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The contribution of agriculture to nutrient pollution in three European rivers, with reference to the European Nitrates Directive

This paper summarises the results of a large-scale analysis of nitrogen and phosphorus fluxes in the Rhine, Elbe, and Po basins. The average contribution of agriculture to the total nutrient load in these rivers is estimated around 45% for nitrogen and 20% for phosphorus. A simulation of measures imposed by the European Directive on Nitrates from agricultural sources suggests that this directive may not be stringent enough to substantially reduce the nutrient loads in these rivers.

Introduction

The flux of nutrients from land to rivers and seas has increased with time by human activities [27, 21]. This has caused ecological changes in fresh and marine waters [30, 31] and has negatively affected the quality of water for human consumption and other uses. There is a general perception that now that discharges of polluted wastewater from households and industry are being reduced, agriculture is becoming the main source of nutrient inputs to fresh and marine waters in Europe [28, 26]. The principal cause of agricultural nutrient pollution in Europe is the input of nutrients to agricultural land (fertilisers and manure) exceeding the output of nutrients from agricultural land (crop yield). The import of fodder and fertilisers from outside Europe maintain this unbalanced system. The agricultural surplus of nutrients may potentially runoff to the aquatic environment.

Many studies have analysed the relation between agricultural activities and nutrient inputs to the aquatic environment [e.g. 29, 7], but only few studies have analysed this issue at large spatial and temporal scales. Such large-scale analyses are needed in order to design, monitor, and evaluate large-scale policies that aim at a reduction of nutrient levels in large river systems and coastal seas. It is not feasible to measure (diffuse) agricultural inputs of N and P to the surface water for large areas. Therefore, large-scale analyses of agricultural nutrient pollution are most often based on extrapolations from small-scale studies [32, 33], or on large-scale nutrient balances [4, 1, 6, 3].

This paper analyses the contribution of agriculture to the nitrogen (N) and phosphorus (P) loads in the Rhine, Elbe, and Po rivers. The analysis is based on an extensive geo-referenced database on nutrient sources (e.g. livestock numbers, data on wastewater treatment), physical characteris-

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tics of landscape and climate and measured river nutrient loads for the Rhine, Elbe, and Po river basins. These data cover the period 1970-1995 and have been used to validate two nutrient balance models that simulate the transfer of N and P from pollution sources to river outlets. Both models are used to quantify the contribution of agriculture to the nutrient load in the Rhine, Elbe, and Po rivers for the period 1970-1995, and to simulate the effect of the implementation of the European Directive on Nitrates from Agricultural Sources [16] on the annual average nutrient load in these rivers for the period 2015-2020. The difference in the outcome of the two models is indicative for the accuracy of our results. Moreover, the results have been compared with the results of other workers, allowing for a critical evaluation of the calculated contribution of agriculture to nutrient pollution of European rivers and coastal seas.

Material and methods

The Rhine, Elbe, and Po river basins

The Rhine, Elbe, and Po river basins cover an area of approximately 400,000 km^2 of which about 45 percent is used for agricultural production. The three river basins have a total human population of around 85 million people and they overlap with the borders of eleven different countries (Figure 1). The Rhine basin is located in Western Europe, the Elbe basin in Central Europe and the Po basin in Southern Europe. Together these basins cover a wide range of landscape, climatic, and socio-economic zones. The average nutrient concentrations in these rivers are approximately ten times larger than in rivers located in sparsely populated areas of Europe (Table 1). Several studies reported that agriculture is the main source of the river nutrient pollution in this part of Europe (Table 2).

Table 1: Average N and P concentrations in the Rhine, Elbe, Po, and other European rivers

River	Period	$N\text{-}NO_3$ (mg l^{-1})	P_{tot} (mg l^{-1})	Reference
Rhine	1990-1995	3-4	0.2-0.3	IAWR (1995)
Elbe	1990-1992	4-5	0.3-0.4	IKSE (1993)
Po	1993-1995	2-3	0.1-0.2	Caggiati et al. (1997)
All European rivers	+/-1980-1995	2.6	0.3	Stanners & Bourdeau (1995)
Near pristine European rivers	+/-1980-1995	0.3	0.03	Stanners & Bourdeau (1995)

Table 2: Calculated contribution of agriculture to surface water pollution

Region	Year	N (%)	P (%)	Reference
North Sea basin	1987	60	25	Lidgate (1987) in Isermann (1990)
West Germany	1989	50	40	Isermann (1990)
West Germany	1987/1989	46	30-40	Werner et al. (1991)
East Germany	1988/1989	42	25	Werner & Wodsak (1994)
Italy	1986	62	33	Gaggino et al. (1986) in Isermann (1990)
Po river	1990	56	32	Italian Ministry of Environment (1992) in Stanners & Bourdeau (1995)
The Netherlands	1989	68	21	RIVM (1992) in Stanners & Bourdeau (1995)
The Netherlands	1990	60	40-50	Boers (1996)
Denmark	1981	81	22	Christensen et al. (1993) in Stanners & Bourdeau (1995)
Poland	1990	62	34	MEPNRF (1991) in Stanners & Bourdeau (1995)

