worden aangemerkt dat door ruimelijke en seizoensvariabiliteit die in deze studie niet is meegenomen, in bepaalde deelgebieden (m.n. kustzones) periodes met belangrijke problemen toch kunnen vóórkomen.

1. Introduction

Nutrient loading by point (wastewater) and non-point sources (agriculture, atmospheric deposition) causes water quality problems both in fresh- and in seawater. Though the problem has a scope (long-range transport) and impact (severe ecosystem- and public health effects) that are comparable to the acid deposition problem, there is at present no model on a European scale comparable to the long-range air pollution models.

In two companion reports (Meinardi et al., 1994a,b), the input and routing of nitrogen to groundwater was described. The pollution of groundwater not only threatens drinking water supplies and natural vegetation, it also affects surface water quality. For phosphorus, the problem has a similar magnitude. Although groundwater pollution by phosphorus is still rare due to the strong adsorption to soil particles, the input of agricultural phosphorus to surface water is a major source of pollution. In addition to this diffuse source of pollution, surface water quality is affected by waste water discharges.

The water pollution problem is typically an international problem, because most of the major rivers of Europe cross at least one international boundary. This is still more the case for the pollution of coastal seas. Obviously, the reduction of pollution problems of European rivers and seas requires an international cooperation, for which several bodies are in existence (among others): ICES, Oslo and Paris commissions for the North Sea and the Helsinki commission for the Baltic Sea and UNEP's Coordinating Unit for the Mediterranean Action Plan in Athens.

The present paper describes a European scale water quality model that could serve policy makers to develop effective strategies for improvement, by indicating "hot spots" with the severest problems, and by being able to calculate the effects of policy measures on water quality.

2. Input and routing of pollutants

Input of nitrogen to the soil is mainly from agriculture; details of the estimation of agricultural N-inputs are given by Meinardi et al. (1994a,b). A similar procedure was followed for the estimation of agricultural phosphorus inputs, but now without the atmospheric component. In both cases, a map of N and P input on the soil was produced, using landuse data to distribute the averages obtained from national statistics (see Appendix 1). These diffuse sources reach surface waters either directly (surface runoff) or via groundwater. The average age of groundwater is determined by infiltration rate (net precipitation, fraction runoff) and storage (aquifer depth and porosity) as described by Meinardi et al. (1994b). In terms of water quality, "old" water generally corresponds to low concentrations, because of the combined effect of lower historical inputs and losses with time. As a first approximation we can relate historical N-inputs (N_t) to current inputs (N_o) as:

$$N_t = N_o \, e^{-t/c} \qquad (1)$$

The value of c corresponds to the period of large-scale fertilizer utilization, and has a value in the order of 50 years.

For nitrogen, the deposition of atmospheric NO_x and NH_y is a considerable contribution, particularly in rural areas and open water. Nitrogen deposition was estimated using TREND model results as reported by Erisman and Heij (1991). For phosphorus, atmospheric deposition may be ignored.

Phosphorus concentrations in groundwater are generally negligible due to the strong adsorption to the soil. With the exception of excessive loading on very poor soils, P-output to surface water is dominated by erosion of topsoil. For a given P-input there will be an increasing P-output with an increasing erosion rate, with a maximum at an output equal to input. This can be described by:

$$P_{output} = P_{input} \, (1 - e^{-b\,R}) \qquad (2)$$

With: P_{output} = runoff load in $ton.km^{-2}.yr^{-1}$
P_{input} = input load in $ton.km^{-2}.yr^{-1}$
R = sediment yield ($ton.km^{-2}.yr^{-1}$)
b = an empirical coefficient ($ton^{-1}.km^{2}.yr$)

Normally, P-output will be only a small fraction of input, so that we may confine ourselves to the initial slope of the equation: $P_{output} \approx P_{input} \, b \, R$.

The erosion rate was estimated using a simplified version of the Universal Soil Loss Equation (Hudson, 1971). It is the product of several factors: an empirical constant to match observations and individual factors for rainfall intensity, slope, texture and landuse:

$$R = c\ P\ S\ T\ L \tag{3}$$

with:
- R - sediment yield ($ton.km^{-2}.yr^{-1}$)
- c - empirical factor ($ton.km^{-2}.yr^{-1}$)
- P - rainfall intensity factor (-)
- S - slope class factor (-)
- T - texture factor (-)
- L - landuse factor (-)

Note that in the present study the term erosion includes all particular material which enters the river system from the land; this is also termed sediment yield (Gleick, 1993). In agriculture-oriented studies the term implies total losses from a particular plot of land. Because part of this loss may be deposited elsewhere, the two are not identical. The same equation used to estimate erosion may be used for sediment yield if we assume that the difference is a constant fraction, which is included in the proportionality factor c.

As detailed information on rainfall intensity was lacking, the amount of precipitation in the wettest month was used as a first approximation. This gives a better approximation than the yearly total (or average) amount of rainfall because a certain amount of rainfall in a strongly seasonal climate (e.g. Mediterranean) has a much more pronounced effect than the same amount in a maritime climate (e.g. Ireland). For slope and texture the data in the FAO-UNESCO soil map of the World (FAO-UNESCO, 1981) with additional data for the EC (CORINE, 1991) were used; the resulting factors are listed in Table 1. Landuse has a strong effect on erosion; landuse was taken from a map prepared by Velde et al (1994), with factors relating erosion rate to a particular landuse class listed in Table 1 (Konsten, 1994).

Table 1 Factors determining erosion rate					
slope		texture		landuse	
class	factor	class	factor	class	factor
0-8%	0.05	very fine	0.25	grass	0.1
0-30%	0.1	fine	1	arable	1
8-30%	0.15	medium-fine	4	permanent crop	0.2
8-30% with inclusions >30%	0.2	medium	4	forest	0.01
>30%	0.3	medium-coarse	5.25	water	0.
		medium-coarse	6.7	urban	0.05
		coarse	9	other	0.05
		histosols	0		

The erosion model results in a total factor for each grid cell. From Gleick (1993) we have a European-average sediment yield of 46 ton.km^{-2}.yr^{-1}. This value was used to estimate the value of c. The spatial pattern of sediment yield matches with the data given by Gleick: high values in mountainous areas (average 120 ton.km^{2}.yr^{-1}), low values in Western Europe (average 12 ton.km^{-2}.yr^{-1}) and intermediate values in the Black Sea catchment areas (average 72 ton.km^{-2}.y^{-1}); see Figure 1.

The fate of pollution via wastewater differs from the previous sources in its more direct pathway to surface water. We estimated sewage production from population density (Fig. 2). In the relation between population numbers and N- and P-inputs we have to take into account that more N and P is produced than by domestic sewage alone (industrial pollution) and that part of the nutrients are removed by treatment. The two factors tend to compensate: in industrialized regions there is generally a high wastewater production per capita, but also a high degree of treatment. As a first approximation, we assumed these two effects to cancel, making N- and P-load proportional to population numbers. The estimation of the proportionality constant is described in section 3 below.

A flow diagram of the N- and P-routing is given in Figure 3.

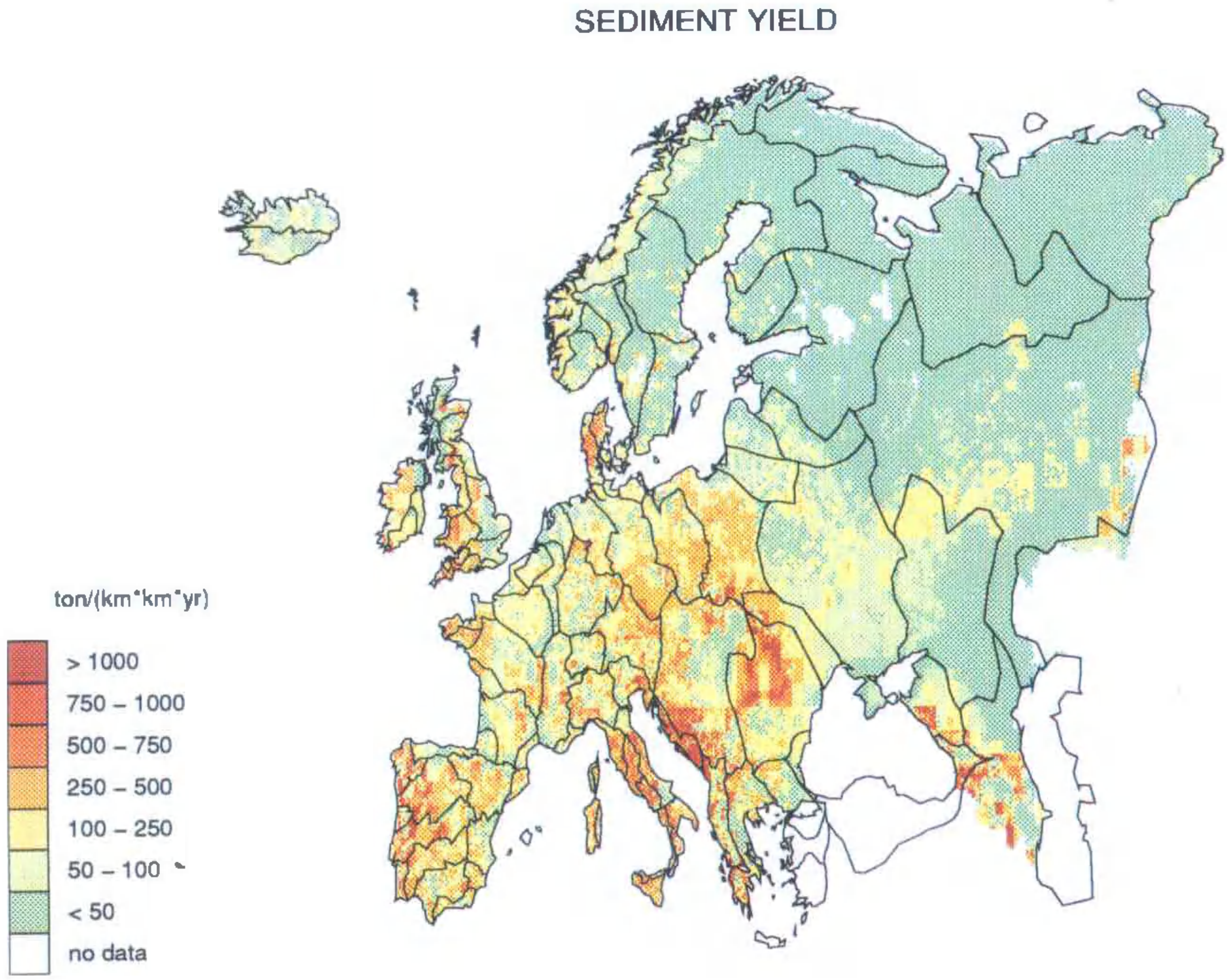

Figure 1. Sediment yield in Europe. Results of model calculations.

