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Benefits of a Reallocation of Nitrate Emission Reductions in the Rhine River Basin[1]

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Abstract. In an attempt to improve ecological conditions of the Rhine, emission reduction targets have been set for different substances. For most substances targets have been met. However, nutrient emission reductions are behind schedule. It may be clear from intuition, and has also often been described in economic literature, that a flat reduction rate applied to all emitting sectors, though appealing because of equity reasons, may not be cost-effective. This paper explores the least cost allocation of nitrate emission reductions for the Rhine river basin, analysing different agricultural sectors and wastewater treatment plants. Results show that costs of meeting emission reduction targets can be brought down by almost 20% through a clever allocation of these targets.

Key words: cost-effectiveness, nutrients, Rhine, river basin policies, water quality

JEL classification: Q10, C6, Q1, Q2

1. Introduction

In the North Sea Action Plan, the national governments of the riparian countries have agreed to reduce emissions. For nutrients, the target is a 50% reduction in 1995, compared to the situation in 1985. Since the Rhine is discharging into the North Sea, the nutrient loads in this river also need to be reduced by this percentage. The phosphate reduction target will probably be achieved due to large-scale use of non-phosphate containing detergents, reduction of discharges by fertiliser industry, and phosphate treatment by waste water treatment plants. For nitrogen, the targets will probably not be met. The diffuse sources have a far larger impact on the total emissions for nitrogen than they have on the total emissions of phosphates, and they have shown little or no reduction in emissions (RIZA 1990).

Most of the reductions have been achieved by restricting point sources. Diffuse sources are much more difficult to regulate. Therefore, their relative importance to overall emissions is increasing. This is especially the case for nutrients and pesticides. At the moment, they are one of the main issues in Rhine river basin management (IRC 1994). The purpose of this paper is to analyse how the target set by the North Sea Conference (50% nutrient emission reduction by the year

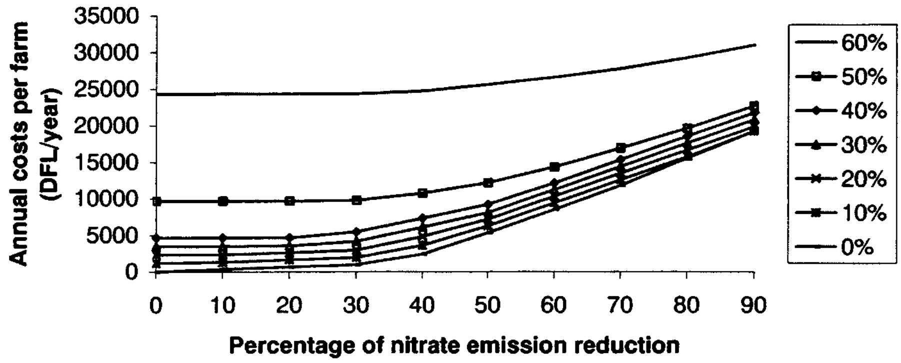

Figure 1. Costs of nitrate emission reduction strategies on an average Dutch dairy farm on clay and peat soils for different levels of ammonia emission reduction (van der Veeren and Tol 1997).

1995, compared to the situation in 1985) can be achieved at least costs. As many environmental economists have shown, a uniform emission reduction rate will most likely not be the cost effective solution to such a problem (Schleich et al. 1996; Ruff 1993; Tietenberg 1992). Nonetheless, a flat reduction rate of 50% is what has been advocated in the Rhine Action Plan. This was done also to enhance ecosystem functioning in this river. Until now, not all sources of nutrient emissions have shown the same rate of reduction. The question to be answered is: Should agriculture be pushed to reduce their share as well, or would it be preferable from a cost-effective point of view, to ask other sources of nutrient emissions to reduce a little more?

This paper presents the results of a cost-effectiveness analysis and a number of sensitivity analyses about its assumptions. Cost-effectiveness analysis is but one aspect. An integrated evaluation framework, which encompasses environmental quality, spatial equity and economic impacts of nutrient emission reduction policies in the Rhine river basin for sustainable development, can be found in Gilbert et al. (1999) and Van der Veeren et al. (1998). This paper concludes with a discussion on the model and the data used, resulting in recommendations for future research activities.

2. Assumptions

Before presenting the model used, the assumptions underlying the cost-effectiveness analysis will be elaborated upon first.

In the analysis, quadratic cost functions are used. They give a good representation of reality since costs of emission reductions, fitted to a linear programming model, will increase at an increasing rate (cf. Figure 1). This figure shows nitrate abatement costs for different ammonia emission reduction levels. The way these cost functions have been estimated, using data of Leneman et al. (1992) for Dutch agricultural sectors and Baan (1991) for Dutch sewage treatment plants has been described in Van der Veeren and Tol (1997). Their abatement functions depend on the resulting emissions. This is also the way the abatement functions are described throughout this paper, since this makes it easier to link emission(-reduction)s to nitrate loads to the Rhine and North Sea.