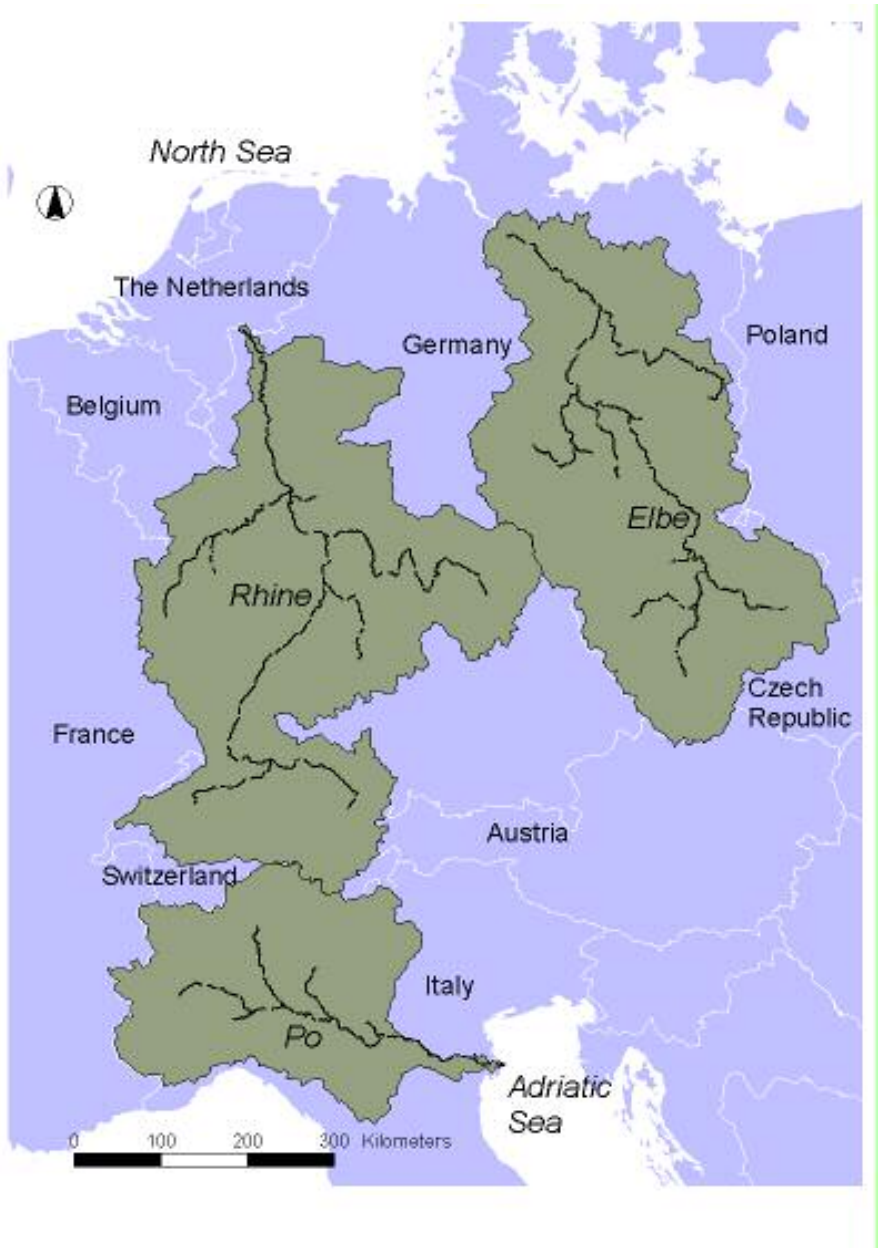

Fig. 1: Location of the Rhine, Elbe, and Po basins
The Dutch part of the Rhine basin has been excluded, because the river Rhine splits into three watercourses after crossing the Dutch/German border. This complicates the analysis of the relation between upstream sources and downstream pollution.

Modelling nutrient fluxes

In previous studies two models were reported that simulate five-year average nutrient fluxes from pollution sources to river outlets [10,13]. The basic structure of both models is presented in Figure 2 and can be described as:

$$L_x = a \cdot (DE_x + IE_x) \quad (1)$$

and

$$IE_x = b \cdot SSS_x \quad (2)$$

where L_x is the average river load at location x (kg·year^{-1}), DE_x is the average direct emission to the river network upstream x (kg·year^{-1}), IE_x is the average indirect emission to the river network upstream x (kg·year^{-1}), SSS_x is the average surplus at the soil surface upstream x (kg·year^{-1}), a is the fraction transferred through the river network *(-)*, and b is the fraction transferred via the soil/groundwater system *(-)*.

The two models differ in the way the transport, retention, and loss of nutrients in the soil/groundwater system (b) and river network (a) is described. The first model is a simple lumped model. This model describes the river nutrient load (L) as a function of nutrient inputs in the upstream basin (DE and SSS). The fraction of the emitted nutrients that reaches the outlet of the river (a) is positively related to the area specific runoff in the upstream basin, and the ratio of transport through the soil/groundwater system (b) is larger for regions with consolidated rocks than for regions with unconsolidated rocks. This model has been reported in De Wit (1999b) and is based on the work of [2] and [11]. The second model (PolFlow) is a GIS-based model, which simulates the transport of nutrients from soil to surface water (b) as a function of soil, lithology, and runoff characteristics. Dynamic functions are used to account for the delay of nutrient transport in the soil and the groundwater. Nutrients are routed through the river network with digital drain direction maps [34]. The nutrient loss in each river segment (a) is described as a function of discharge, the occurrence of lakes, and river gradients. PolFlow consists of a hydrological module [12] and a nutrient module [13].