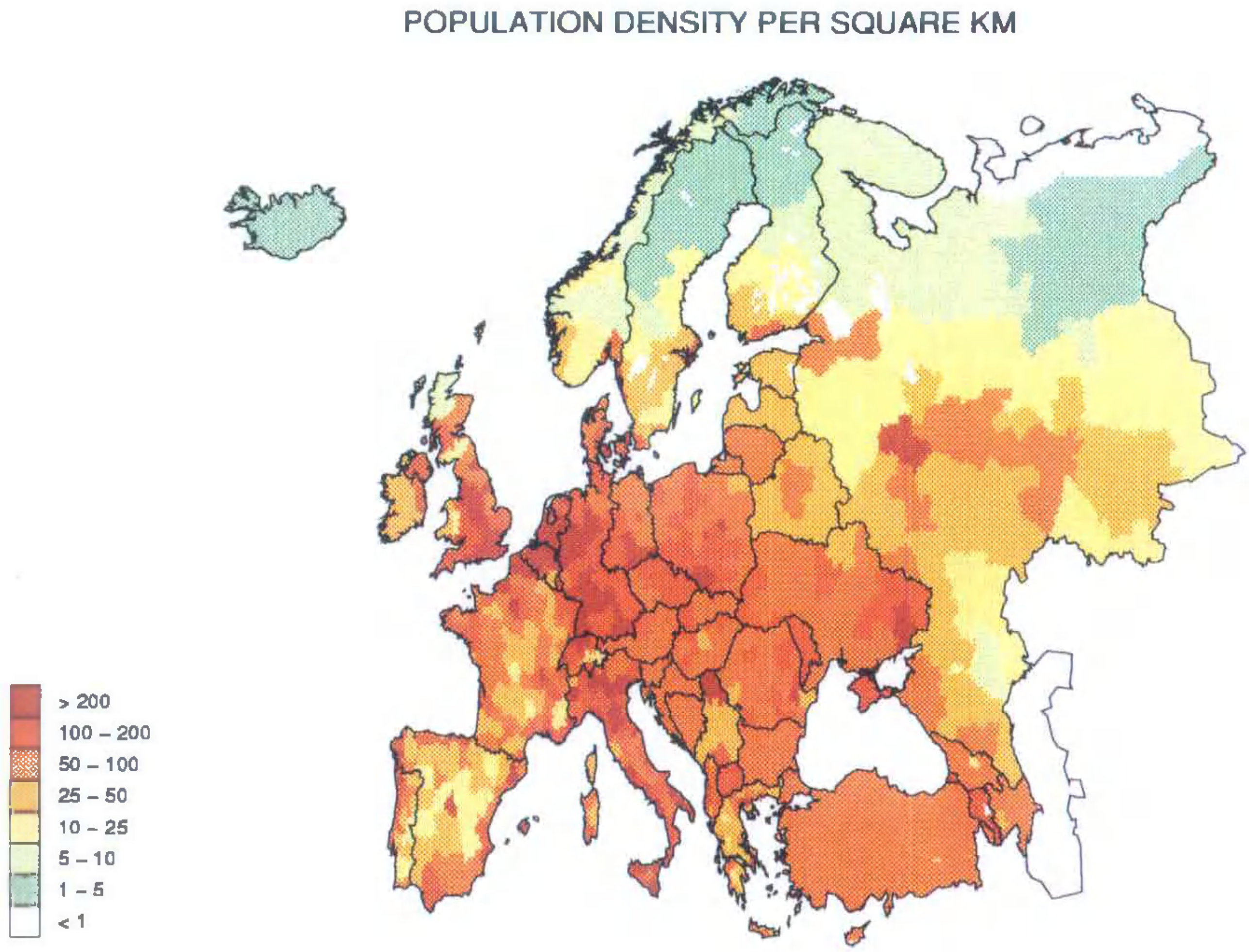

Figure 2. Population density in Europe. Source: RIVM-ISC, WHO.

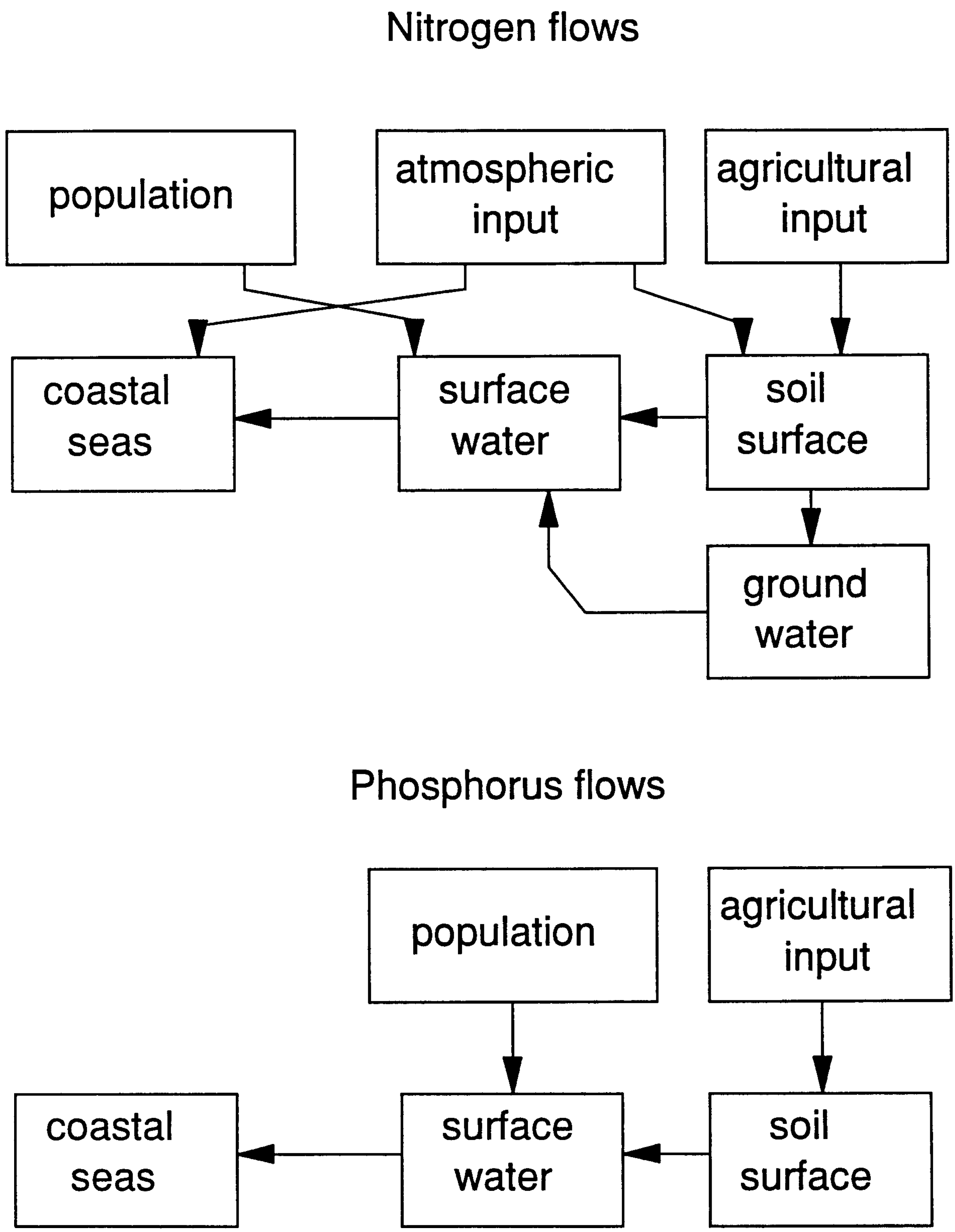

Figure 3. Flow diagram of nitrogen and phosphorus routing.

3. Calibration

Water

The model to determine river discharge is described by Meinardi et al, 1994a. In summary, long-term average precipitation and temperature data are used to estimate runoff. In a comparison of models (Klepper and Beusen, 1994), the Turc-Langbein equation for actual evapotranspiration gave an excellent fit to observed (UNESCO, 1974-1994; Gleick, 1993) discharges (Fig. 4). In the present work we need river flows to calculate concentrations of pollutants. For this purpose, we use the calculated rather than the observed discharges for two reasons. In the first place, climatological averages cover a much longer period than discharge records: it may be assumed that a large part of the scatter in figure 4 is the result of inter-annual variability in flows. In some sense, the model may therefore be a better estimate of long-term average flow than the average of a short observation period. In the second place, many of the catchment areas in the model are not, or at least not completely gauged. The model is then a means to estimate missing flows.

Catchment areas as used in the model are generally those defined by Eurostat (Statistical office of the European Community, Brussels). In some regions (e.g., Iceland, Great Britain) the original catchment areas were lumped to larger units. The very large catchment areas of the Rhine and Danube rivers were sub-divided in an upper, middle and lower course, using data from the UNESCO hydrogeological map of Europe.