Their cost functions include nitrate, ammonia and phosphate emissions. However, this paper focuses on nitrate emissions only, assuming no restrictions on ammonia or phosphate emissions. Taking into account ammonia and phosphate emissions in the cost-functions would be interesting from a practical point of view, but makes the analysis more difficult. In future research, other emissions might be analysed as well.

Another adjustment in the cost functions presented in van der Veeren and Tol is related to the initial conditions. According to Leneman et al. (1989), some agricultural sectors can increase profits by reducing nutrient emissions (e.g. spring application of manure with ploughing under the manure directly; or changing animal diets so as to feed more according to the animals' needs). These sectors are not producing efficiently in the initial situation. This, for some sectors, inefficient initial situation has been the starting point for the cost functions presented in van der Veeren and Tol. The quadratic cost functions used in the analysis are based on the assumption that farms are producing efficiently in the initial situation, cf. they do not have any opportunities to reduce emissions profitably.[2]

Urban storm runoff and construction runoff have not been included as separate sources, but are included in the quantities sewage treated. Sewage treatment plants, as analysed in this paper, include wastewater treatment plants as well. Since the technologies used do not differ significantly between those two types of plants, the amounts treated, expressed in so-called Inhabitant Equivalents, have been added and treated the same.

In contrast to studies by Schleich et al. (1996) and Gren et al. (1997), a distinction has been made between different agricultural activities, e.g. arable farming on clay/peat soils, arable farming on sandy soils, dairy farming on clay/peat soils, dairy farming on sandy soils, pig breeding farms, and pig feeding farms. Poultry farms have not been included in the analysis, since manure produced in this sector is in high demand by and applied at arable farms. The nitrate emissions do not take place at the poultry farms. Reducing nitrate emissions at arable farms may cause problems with manure disposal from intensive livestock farming. But, since poultry manure is of a high quality (high nutrient content), compared to pig manure or cow manure, a decrease in demand for manure is supposed to affect the pig farms and dairy farms more significantly than the poultry sector. Other agricultural sectors have not been included since data, especially on costs, were not available (e.g. vineyards, horticulture), and/or the sectors have been assumed to be not significant in size and impact on total nutrient emissions.

In this paper, cost functions per source are assumed to be the same across regions. For the Rhine river basin, situated in the highly industrialised and

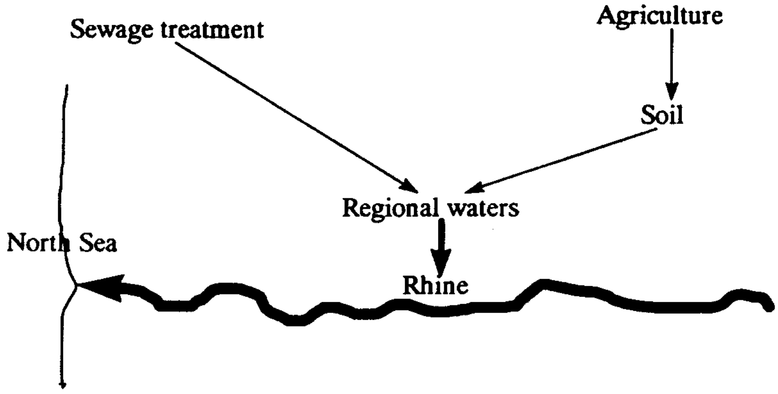

Figure 2. Linking the economic model with a nutrient transport model.

developed area of North-west Europe, this assumption seems to be realistic, since agricultural activities in Germany and other countries along the Rhine, can be supposed to be similar to their Dutch counterparts. However, Dutch agriculture is often thought to have (relatively) low nutrient emissions per kilogram agricultural production.[3] In a sensitivity analysis, the consequences of possible differences in nutrient emissions per kilogram agricultural production for various regions are studied in some detail by analysing the consequences of lower abatement costs outside the Netherlands.

Transport of nitrates is described using so-called transport coefficients. They describe the percentage of nitrates emitted by each sector reaching a particular effect region. These coefficients differ significantly between agricultural sources and sewage treatment plants. For agricultural activities, emissions to regional waters take place indirectly by transport through the soil, during which a part of the nitrates are retained in biochemical processes. Emissions from sewage treatment plants are most often direct emissions, and consequently, all of the nitrates emitted will eventually end up in the regional surface waters. Differences in length of regional surface waters before they reach the mainstream of the river Rhine result in differences in retention of nutrients in those waters. Therefore, transport coefficients are lower when regions are located further away. Finally, the percentage of nutrients in the various Rhine river sections that finally end up in the North Sea differ for the various regions. Figure 2 gives a graphical representation of nutrient transport.