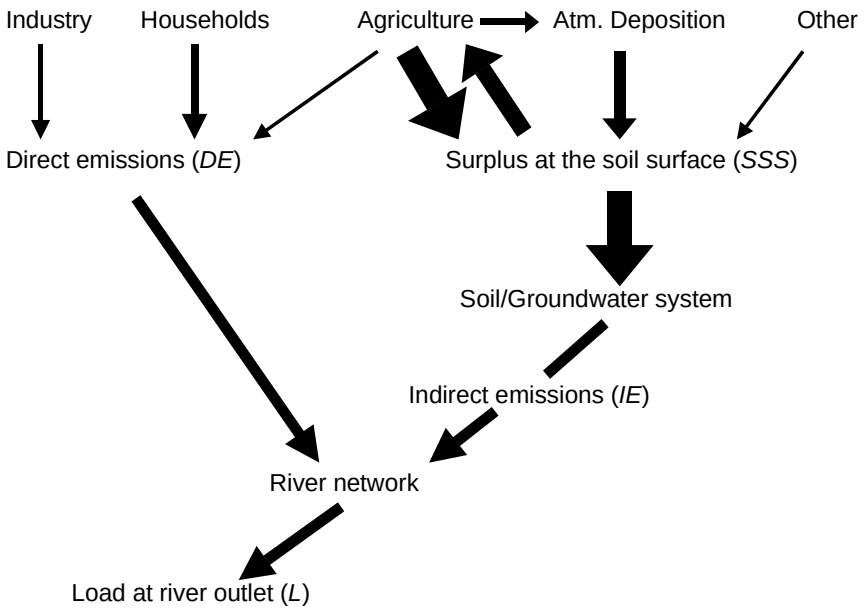

Fig. 2: Nutrient fluxes from pollution sources to river outlet
All inputs to the surface water (river network) are divided into direct emissions (DE) and indirect emissions (IE). Direct emissions (mainly point sources) are not in contact with the soil/groundwater system, whereas indirect emissions (the major part of the inputs from diffuse sources) enter the surface water via the soil/groundwater system (e.g. erosion, groundwater flow). The indirect emissions are first of all estimated as the surplus at the soil surface (SSS), the amount that can potentially runoff to the surface water.

Nutrient pollution sources (DE and SSS)

Emission estimates were used as input (*DE* and *SSS*) for the two transport models. The emission estimation methods are described in detail in [9] and [15]. The contribution of agricultural sources to these emissions is specified as respectively DE_{agr} and SSS_{agr}. The average agricultural surplus at the soil surface (manure production + fertiliser consumption - crop yield) of N and P were estimated for all five-year periods between 1970-1995 for the entire Rhine, Elbe, and Po basins. These estimates are based on coefficients (e.g. livestock excretion coefficients, and nutrient uptake for the different crops) applied to regional data on livestock numbers, fertiliser consumption, and crop yields. Figure 3 shows national numbers for four countries. For the estimates used in this study more detailed administrative data were used (the average size of the administrative units used is 1,000 km^2). Also non-agricultural emissions were estimated, both direct inputs to the surface water (e.g. wastewater from households and industry) and inputs at the soil surface (e.g. atmospheric deposition). A land cover map (resolution one square kilometre) was used to allocate the emissions within the administrative units.

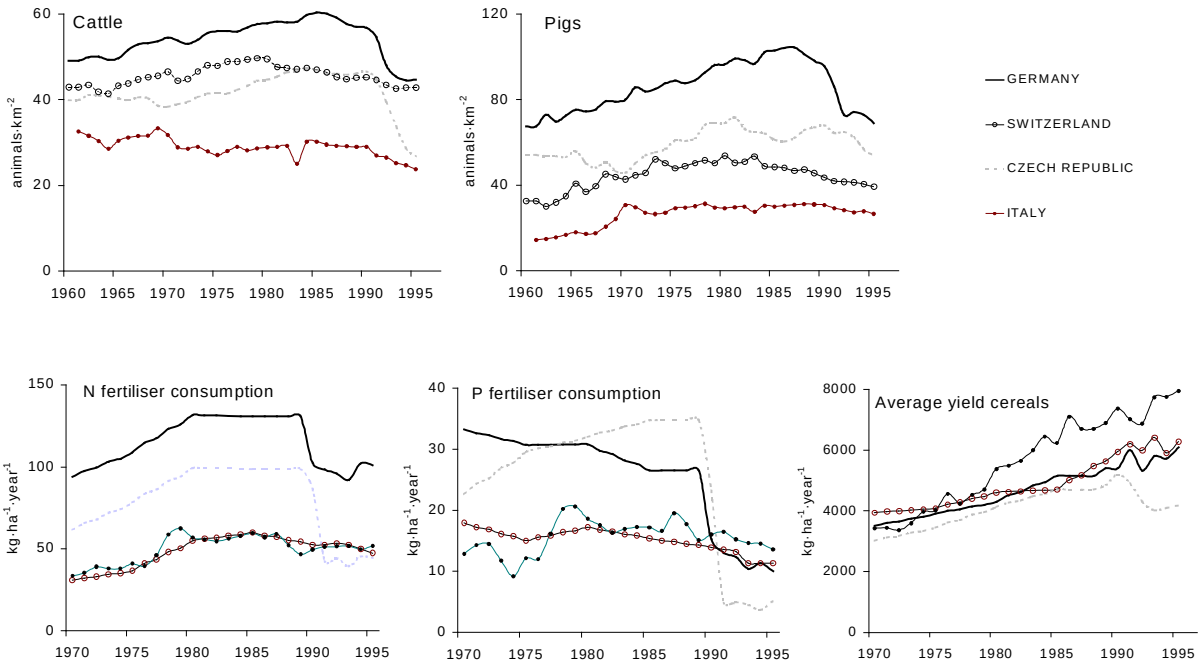

Fig. 3: Livestock density, fertiliser consumption, and crop yields (source FAO, various years)
Fertiliser use and livestock densities decreased drastically after the political changes in Eastern Europe in 1989/1990. This is reflected in the numbers for the Czech Republic and (Eastern) Germany.