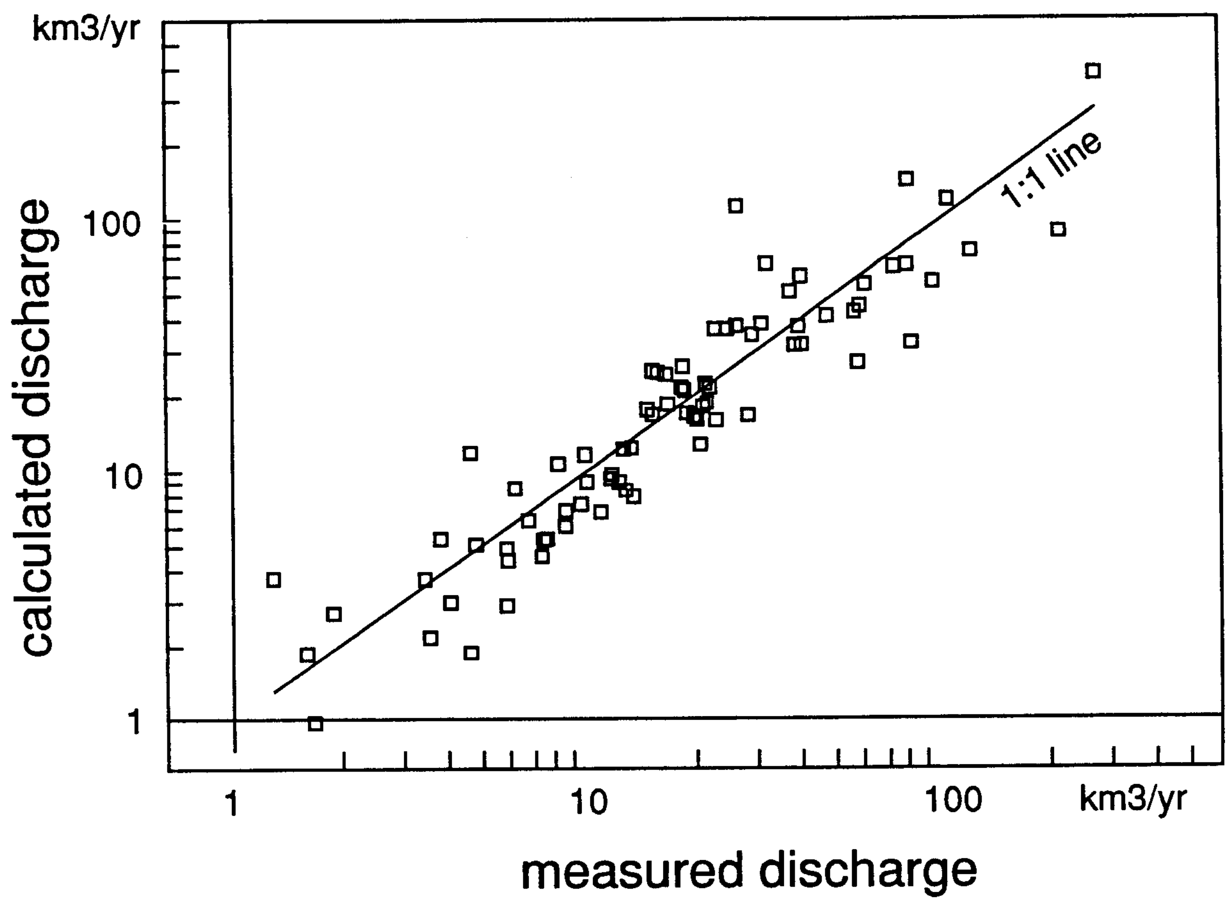

Figure 4. Observed and calculated river discharge.

Nitrogen

The model for diffuse sources and pathways of nitrogen results in a load of nitrate to surface waters. Although nitrate is the only fraction with an appreciable mobility in the soil, this is no longer the case in surface waters: due to uptake by algae and higher plants and subsequent mineralization, there are now three major N-species: nitrate, ammonia and organic nitrogen. The latter two fractions make up approximately 30% of the total (BMEPC, 1993; CEG, 1989). It should be realized that the factor is not a true constant, and depends on the trophic status of the water: in heavily polluted waters, the contribution by nitrate is lower. In completely anaerobic water there may be no nitrate at all despite a heavy load of nitrogen. For most stations, only the inorganic fractions of N are measured, often only nitrate. For this reason, it is most efficient to use the nitrate data for model calibration. In order to take into account the missing ammonia and organic fractions, the diffuse source of N was multiplied by the proportionality factor 0.7 (i.e. 100% minus 30%).

The model calculates N-input from wastewater as proportional to population density; as mentioned before the proportionality factor is not exactly known, as it is a combination of a multiplier that incorporates industrial wastewater on a per capita basis and losses due to purification.

It was preferred to calibrate loads rather than concentrations, because total loads are more relevant for the receiving seas. For a perfect model, the results would be identical. Otherwise, calibration of concentrations would have give more emphasis on the high concentrations (which may occur in a small catchment only). Data were compiled from CEG (1989), Eurostat (1989), Mee (1992), BMEPC (1993). They generally refer to the total river load (i.e. stations close to the river mouth), with the exception of the Rhine and the Danube rivers, which were subdivided into an upper, middle and lower course.

As a first approach, we used a linear regression to observed loads:

$$L_{observed} = 0.7\ [c_1\ L_{diffuse} + c_2\ N_{population}] \qquad (4)$$

The coefficient c_1 is dimensionless ($L_{observed}$ and $L_{diffuse}$ are already expressed in the same units), with a value in the range [0,1]. The coefficient c_2 gives the (effective) nitrate load per capita. A linear regression gave:

$c_1 = 0.366$ std.error 0.140

$c_2 = 19.6\ gN.inh^{-1}.day^{-1}$ std.error $2.47\ gN.inh^{-1}.day^{-1}$

with $r^2 = 0.92$

The value of 19.6 g $N.inh^{-1}.day^{-1}$ is rather high in comparison with the figures of 13 g $N.inh^{-1}.day^{-1}$ given by CBS (1987) and 6-10 $gN.inh^{-1}.day^{-1}$ by Hellmann (1989). Also, the fact that we recover only 37% of the diffuse loads seems very low: as all factors in the soil like denitrification and groundwater delay are already incorporated in the model, the 63% loss must be attributed to in-stream processes, i.e. denitrification. In practice, denitrification does not seem to be a major factor in well-oxygenated rivers, and plays a significant role in a limited number of heavily polluted rivers only (e.g., the Scheldt in Belgium). Also, the total contribution of agriculture and atmospheric deposition is calculated to be only 26%, which is lower than generally assumed (Behrendt, 1993; Werner and Wodsak, 1994).

Incorporating these considerations, the model was calibrated with a fixed value of c_1=1. This leads to a fit:

$c_2 = 9.20$ gN.inh^{-1}.day^{-1} std.error 1.10 gN.inh^{-1}.day^{-1}

with $r^2 = 0.88$

In this case, the European average contribution of diffuse loads is 57%, which is reasonable in view of the value of 49% for the Rhine basin (Behrendt, 1993) and of 57% for Germany (Werner and Wodsak, 1994).

The fact that the fit to observed values shows only a modest decrease despite a considerable change in parameter values can be attributed to the high correlation between the two inputs: domestic and agricultural. In most of Europe, the densely populated lowlands are also the areas of intensive agriculture; this high correlation is illustrated in Figure 5.

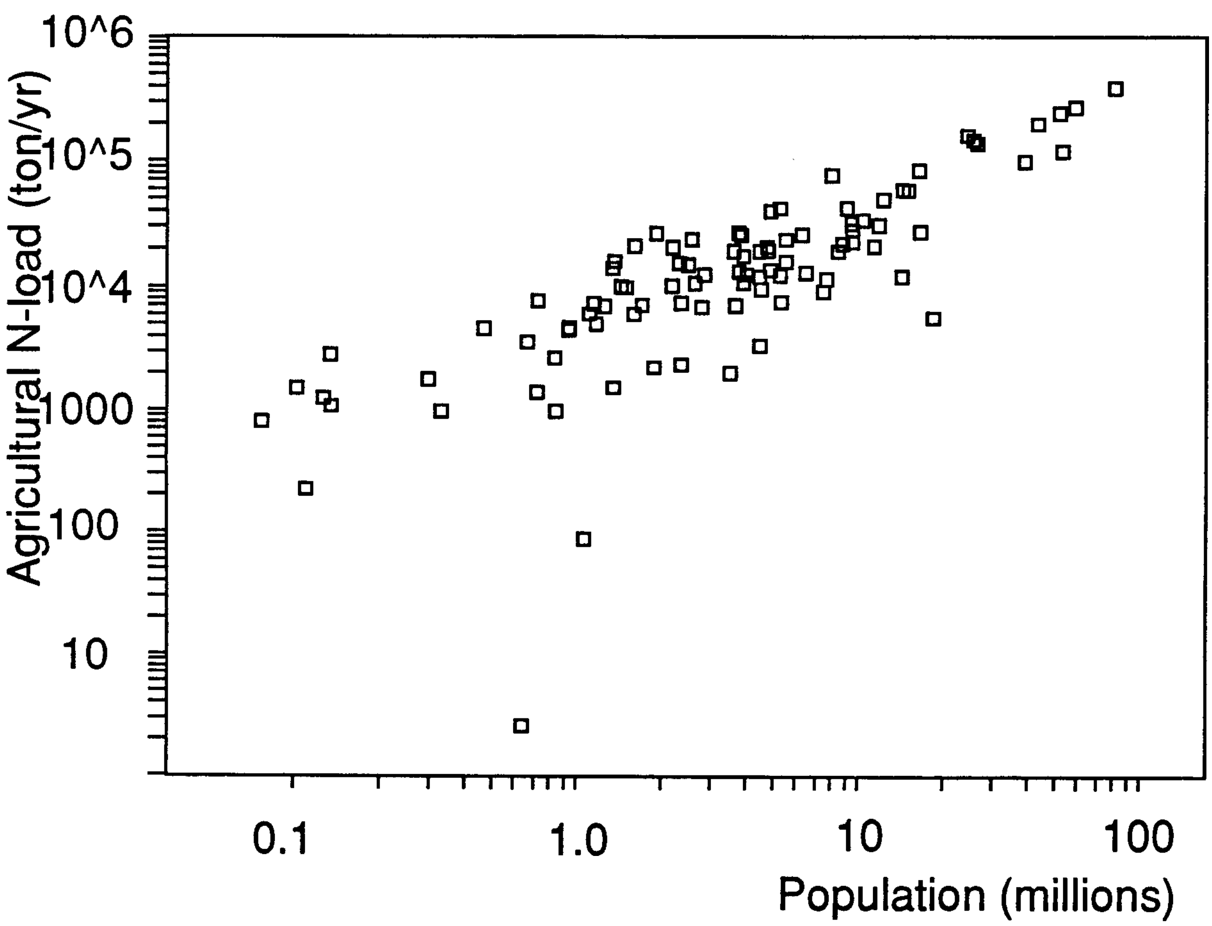

Figure 5. Agricultural load of nitrogen vs. population numbers in the catchment areas of Europe.

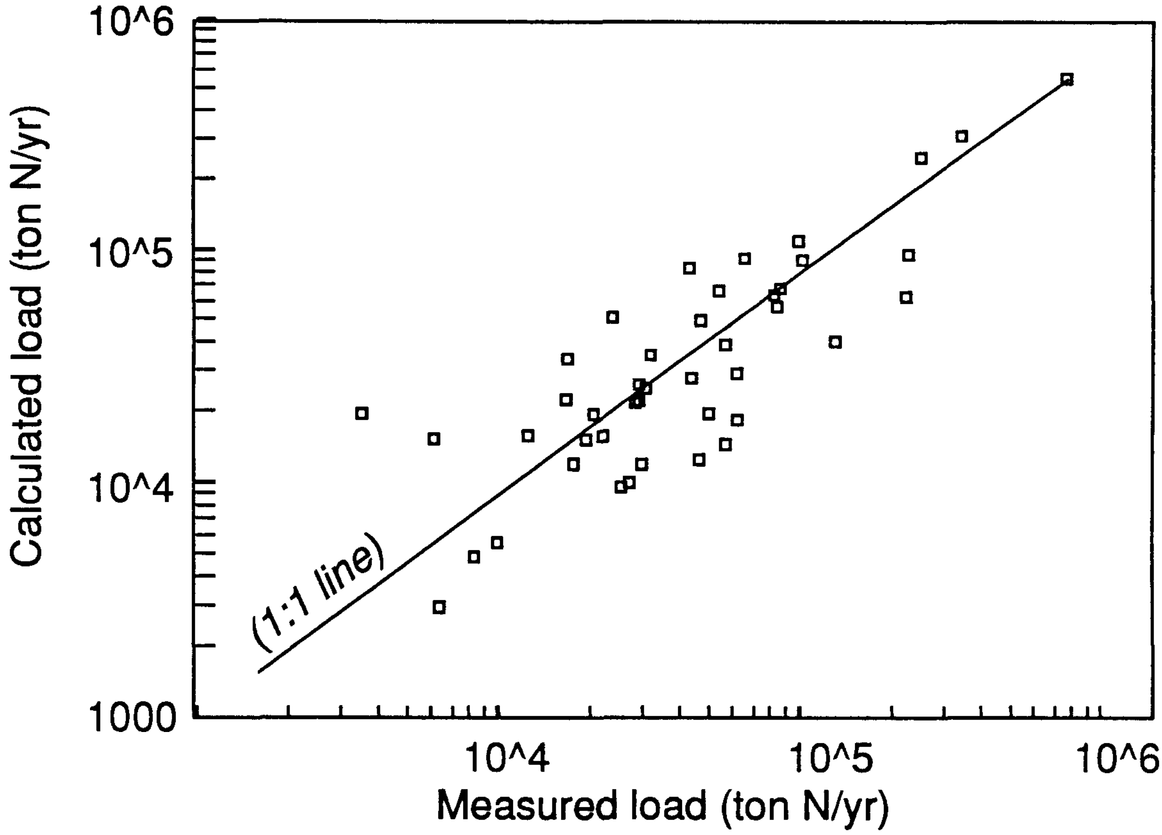

Figure 6. Calculated and observed nitrate loads.

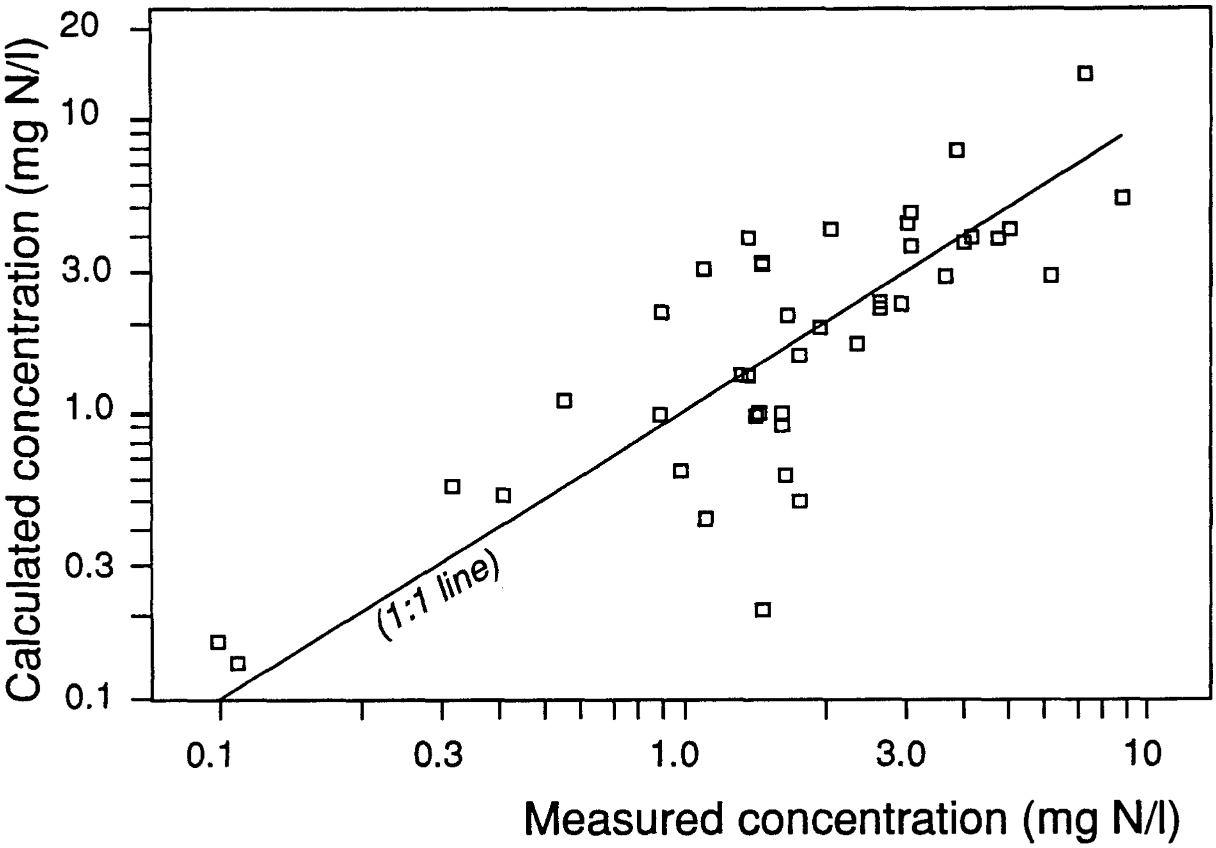

Figure 7. Calculated and observed nitrate concentrations.

Phosphorus

For phosphorus, we have to estimate both the coefficient b in equation (2) and the P-load per inhabitant. Data were compiled from CEG (1989), Eurostat (1989), Mee (1992), BMEPC (1993). Again, linear regression of calculated on observed loads was employed.

$$L_{observed} = d_1 \; N_{population} + d_2 \sum R \; P_{input}$$

with: $L_{observed}$ - phosphorus load
d_1, d_2 - coefficients
$N_{population}$ - number of inhabitants
$\Sigma R \; P_{input}$ - the summation of sediment yield times p-input over the catchment area

The resulting coefficient values area:

$d_1 = 2.12$ gP.inh^{-1}.day^{-1} std.error 0.25 gP.inh^{-1}.day^{-1}

$d_2 = 1.18 \; 10^{-4}$ (ton$_{soil}$.ton$_p$)$^{-1}$km^{-2}.yr^{-1} std.error $5.43 \; 10^{-5}$

The model fits closely to measurements, with an r^2 of 0.89 (Fig. 8 and 9).

The per capita P-input is in reasonable agreement with the CBS (1987) figure of 3.4 gP.inh^{-1}day^{-1}, implying a combined effect of extra industrial input and in-stream loss rate of some -40%. In contrast to nitrogen, in-stream losses (due to sedimentation) can be considerable. The model calculates present European-average agricultural contribution to P-**output** as 18%. In a comparison with literature values, it should be noted that agricultural sources are not identical to diffuse sources: the latter include all non-constant (discharge-dependent) inputs. For this reason, both Behrendt (1993) and Werner and Wodsak (1994) give higher diffuse contributions of 37% and 39% for the Rhine and for Germany, respectively. However, even if we consider agriculture separately, our figure is still below the 28% estimated by Werner and Wodsak (1994).

The erosion-related coefficient d_2 implies a loss rate of on average 97% of agricultural P-input. It may be noted that the role of this loss-rate is similar to the loading of groundwater with nitrates: the accumulation of P in the soil (or in riverine sediments) is not a definitive storage, but will ultimately cause eutrophication problems. If we compare different types of P-**input**, agriculture makes up 75-85% of inputs. This much higher contribution is a result of the higher retention in the environment of agricultural inputs in comparison with domestic and industrial discharges.

If we compare our results for the Rhine and Vistula catchment areas with those of Tonderski et al. (1994) we see that the model accurately estimates N- and P-concentrations for the Rhine, but overestimates both substances for the Vistula by approximately a factor 2 (Table 2). For nitrogen, this is probably due to estimated input: the present model has a factor 2 higher input as was estimated by Tonderski et al. The model does agree with the lower specific export of N for the Vistula: for the Rhine, 8.0% of agricultural N-input is exported, for the Vistula this is 5.5%. For P, the difference between model and observations cannot be explained by inputs; furthermore, the model calculates a higher specific export coefficient for P in the Vistula as compared to the Rhine, due to higher erosion rates: this is in contrast to the analysis of Tonderski et al. In this case, the explanation is probably the effect of the Wloclawek reservoir, in which some 45% of the P-load settles (Tonderski et al., 1994): this feature is not included in the model, and would explain the twofold over estimate by the present model.

Table 2. A comparison between Rhine and Vistula catchments

	Rhine		Vistula	
	P	N	P	N
observed conc. (mouth)	0.5 $mgP.l^{-1}$	4.1	0.2	1.5
model conc.	0.51 $mgP.l^{-1}$	3.86	0.48	3.25
Agric. input Tonderski e.a.	1.35 10^5 ton P	1.15 10^6 ton N	1.20 10^5 ton P	8.15 10^5 ton N
Agric. input own estimate	2.91 10^5 ton P	1.99 10^6 ton N	1.28 10^5 ton P	1.72 10^6 ton N
Percentage of agric. input exported	2.6%	8.0%	4.0%	5.5%

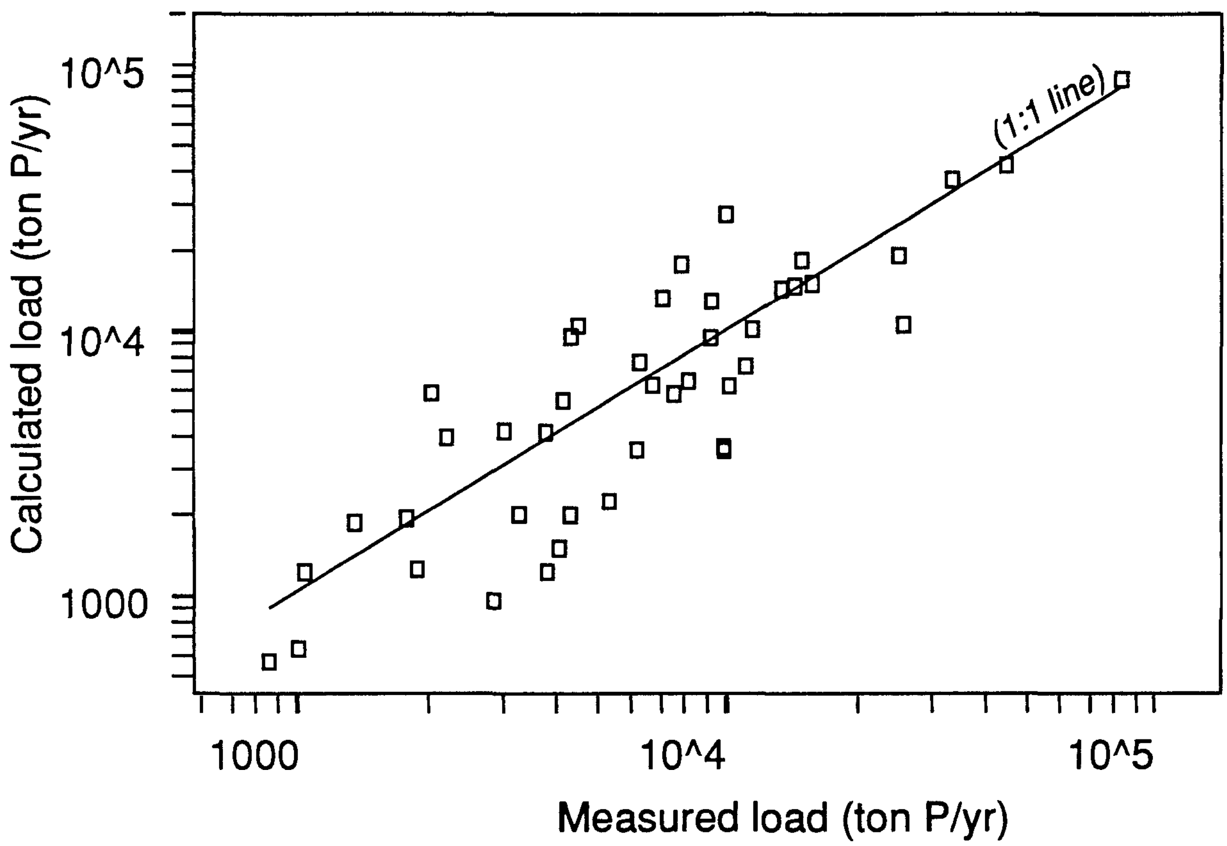

Figure 8. Calculated and observed loads of total phosphorus.