Model performance

Both models have been applied to the entire Rhine, Elbe, and Po basins for the period 1970-1995. The modelled five-year average river loads (L) have been validated with river loads derived from discharge, and N and P concentration measurements from sixty different monitoring stations all over the Rhine, Elbe, and Po basins [9,14,15]. Both models explained most of the observed spatial and temporal variation in N and P river loads. Although the second model is more complex and uses more physically based descriptions of nutrient transfer than the first model, it appeared that the river load predictions of this second model were not any better than the river load predictions of the first model [9,14]. In this paper both models are used to specify the contribution of agriculture to the river nutrient load. The difference in outcome of the two models is indicative for the accuracy of the results.

EU Nitrates Directive

In 1991 the European Commission Council has issued the European Directive on Nitrates from Agricultural Sources [16]. The directive is now included in the European Water Framework Directive [17]. The objective of the Directive on Nitrates from Agricultural Sources is to reduce water pollution (groundwater, lakes, rivers, and coastal seas) caused or induced by nitrates from agricultural sources and prevent further such pollution. In order to do so the states of the European Union have to identify waters affected by nitrate pollution or which might be affected in the near future if action is not taken. In these so-called vulnerable zones, the member states are required to establish measures. The most significant measures in the directive are:

i) the requirement for the land application of livestock manure to be limited to 170 kilogram of N per hectare per year for each farm,
ii) the requirement for the land application of fertilisers to be based on a balance between the requirements of the crops and the supply to the crops from the soil and from fertilisation,
iii) the requirement for each farm to have sufficient livestock manure storage capacity for the period when they are not permitted to apply the manure to the land and
iv) to draw up at least one code of good agricultural practice.

The directive should be implemented by the year 2002, but is severely behind schedule [19]. As a second step in this study the two models are used to simulate the effect on the river nutrient load in the Rhine, Elbe, and Po for the period 2015-2020 if the Nitrates Directive would be implemented by 2005-2010 all over the Rhine, Elbe, and Po basins. Only the first two measures will be evaluated, because the last two measures cannot be simulated with the models described above. The effect of the first two measures has been evaluated using the following three scenarios:

i) scenario ‘no changes’ assumes that the emissions (DE and SSS) do not change between 1990-1995 and 2015-2020,
ii) scenario ‘170 max’ assumes that all farms that currently apply more than 170 kilogram N of manure per hectare agricultural land will manage to comply with the Nitrates Directive in the period 2005-2010. All other emissions are assumed not to change and
iii) scenario ‘balance farming’ assumes that by the year 2015-2020 the annual agricultural surplus at the soil surface (manure production + fertiliser consumption – crop yield) does not exceed 25 kg of N per hectare agricultural land and 5 kg of P per hectare agricultural land. This might not be a realistic scenario, but it gives more or less the maximum possible effect of the second requirement in the Nitrates Directive.

All three scenarios are run for average climatic conditions. The Nitrates Directive is designed for N, but here also the possible effects for agricultural P pollution will be analysed.

Results

Contribution of agriculture to the river nutrient load: 1970-1995

The slight decrease of fertiliser consumption (especially for P) and the increase of crop yields (Figure 3) resulted in a decrease of the agricultural surplus of N and (especially) P in the Rhine and Po basins between 1985 and 1995 (Table 3 and Figure 4). In the Elbe basin (former Eastern Europe) a more drastic decrease of the agricultural N and P surplus is calculated, due to the reduction of fertiliser consumption and livestock numbers after the political and economical changes in 1989/1990 (Figures 3 and 4).

Figure 4 shows the spatial and temporal variation of the estimated agricultural N and P surplus at the soil surface (SSS_{agr}). The values in Table 3 are total values for the total emissions (*E*), direct emissions (*DE*), surplus at the soil surface *(SSS*), indirect emissions (*IE*), and simulated river loads (*L*) for the Rhine (upstream of Dutch/German border, 160,000 km^2), Elbe (upstream of Hamburg, 140,000 km^2), and Po (upstream of Ferrara, 70,000 km^2) river basins. The contribution of agricultural sources has been specified (L_{agr}, SSS_{agr}, IE_{agr}, DE_{agr}). For *IE*, E_{tot} and *L* two values are given, one value calculated with model 1 and one value calculated with model 2.
Table 3 shows that most of the variation in the river N and P load (*L*) between 1970 and 1995 can be explained by changes of direct emissions (*DE*). The differences between the model estimates for the indirect emissions (*IE*) are even larger than the differences for the indirect emissions (*IE*) between the different five-year periods. Therefore, no clear trend can be observed for the indirect emissions between 1970-1995. This is most striking for the Elbe basin, where both models react different to the sudden changes in 1989/1990. This explains why for the Elbe basin 1990-1995, the indirect emissions (*IE*) calculated with the two models differ considerable.