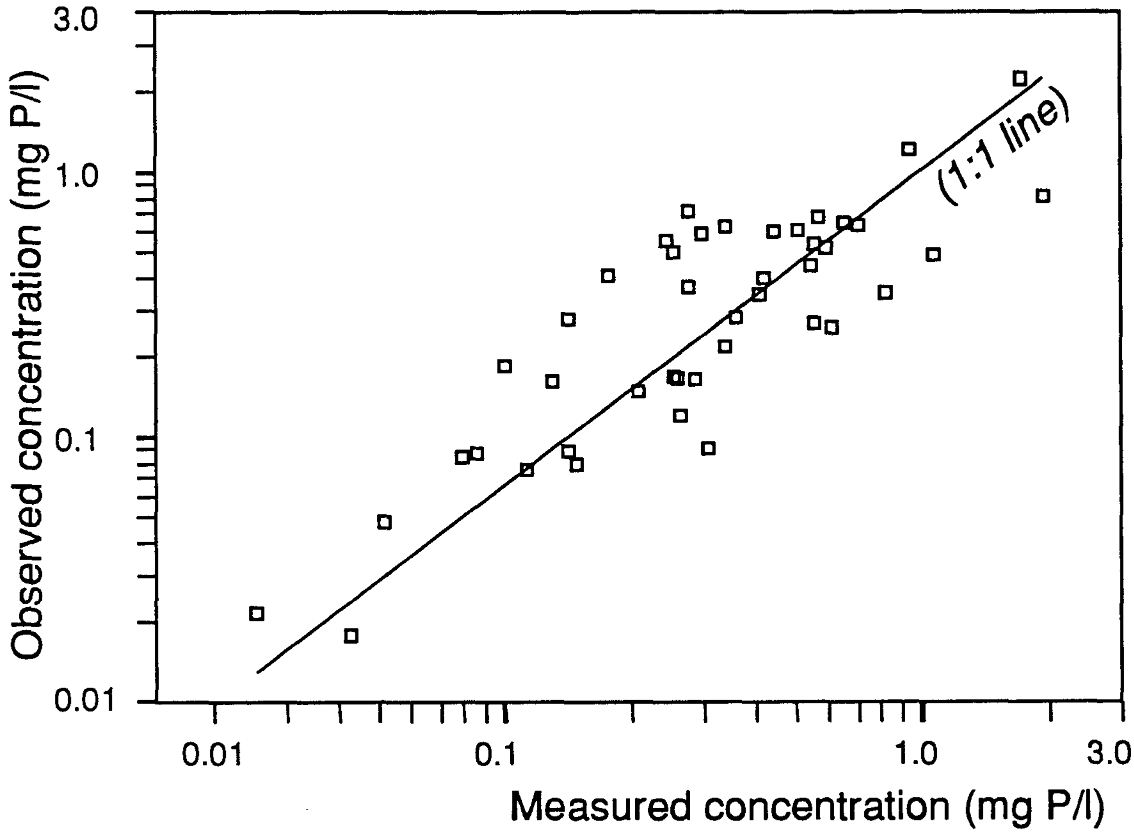

Figure 9. Calculated and observed concentrations of total phosphorus.

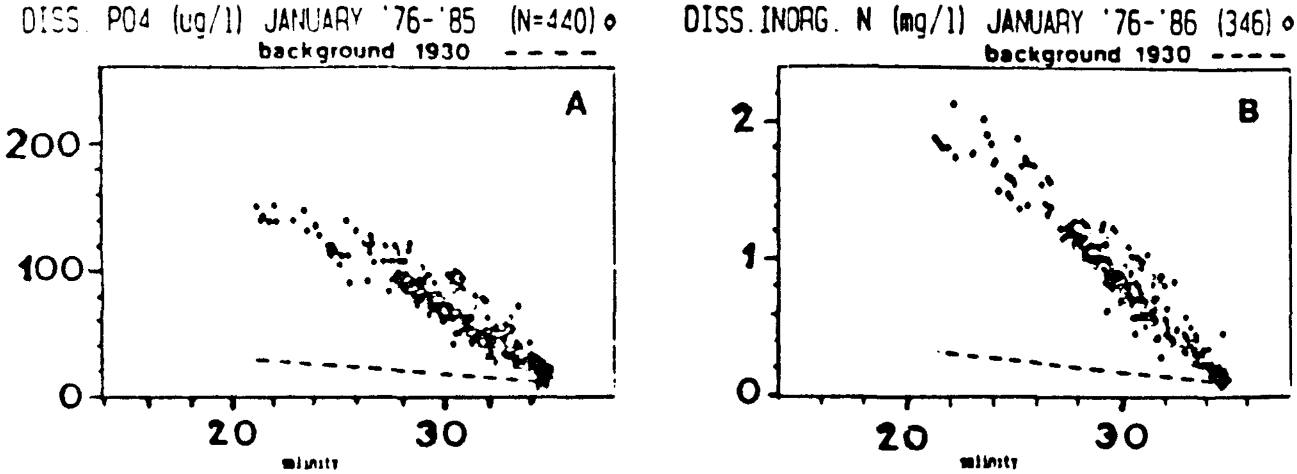

Figure 10. Dissolved phosphate (A) and dissolved inorganic nitrogen (NO_3+NO_2+NH_4) versus salinity in the Dutch coastal zone (from Zevenboom et al., 1990).

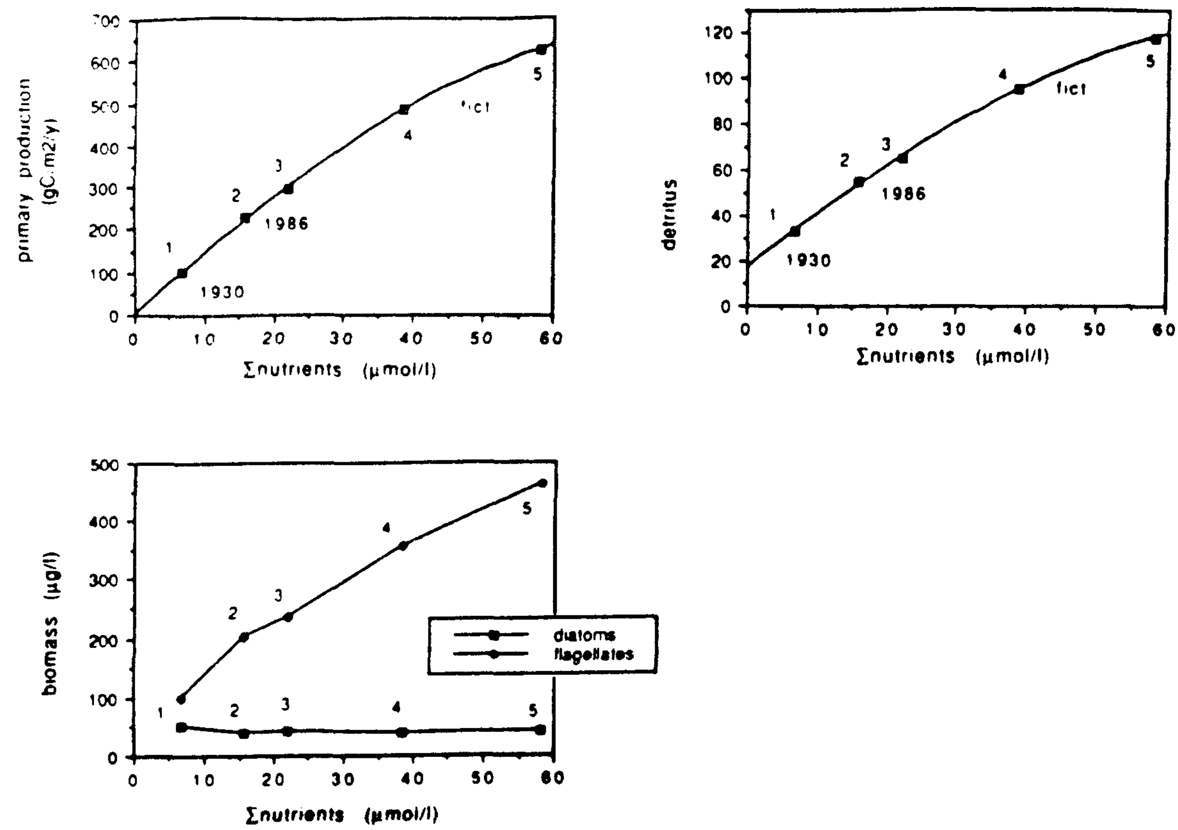

Figure 11. Ecosystem effects of North Sea pollution as calculated by the ERSEM model (from Hoppema and De Baar, 1991).

4. Concentrations in coastal seas

During winter months, phosphate and nitrate concentrations in the North Sea can be accurately predicted by fresh water fraction (calculated from salinity), i.e.: polluted river water is simply diluted by clean ocean water (Zevenboom et al., 1990, see Fig. 10). During the summer, this simple relation no longer holds, and nitrate is transformed to various other forms of nitrogen, temporary stored in water below the thermocline, etc. bottom sediments. As a first approximation however, there is a linear relation between nutrient inputs and effects (biomass, productivity, Fig. 11) which makes the calculation of nitrate concentrations *as if* these would behave conservatively a very good proxy to ecosystem effects.