Table 3: Calculated total N and P fluxes (106 kg year^{-1})

Nitrogen

	Rhine			Elbe			Po		
	1970-1975	1980-1985	1990-1995	1970-1975	1980-1985	1990-1995	1970-1975	1980-1985	1990-1995
SSS_x	737	786	643	761	895	282	419	481	381
$SSS_{agr,x}$	561	621	482	596	743	136	329	395	299
IE_x	218-241	249-253	234-288	159-201	179-229	99-257	92-110	105-131	94-151
$IE_{agr,x}$	166-183	197-200	175-227	125-157	149-190	48-203	72- 86	86-108	74-123
DE_x	267	257	189	150	155	108			77
$DE_{agr,x}$	5	5	5	14	18	10			7
E_x	485-508	506-510	423-477	309-351	334-384	207-365			171-228
L_x	380-397	405-431	324-383	165-168	153-193	111-152			125-203
$L_{agr,x}$ %	35- 37	40- 40	43- 49	45- 49	50- 54	28- 58			47- 57

Phosphorus

	Rhine			Elbe			Po		
	1970-1975	1980-1985	1990-1995	1970-1975	1980-1985	1990-1995	1970-1975	1980-1985	1990-1995
SSS_x	213	166	82	214	200	4	116	140	111
$SSS_{agr,x}$	194	152	70	195	185	-8	104	131	104
IE_x	2.3-7.7	3.1- 9.7	3.6- 7.1	2.0-5.4	2.8- 6.9	3.2-4.8	1.1-4.7	1.6-5.2	2.0- 5.4
$IE_{agr,x}$	2.1-2.1	2.3- 2.8	1.9- 3.3	0.9-1.8	1.4- 2.6	0.7-2.9	1.0-1.6	1.5-1.9	1.8- 2.2
DE_x	65.7	48.4	17.7	30.6	28.5	12.9			8.8
$DE_{agr,x}$	2.0	1.8	1.3	2.3	2.5	0.9			0.9
E_x	68.0-73.4	51.5-58.1	21.3-24.8	32.6-36.0	31.3-35.4	16.1-17.7			10.8-14.2
L_x	46.1-53.0	44.8-38.5	16.8-14.6	11.8-12.3	9.9-14.3	5.5- 6.5			7.1- 9.9
$L_{agr,x}$ %	6 - 6	7 - 9	13 -21	9 -13	11 -16	9 -24			22 -25

L_x = average river load at location x (kg·yr^{-1}),
E_x = average emissions to the river network upstream x (kg·yr^{-1})
DE_x = average direct emissions to the river network upstream x (kg·yr^{-1})
IE_x = average indirect emissions to the river network upstream x (kg·yr^{-1})
SSS_x = average surplus at the soil surface upstream x (kg·yr^{-1})
x = Dutch/German border (Rhine), Hamburg (Elbe), and Ferrara (Po)
agr = agricultural

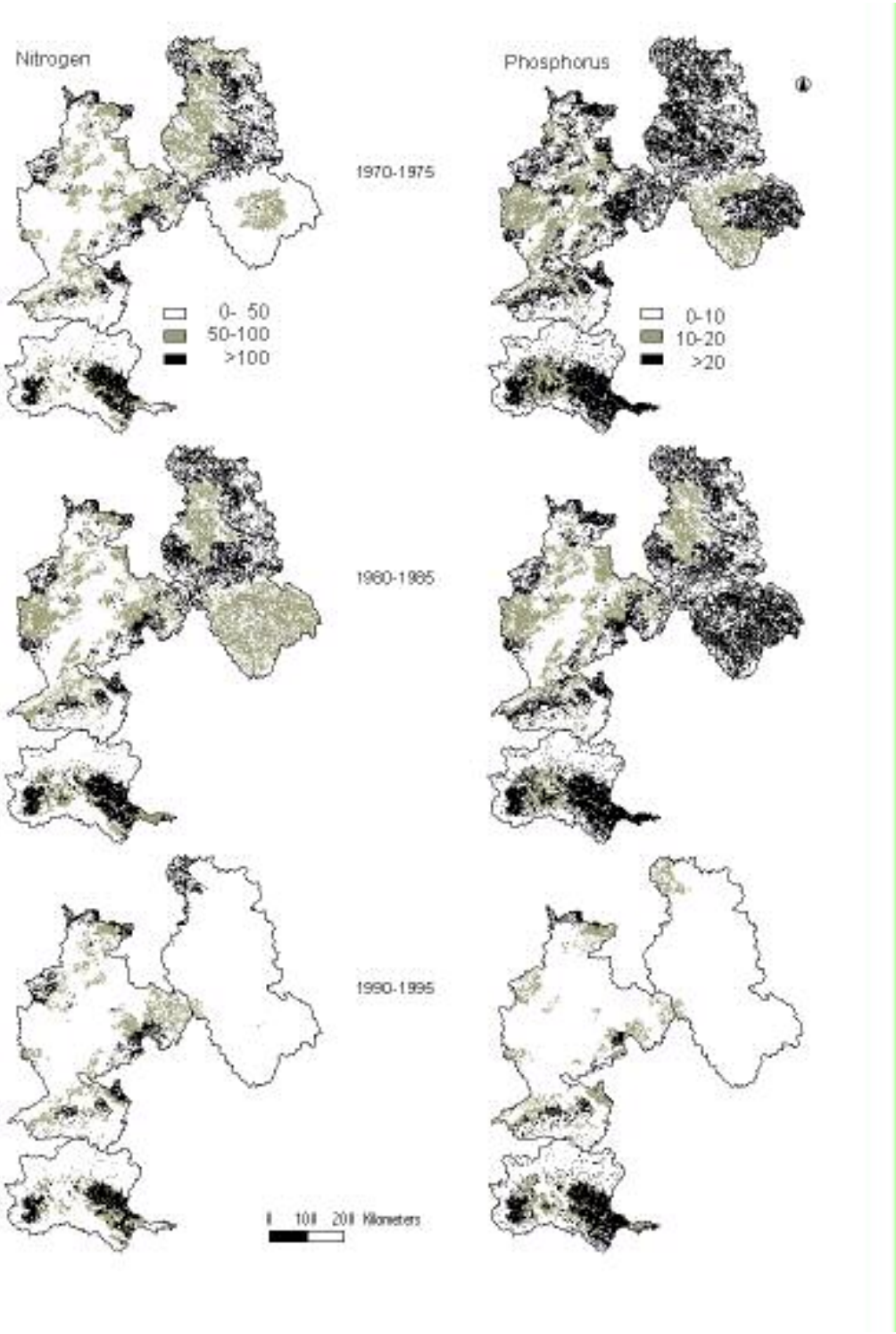

Fig. 4: Agricultural N and P surplus at the soil surface (SSSagr) (kg ha-1 year-1)
A value for SSSagr (manure production + fertiliser consumption – crop yield in kg ha-1 year-1) has been estimated for each km2 and each 5-year period. This figure gives a rough impression of the spatial and temporal distribution of SSSagr.