For the coastal seas of North-Western Europe (Gulf of Biscay, Irish waters, North Sea, Baltic Sea and Norwegian coastal seas) there are simple advection-diffusion models from Jefferies and Steele (1989) and Hallstadius et al. (1987). The models differ slightly in details of schematization; both models are calibrated to observations of Caesium-137 (mainly originating from nuclear plants). In addition, there is a similar model of the Black Sea of Fonselius (1974) based on salinity data. For the remaining coastal seas (between eastern coast of Spain, Balearic Islands, Corsica and Sicily; the Adriatic and the Aegean seas) a similar schematization was defined for the present project. Model schematization of the coastal shelves is given in figure 12. Observations of salinity were obtained from the Levitus (1982) database; see appendix 2.

The freshwater fraction is determined by the freshwater input and exchange coefficients between the compartments and across the boundaries with the deep sea (Atlantic or Mediterranean). Discharge from rivers was discussed above. In addition, there is input from rain and output as evaporation. The Leemans and Cramer (1991) database that was used to calculate river discharge contains only data for land areas. Climatological data at sea were estimated by selecting all climatic data from the database with an elevation <10m, and interpolating across sea areas by inverse distance weighing. The resulting net input is given in appendix 2.

With these data one can calculate the salinity per compartment. In one-dimensional cases (for example, Adriatic sea, Gulf of Biscaye, etc.) this gives a unique value for the exchange coefficient that matches observed salinity. In the two-dimensional case (for example, North Sea), several solutions are possible: for example, one can exchange water with the Atlantic either as exchange via the Channel or via the northern boundary, or as an advective flow through the Channel. In a way, this ambiguity is no problem: for a given salinity the resulting nitrogen dilution will be approximately the same, as in all cases we are mainly diluting with constant-salinity constant-nitrogen Atlantic water. Small differences may occur however: for example, if we match our salinity by a high Channel flow, there will be more

Seine-derived nitrogen than if we obtain the same salinity by northern exchange. In order to obtain a unique solution, a constrained optimization was performed: match observed salinities (1st priority) by staying as close as possible to the ^{137}Cs-derived flow pattern (2nd priority).

By replacing freshwater input with nitrate or phosphorus input, and replacing salinity boundary conditions by constant concentrations of 0.1 mg $N.l^{-1}$ and 0.01 mg $P.l^{-1}$, we can calculate potential nitrate and phosphate concentrations. The term potential indicates that all biological transformations and loss rates are disregarded. It should be further realized that only yearly average concentrations for rather large compartments are given: the model does not treat seasonal or sub-grid variability. Finally, the relation between potential concentrations and ecosystem effects should be treated with considerable caution: because of differences in climate (e.g. length of growing season) and physical conditions (e.g. stratification), the effect of a certain potential nutrient concentration may be quite different in different areas (e.g. North Sea vs. Adriatic Sea).

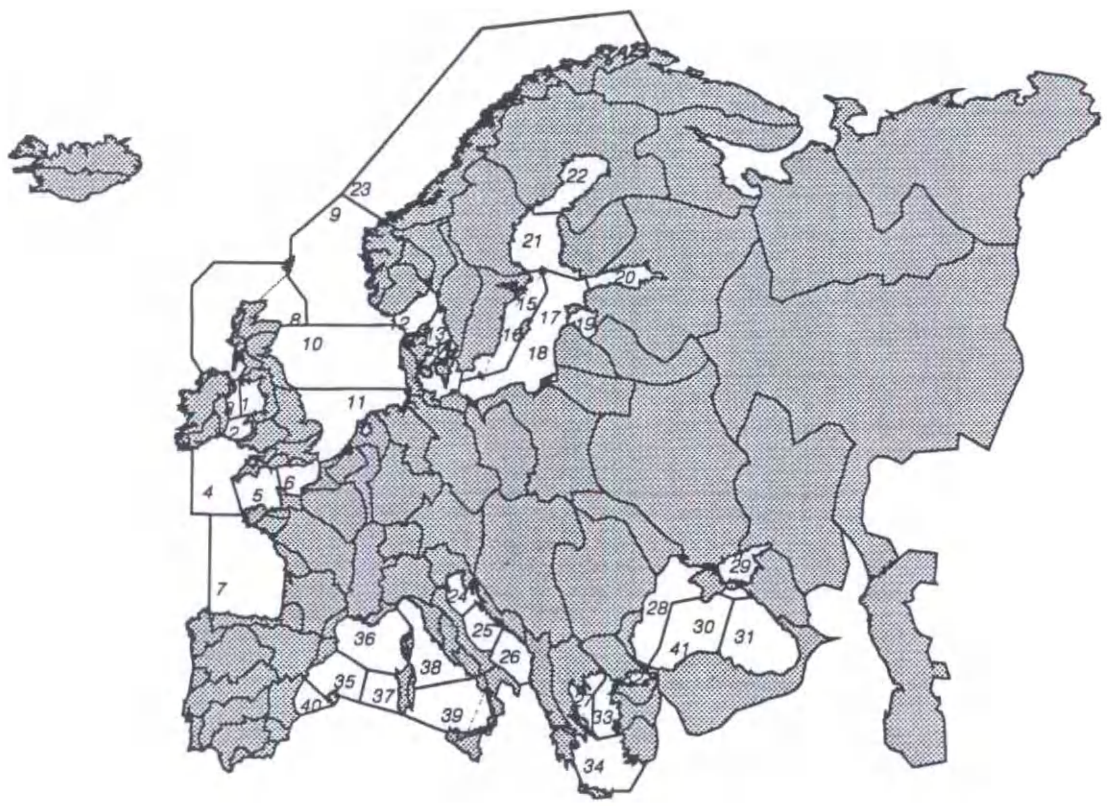

Figure 12. Coastal sea compartments with numbering.

5. Results

At present, the emphasis in the project has been on model formulation and calibration. The formulation of explicit policy-directed instruments is still under development. For the present situation we are able to show maps of the current status of water quality in the three major compartments: ground water (Meinardi et al., 1994b), fresh surface water and coastal seas.

Fresh surface water (Fig. 13, 14) shows two major problem areas in Europe: the North-Western part of Europe where low quality is mainly related to high inputs (intensive agriculture, high population density) and some areas of the Mediterranean, where low quality is not so much due to high inputs (per unit area they are lower than in Western Europe), but to the combination with relatively low water discharges, which gives a much smaller dilution of the pollutant load.

Seawater quality shows two major problem areas: the Baltic Sea and the Black Sea/ Sea of Azov (Fig. 15, 16). In comparison with the North Sea, absolute loads are not excessive, but a major problem is the low exchange of these brackish seas with unpolluted oceanic or mediterranean waters.

Obviously, these results correspond to well-known facts. Once we have established that the model is able to reproduce these, at least in the correct order of magnitude, we can use it for subsequent exercises. As an example, the contribution of atmospheric deposition of N to the increase relative to background levels was calculated. It can be observed that this contribution is considerable, even in polluted areas (Fig. 17). In this fashion we can use the model to establish the contributions of various countries, economic sectors, etc., to a particular effect. Conversely, one can calculate the effect of various abatement strategies.