Simulation of the EU Nitrates Directive: 2015-2020
Figure 5 shows the areas where the average livestock manure application exceeds 170 kilogram N per year per hectare agricultural land for the period 1990-1995. It should be noted that this map is based on average data for administrative units (on average about 1,000 km^2). This means that within these units there might be farms where the livestock manure application is larger than the average application rate in the administrative unit. However, this map shows that only in a relatively small part of the total area of the Rhine, Elbe, and Po river basins the average livestock manure application rate exceeds the threshold given in the EU Nitrates Directive.

Table 4 shows the results of the three scenarios. From a comparison between scenario 'no changes' and scenario '170 max', it can be observed that the requirement for the land application of livestock manure to be limited to 170 kg N hectare per year for each farm hardly results in a reduction of the predicted river N loads. Scenario 'balance farming' leads to a substantial reduction of the river N load and a slight reduction of the river P load.

Table 4: Calculated effect of the requirement for the land application of livestock manure to be limited to 170 (kg N ha^{-1} $year^{-1}$) (scenario '170 max') and the requirement for the land application of fertilisers to be based on a balance between the requirements of the crops and the supply to the crops from the soil and from fertilisation ('scenario balance farming')

Nitrogen (10^6 kg·$year^{-1}$)

	IE_x			
Period	1990-1995	Scenario 'no changes' 2015-2020	Scenario '170 max' 2015-2020	Scenario 'balance farming' 2015-2020
Rhine	228-234	234-252	230-250	111-157
Elbe [a]	99-257	84- 99	84- 99	77- 91
Po	94-151	94-131	85-123	55- 86

	L_x					
Period	1990-1995	Scenario 'no changes' 2015-2020	Scenario '170 max' 2015-2020	Reduction Scenario '170 max' %	Scenario 'balance farming' 2015-2020	Reduction Scenario 'balance farming' %
Rhine	324-383	324-354	321-352	1	230-278	21-29
Elbe [a]	111-152	86-103	86-103	0	83- 99	3- 4
Po	125-203	125-194	118-178	6-8	96-145	23-25

Phosphorus (10^6 kg·$year^{-1}$)

	IE_x			L_x			
Period	1990-1995	Scenario 'no changes' 2015-2020	Scenario 'balance farming' 2015-2020	1990-1995	Scenario 'no changes' 2015-2020	Scenario 'balance farming' 2015-2020	Reduction Scenario 'balance farming' %
Rhine	3.6-7.1	4.6-8.6	4.1-7.8	14.6-16.8	15.5-17.6	15.0-17.2	2- 3
Elbe [a]	3.2-4.8	3.4-5.8	3.4-5.8	5.5- 6.5	5.6- 6.9	5.6- 6.9	0
Po	2.0-5.4	3.1-6.4	2.7-4.7	7.1- 9.9	7.6-10.9	6.8-10.5	4-10

L_x = average river load at location x (kg·yr^{-1}),
IE_x = average indirect emissions to the river network upstream of x (kg·yr^{-1})
x = Dutch/German border (Rhine), Hamburg (Elbe), and Ferrara (Po)

[a] Scenario 'no changes' may be unrealistic for the Elbe basin. Recent data of the FAO show that agricultural production in the former Eastern Europe is moving towards similar levels as in the former Western Europe. This means an increase of the fertiliser consumption compared to the situation just after the political changes in 1989/1990. The decrease of the indirect emission (*IE*) in the Elbe basin for scenario 'no changes' is caused by the slow response time of one of the two models.