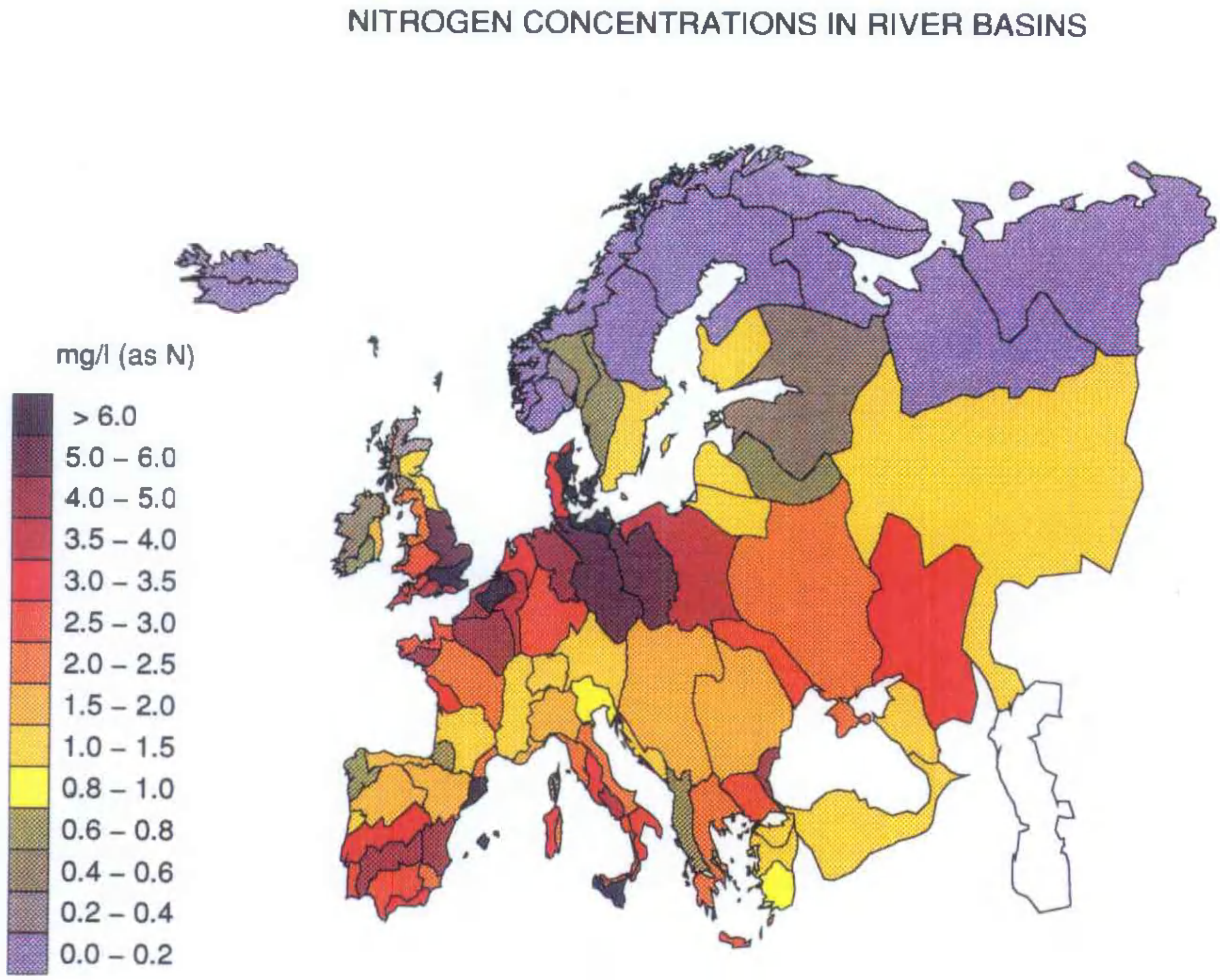

Figure 13. Fresh water phosphorus concentration

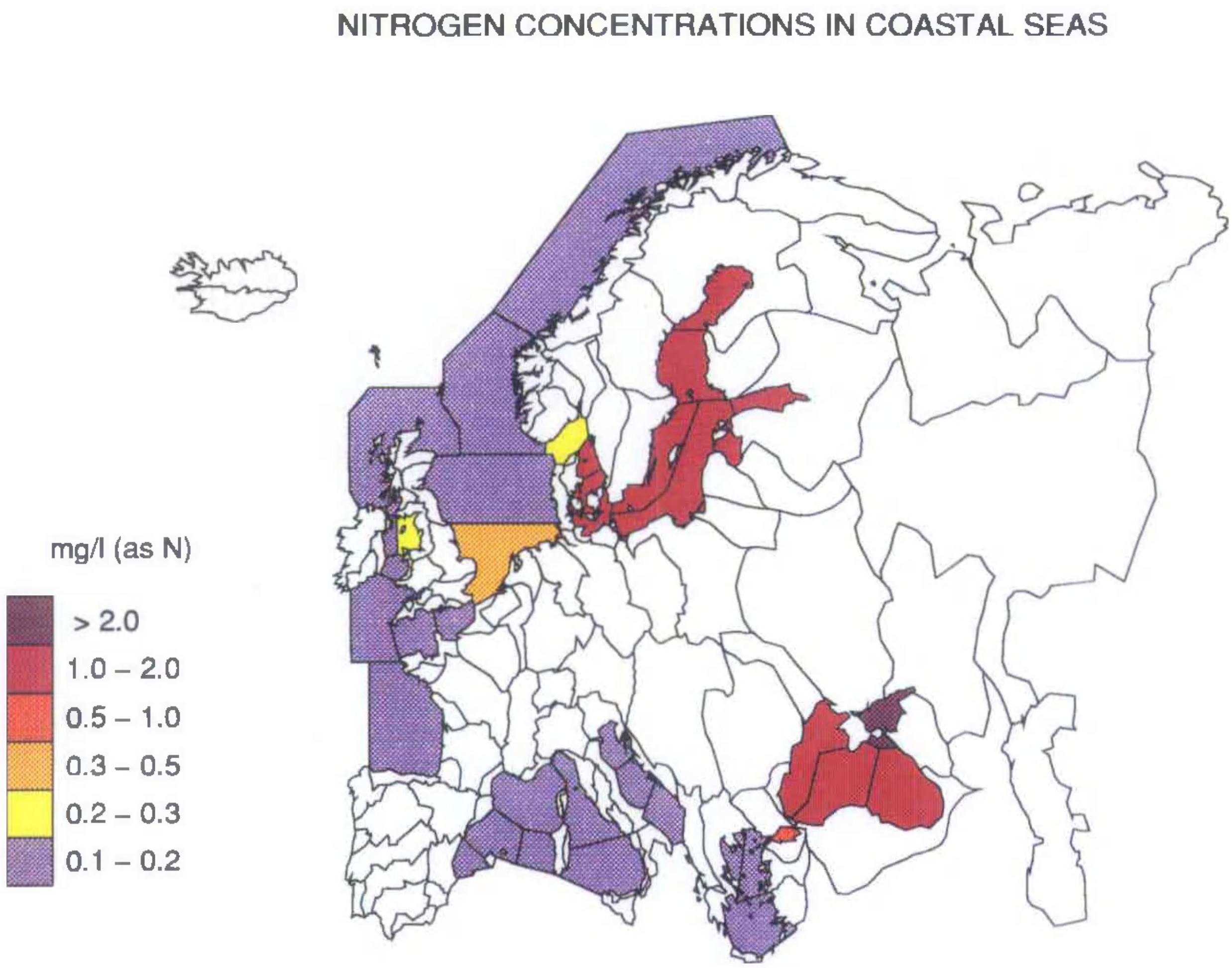

Figure 14. Fresh water nitrate concentration.

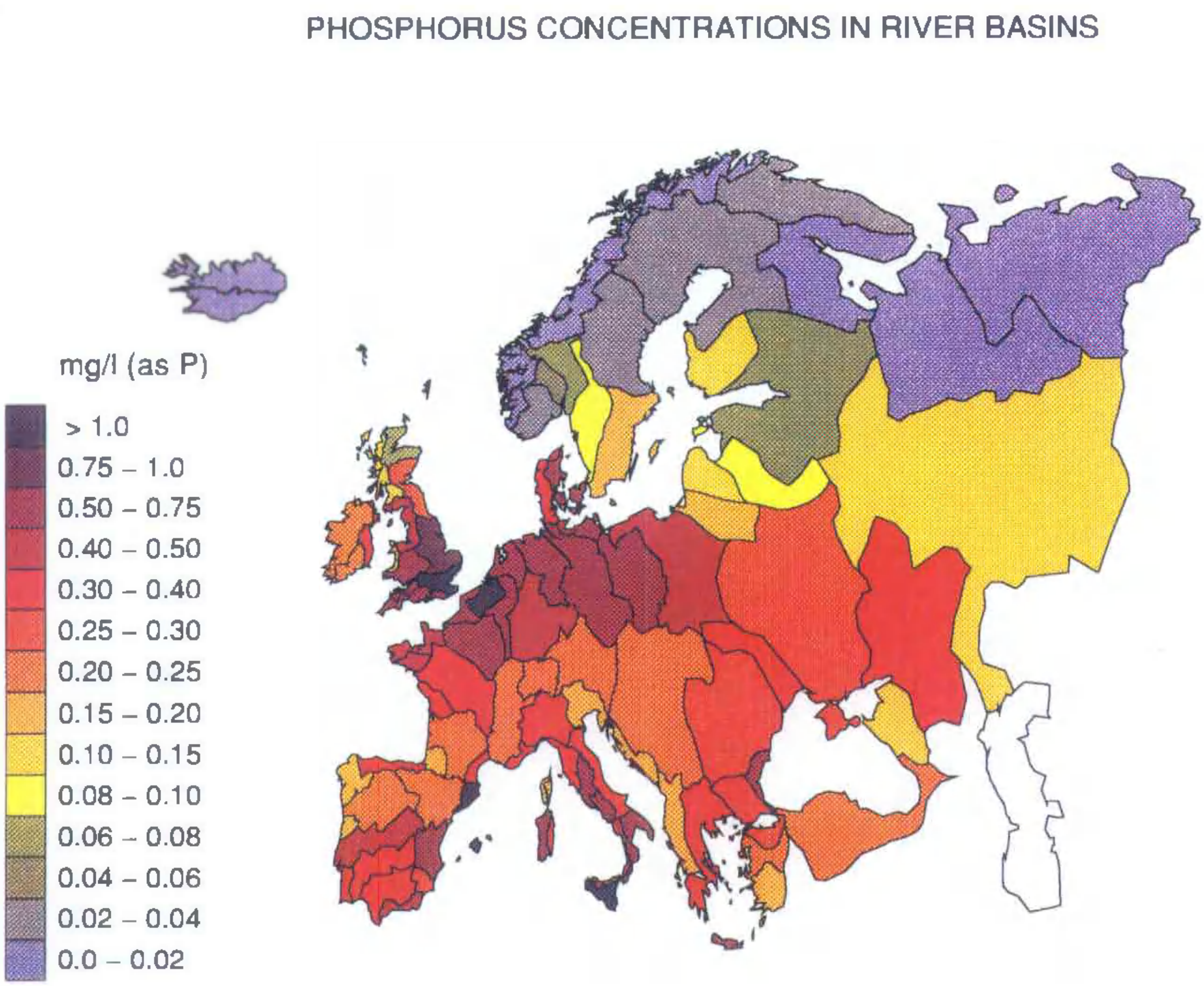

Figure 15. Potential phosphorus concentration in coastal seas.

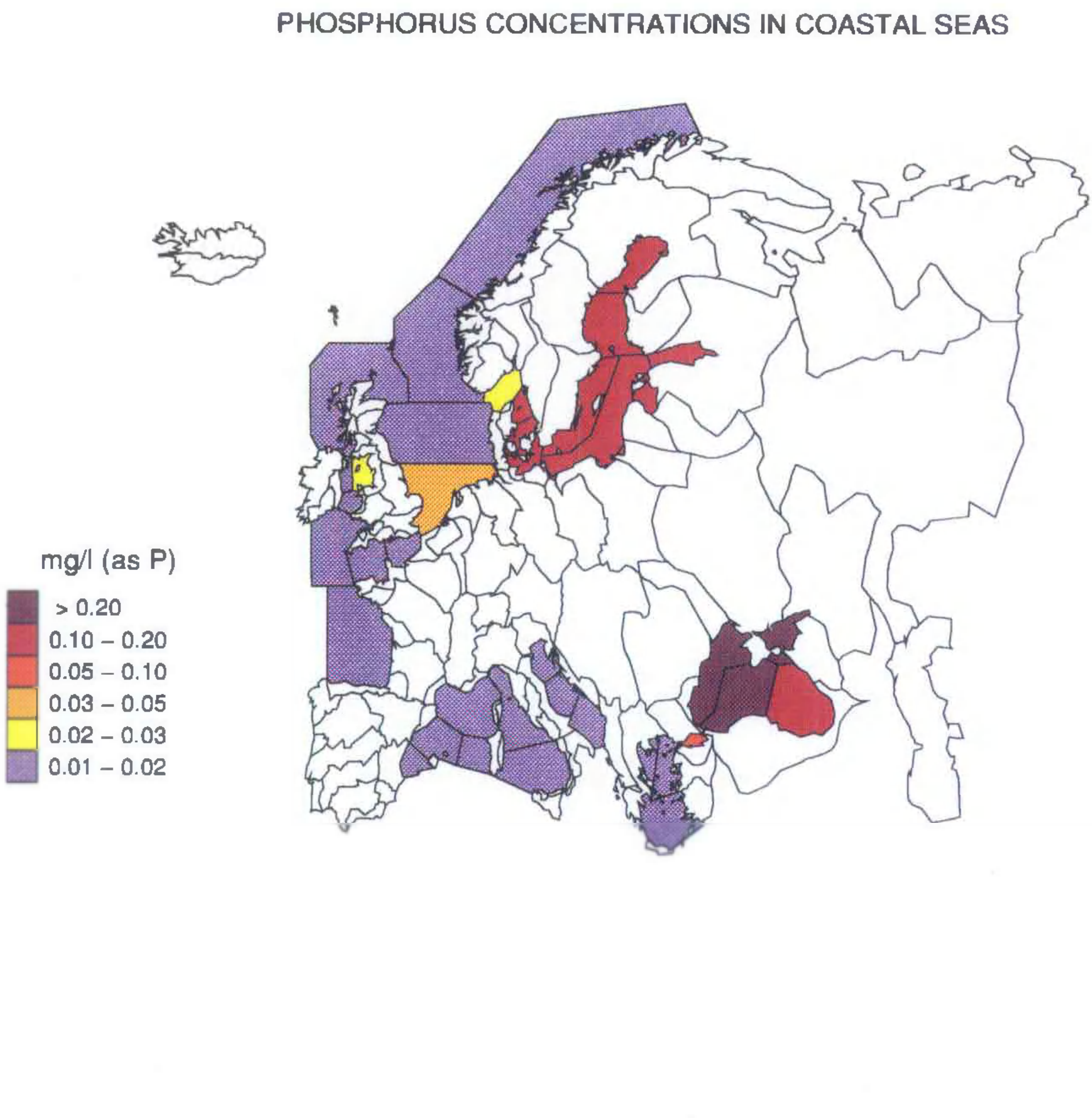

Figure 16. Potential nitrate concentration in coastal seas.

FRACTION ATMOSPHERIC DEPOSITION IN COASTAL SEAS

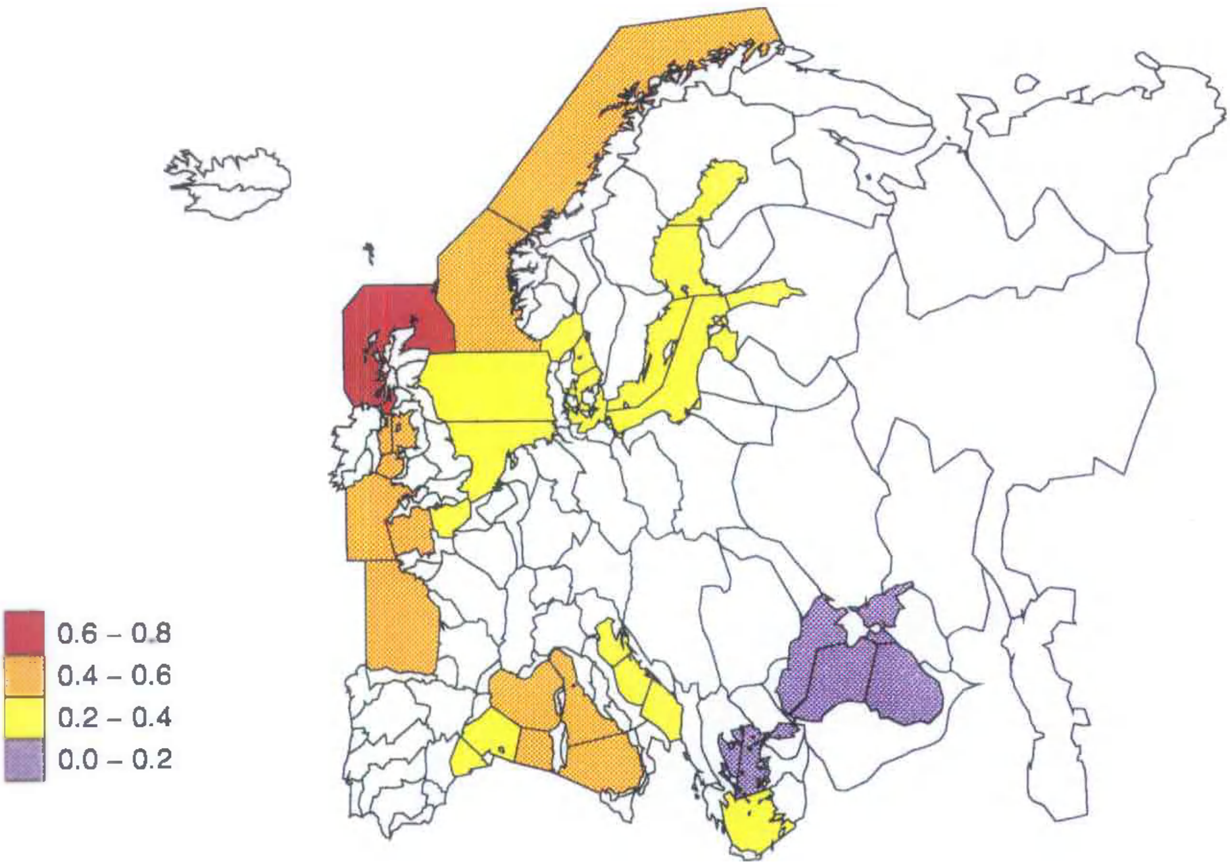

Figure 17. Contribution of atmospheric deposition to increase in nitrate concentration relative to background.

6. Conclusion

The present results show that it is possible to calculate N- and P loads on a European scale with a relatively simple model. The main use of the model appears to be the development and testing of European-scale policies: what is the contribution of a particular source area to a certain nutrient load? Which measures (wastewater treatment, agricultural policies) are the most effective to ameliorate the situation? Although local models (a particular river or coastal sea) may be fine-tuned to the particular situation and give locally more detailed answers, it seems that for the European-scale applications the present approach may provide a useful tool for the screening of policy alternatives. It is recommended to incorporate some quantitative information on temporal and spatial variability into the modelling approach in order to further increase the model's applicability for policy analysis.

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Appendix 1 The average phosphorus and nitrogen inputs on agricultural land in Europe in 1989.

	Phosphorus input (kg/ha/yr) (as P)		Nitrogen input (kg/ha/yr) (as N)	
	fertilizer	manure	fertilizer	manure
Albany	5.0	7.3	63.0	40.5
Austria	4.4	7.1	37.8	39.2
Belgium	12.5	22.3	129.2	122.7
Bulgaria	5.3	5.3	67.8	29.4
Denmark	6.9	10.9	129.4	60.2
Finland	10.0	5.5	81.9	30.3
France	9.6	6.9	78.3	37.9
Germany	11.3	12.5	115.9	66.6
Great Britain	5.1	7.4	83.4	40.7
Greece	4.4	3.1	43.4	16.9
Hungary	4.3	4.6	84.2	25.3
Iceland	2.0	0.7	81.0	4.0
Ireland	5.3	7.7	57.3	42.2
Italy	7.7	5.7	54.9	31.5
Luxembourg	12.5	22.3	142.9	89.3
Netherlands	8.0	36.9	243.8	203.6
Norway	7.5	13.3	108.8	73.2
Poland	3.9	6.2	66.5	34.4
Portugal	4.4	6.4	43.2	35.3
Rumania	4.6	6.4	48.0	35.4
Russia	2.9	2.0	32.0	11.0
Spain	3.8	2.7	33.7	14.9
Sweden	3.7	4.7	65.1	25.9
Switzerland	4.1	8.6	33.6	47.5
Czechoslovakia	11.7	7.6	90.6	41.8
Turkey	3.7	2.6	26.7	14.5
Yugoslavia	3.0	4.3	34.1	23.9
	Source: FAO (1994)		Source: RIVM 1992	

Appendix 2. Basic data for the 41 sea compartments. Compartments 16 underlies 15, 18 underlies 17 and 41 represents the deep Black Sea. The "p-e" columns give precipitation minus evaporation both per unit area (mm) and total. The "net" column gives total freshwater input into the compartment. Reference salinity is the salinity boundary condition relevant for that particular compartment (North Atlantic, South Atlantic, Western, different parts of the Mediterranean) as given by Levitus (1982).

sea	area (1000 km2)	river discharge (km3/yr)	p-e (mm/yr)	p-e (km3/yr)	net (km3/yr)	volume (km3)	salinity (o/oo)	reference salinity (o/oo)
1	21.83	16.7	47	1.0	17.7	750		
2	13.19	3.8	44	0.6	4.4	1000		
3	12.81	2.5	45	0.6	3.1	800		
4	136.37	27.6	41	5.6	33.2	20000	35.19	35.40
5	34.10	12.7	39	1.3	14.0	3200		
6	31.60	28.1	92	2.9	31.0	1300		
7	247.79	99.8	36	9.0	108.8	330000	35.39	35.60
8	186.21	39.9	139	25.9	65.8	13000	35.16	35.30
9	209.96	56.2	158	33.1	89.3	56000	34.81	35.20
10	212.58	25.6	189	40.2	65.8	14000	34.66	35.30
11	147.24	175.4	211	31.0	206.4	5000	32.79	35.30
12	37.20	57.6	44	1.6	59.2	7237	31.98	35.30
13	15.08	18.8	-27	-0.4	18.4	515	8.51	35.30
14	49.23	11.4	17	0.8	12.3	1000	8.33	35.30
15	93.22	19.4	7	0.7	20.1	3800	7.52	35.30
16		0.0	0	0.0	0.0	770		
17	121.74	117.4	81	9.9	127.3	7000	7.02	35.30
18		0.0	0	0.0	0.0	1500	12.53	35.30
19	22.92	37.8	182	4.2	42.0	400	4.80	35.30
20	32.45	120.2	156	5.1	125.3	1100	5.82	35.30
21	75.86	63.5	50	3.8	67.3	4900	5.51	35.30
22	50.90	74.3	65	3.3	77.6	1500	4.24	35.30
23	225.44	64.3	87	19.7	84.0	100000	34.61	35.10
24	35.15	85.1	-174	-6.1	79.0	1700	37.98	38.40
25	54.30	23.5	-172	-9.4	14.1	4600	38.21	38.40
26	50.85	30.1	-172	-8.8	21.4	16000	38.28	38.40
27	33.25	21.7	-372	-12.4	9.4	6700	37.79	39.00

Appendix 2. Basic data for the 41 sea compartments. Compartments 16 underlies 15, 18 underlies 17 and 41 represents the deep Black Sea. The "p-e" columns give precipitation minus evaporation both per unit area (mm) and total. The "net" column gives total freshwater input into the compartment. Reference salinity is the salinity boundary condition relevant for that particular compartment (North Atlantic, South Atlantic, Western, different parts of the Mediterranean) as given by Levitus (1982).

sea	area (1000 km2)	river discharge (km3/yr)	p-e (mm/yr)	p-e (km3/yr)	net (km3/yr)	volume (km3)	salinity (o/oo)	reference salinity (o/oo)
28	114.78	461.1	-364	-41.8	419.3	7000	16.50	39.00
29	53.64	103.8	-275	-14.8	89.0	1200	11.00	39.00
30	144.24	29.6	-329	-47.5	-17.8	22000	17.00	39.00
31	149.69	29.6	-285	-42.6	-13.0	23000	17.98	39.00
32	11.70	4.5	-372	-4.4	0.2	1700		
33	52.44	19.1	-372	-19.5	-0.4	12000	37.79	39.00
34	128.59	18.0	-372	-47.9	-29.8	63000	38.81	39.00
35	72.61	19.6	-51	-3.7	15.9	72000	37.80	37.50
36	118.86	65.1	-162	-19.3	45.8	230000	37.91	37.50
37	47.30	3.0	-191	-9.0	-6.1	120000	37.39	37.50
38	82.41	28.3	-187	-15.4	12.9	87000	37.88	37.50
39	138.19	7.0	-180	-24.8	-17.9	270000	37.80	37.50
40	28.58	3.7	15	0.4	4.2	14000	37.69	37.50
41		0.0	0	0.0	0.0	420000	22.50	39.00