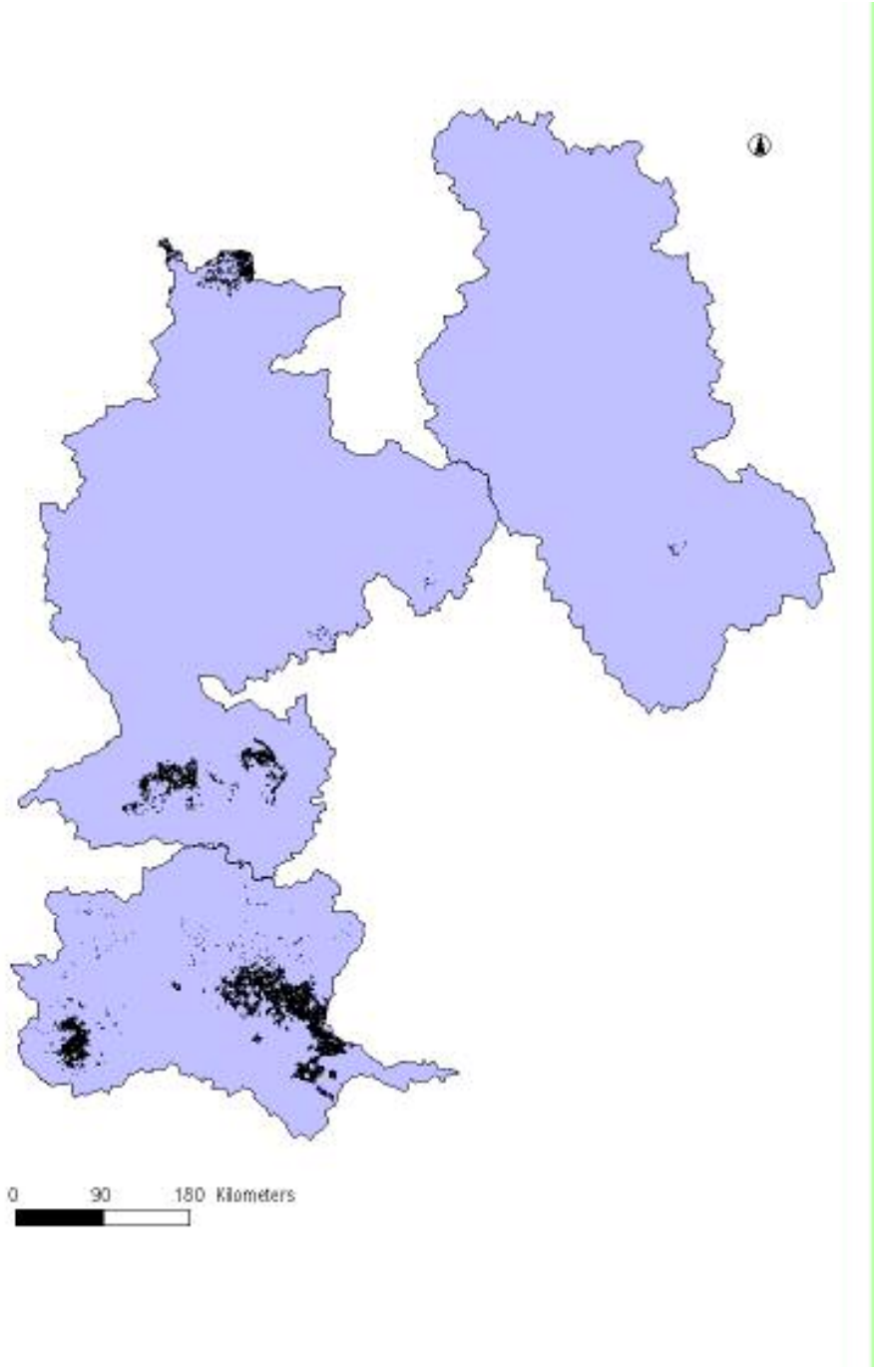

Fig. 5: Areas where the average manure application exceeds 170 (kg N ha-1 year-1) (1990-1995)

Discussion

Contribution of agriculture to the river nutrient load: 1970-1995

From a comparison between Tables 2 and 3 it can be observed that the estimates for the contribution of agriculture to the N and P river load (L_{agr}) are in general lower than the values reported in previous studies. Moreover, it appears that different studies (Table 2) and different models (Table 3) result in different estimates for the same regions. This suggests that one cannot precisely estimate the contribution of agriculture to the river N and P loads. There are a few possible explanations for the discrepancies between the different estimates.

Large-scale analyses of agricultural nutrient pollution are often based on extrapolations from small-scale studies. The most rigorous way to estimate nutrient fluxes for large areas is to apply export coefficients derived from small-scale studies to large-scale areas. However, there is a wide range of export coefficients for agricultural land published in literature [18] and it makes a large difference which export coefficients are used. Another way to analyse large-scale nutrient fluxes is to apply large-scale nutrient balances, which is done in this paper. Such methods describe the nutrient output (river load at basin outlet) as a function of the nutrient input (pollution sources in the upstream basin). The difference between inputs and outputs are related to losses in the soil/groundwater system and losses in the river network. The problem with this methodology is that both the estimates for the losses in the soil/groundwater system and the estimates for losses in the river network are hard to validate at large spatial scales. A model that estimates large nutrient losses in the soil/groundwater system and few nutrient losses in the river network, might predict the same river nutrient load as a model that estimates few nutrient losses in the soil/groundwater system and large nutrient losses in the river network. However, these two models will estimate different ratios between direct and indirect emissions. This explains part of the difference in the outcome of the two models used in this study.

Another reason to explain the deviation in the estimates is that it is often not clear what one considers as agricultural nutrient pollution and what not. A major part of the industrial nutrient inputs to the river network originates from fertiliser industry [25,24]. In this study these industrial emissions were not considered as agricultural emissions. A major part of the atmospheric deposition of N results from livestock breeding [28]. Here the agricultural related atmospheric deposition was considered to have agricultural sources, but other studies report atmospheric deposition as a non-agricultural source. Agricultural activities can also affect the transport of nutrients from soil to surface water by drainage [7] and erosion [29]. Especially for P it makes quite a difference whether one considers erosion of non-fertilised soils due to agricultural activities as agricultural P pollution or not. It also makes a difference whether one considers historical agricultural inputs at the soil. Up to 1990 the agricultural soils in Central and Eastern (and Western) Europe were heavily fertilised. After the political changes the fertiliser consumption dropped dramatically in Central and Eastern Europe [20].

For the period after 1990, it makes a large difference whether one estimates the indirect emissions of P to the surface water as a function of historical inputs at the soil surface or as a function of

actual inputs at the soil surface. Both models used for the analysis presented in this paper take account of past inputs at the soil surface, but they use different functions to simulate the 'memory effect'. This explains why the two models estimate different indirect agricultural emissions (IE_{agr}) for the Elbe basin (1990-1995) (Table 3), after the sudden changes in 1989/1990.

The analysis described in this paper is based on an extensive large-scale database with, given the extent of our study area, a relatively detailed spatial resolution [9,15]. This suggests that despite the problems listed above the following generalisations can be made from the results:

i) In the early 1990s the average contribution of agriculture to the total nutrient load in the Rhine, Elbe, and Po rivers was approximately 45% for N and 20% for P, which is somewhat lower than the estimates reported by other workers.
ii) Between 1980-1985 and 1990-1995 there has been a slight (Rhine and Po basins) and drastic (Elbe basin) reduction of the agricultural surplus at the soil surface (manure production plus fertiliser consumption minus crop yields). However, this reduction has not (yet) resulted in a similar reduction of the agricultural inputs to the river network.

Simulation of the EU Nitrates Directive: 2015-2020
In the early 1990s only in a small part of the total area of the Rhine, Elbe, and Po basins the livestock manure application rate exceeded the maximum value given in the EU Nitrates Directive (Figure 5). Moreover, the intensive livestock breeding farms are in general located in flat areas with unconsolidated rocks, for which a relatively low nutrient transfer from soil to surface water was simulated (*b*, equation (2)) [11]. This explains why the requirement for the land application of livestock manure to be limited to 170 kg N hectare per year hardly results in a reduction of the predicted N loads in the Rhine, Elbe, and Po rivers. This measure might even lead to a relocation of livestock breeding farms towards areas that are more vulnerable for N pollution of surface waters (undulating areas with consolidated rocks) and that currently have livestock manure application rates of less than 170 kilogram per year per hectare. The above-mentioned requirement of the EU Nitrate Directive might locally result in improved environmental conditions, but this analysis suggests that it will not result in a substantial reduction of N and P loads in large European rivers.

The principal solution of agricultural nutrient pollution in Europe is to change towards agricultural systems where the input (manure and fertilisers) is balanced with the requirements of the crops (output). Only a complete balance between nutrient inputs and nutrient outputs at the farm level may on the long-term result in a hundred percent reduction of agricultural inputs to the river network. However, achieving a complete balance between nutrient inputs and nutrient outputs on agricultural soils may not be a realistic goal. Also the values used in scenario 'balance farming' (maximum agricultural surpluses of 25 (N) and 5 (P) kg hectare per year) are rather optimistic. This implies that the reductions simulated for scenario 'balance farming' illustrate more or less the maximum possible effect of the second requirement in the Nitrates Directive. Based on this analysis it is predicted that up to the year 2020, the maximum possible effect of the second requirement in the Nitrates Directive will be a 20 to 30% reduction of the total river N load (Table 4). Table 4 shows that for P the range of the estimates of *IE* is larger than the predicted change of *IE* between 1990-1995 and 2015-2020. So based on the analysis presented here one cannot say

much about the rate and extent of the reduction of indirect P emissions when the agricultural surpluses are drastically reduced.

Towards more precise estimates
Measures that aim at a reduction of nutrient levels in rivers require financial means and it is important to know which measures are most effective. Over the last two decades one could observe that the extension of wastewater treatment plants and the improvement of wastewater treatment technology resulted in reduced N and (especially) P levels in many European rivers [28]. This analysis shows that the effects of the measures that aim at a reduction of agricultural nutrient pollution (such as those defined in the EU Nitrates Directive) on the N and P loads in European rivers cannot yet be predicted precisely.

This stresses the need for a further improvement of models that can simulate large-scale nutrient fluxes from agricultural sources to river outlets. To do so one needs large-scale case studies where one can actually observe the effect of changes in agricultural land use on the agricultural inputs to river networks.

One interesting case study, which was only partly covered in this study, is the agricultural change in Central and Eastern Europe after 1989/1990. Many of the proposed measures in the EU Nitrate Directive have recently been realised by political change in Central and Eastern Europe (although not for environmental reasons). Fertiliser use dropped, livestock numbers decreased, and some agricultural land was taken out of production during the change of the political system. Several years have now passed and it will be interesting to analyse further how these changes in agricultural practices are reflected in nutrient levels in Central and Eastern European rivers. Although less striking, the recent changes in agricultural balances in Western Europe (a slight increase of crop yield and a slight reduction of fertiliser use), also allow for a further analysis of the effect of large-scale changes. Diffuse inputs are often estimated as the difference between river load at the outlet, losses in the river network, and inputs to the river network originating from point sources. This means that improved estimates of these fluxes will also improve the estimates of diffuse (agricultural) inputs.

Conclusions

In the early 1990s the estimated average contribution of agriculture to the total nutrient load was 43-49% (Rhine), 28-58% (Elbe), and 47-57 % (Po) for N and 13-21% (Rhine), 11-16% (Elbe), and 22-25 % (Po) for P. The reduction of the fertiliser consumption and the increase of crop yields resulted in a slight (Rhine and Po basins) and a drastic (Elbe basin) reduction of the agricultural surplus of N and (especially) P between 1985 and 1995. However, this reduction has not (yet) resulted in a similar reduction of the agricultural inputs to the river network. The results of this study suggest that the EU Nitrates Directive may not be stringent enough to substantially reduce the river N and P load in the nearby future (2015-2020). The principal solution of agricultural nutrient pollution in Europe is a large-scale change towards agricultural systems where the input (manure and fertilisers) is balanced with the requirements of the crops (output). There is a need to further improve models that can simulate large-scale nutrient fluxes from agricultural sources to river outlets. Such models need to be validated with large-scale evidence from the

field. The dramatic changes of agricultural practices after the political and economical changes in Central and Eastern Europe might provide more insight in the effects of large-scale changes in agricultural practices on nutrient loads in European rivers.

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