Transport coefficients are very simple representations of transport mechanisms used in a water quality model. In cost-effectiveness analyses such as the one presented here, simple representations are preferred, since using more sophisticated water quality models may increase both model size and calculation time considerably.

The administrative level at which agricultural and environmental policies are imposed was taken to be the level of analysis. For the Rhine river basin, this resulted in ten different regions: Five Länder in Germany, one Agence de l'Eau in France, one Kanton in Switzerland, and the Netherlands, Belgium and Luxembourg.

In the data set by De Wit (1999), the amount of arable land (in hectares), the number of cows, sows, and feeding pigs, and the number of inhabitants (in inhabitant equivalents) are given for the entire Rhine river basin on a very detailed level (sometimes even on municipality level). This data set has been used to estimate the numbers of the so-called emission declaring variables in various regions.

The quadratic cost functions for agricultural activities were estimated at the farm level. The cost functions for sewage treatment plants were estimated for the Netherlands as a whole. Costs were initially expressed as costs per cow, per pig, per hectare arable land or per Inhabitant Equivalent. The number of hectares (et cetera) per regions was used to express the costs per region (cf. Equations 8–11 in the section on the data used). It is assumed that farmers will not be able to increase prices to shift the burden of the abatement costs to the consumers. The same applies to industries located in the Rhine river basin, where competition by other industries, located outside this river basin, prevents prices to be increased to compensate for higher costs. On the other hand, sewage treatment plants are mostly monopolistic in nature; citizens often do not have a choice to which sewage treatment plant they want to be connected. Abatement costs incurred by these plants are therefore completely paid for by local citizens, the consumers of the service provided. In either case, consumer demand is assumed to be independent of nitrate abatement strategies of the various sources.

Also, impacts on the suppliers of agricultural and other goods and services are assumed to be insignificant, thus allowing for a partial equilibrium analysis. However, in case of severe emission reductions, these impacts may become significant, especially on a local scale. For example, an increased supply of manure to the manure processing and distributing industry would result in increased activity in this part of agribusiness. At the same time, this would decrease possibilities to dispose all manure locally. Therefore, costs of processing and transport may increase, whereas application costs to arable farmers may decrease. However, one of the most important ways to decrease nitrate emissions by this agricultural activity is by shifting from manure to fertiliser application, thus decreasing the demand for manure. The increased demand for fertilisers may increase activities in this part of agribusiness. This paper does not include this type of secondary effects of nitrate abatement strategies.

3. The Model

The previous section described the assumptions underlying the analysis. In this section, the model will be presented as it has been used to estimate cost-effective allocations of nutrient abatement strategies in the Rhine river basin.

3.1. INDICES

h	=	Area
i	=	Source of nutrient emission

In this paper the following ten different regions (h) have been analysed and numbered as follows:

1	=	Switzerland
2	=	Baden-Wurthemberg
3	=	Bayern
4	=	Rheinland-Pfalz
5	=	Hessen
6	=	Belgium
7	=	Luxembourg
8	=	France
9	=	Nordrhein-Westfalen
10	=	The Netherlands

The indices for the respective sources (i) are numbered:

1	=	Arable farm on clay/peat soils
2	=	Arable farms on sandy soils
3	=	Dairy farms on clay/peat soils
4	=	Dairy farms on sandy soils
5	=	Pig breeding farms
6	=	Pig feeding farms
7	=	Sewage treatment plants

3.2. VARIABLES

In this model, the nitrate emissions by the various sources are described by:

$N_{h,i}$ = Nitrate emission by source i in region h in kg/year

Another variable that will be used in this model is λ, the shadow price. This variable describes what happens with the objective value when the constraint is made less strict.

3.3. DATA

$C_{h,i}(N_{h,i}) = \alpha_{0,h,i} + \alpha_{1,h,i} + \alpha_{2,h,i} N_{h,i}^2$ = Cost function describing for region *h* and source *i* the costs related to nitrate emissions.

Costs are presented in DFL/year, and nitrate emissions in kilograms/year.

T_{hi} = Transport coefficient, measuring the impact of source *i* in region *h* on the nitrate load in the North Sea.

G = Nitrate reduction target expressed as load to the North Sea in kilograms/year.

3.4. OBJECTIVE FUNCTION

The aim of this paper is to analyse cost-effective nutrient emission reduction strategies. Therefore, the model tries to reach the nutrient reduction targets, which are the restrictions of the model, at the lowest costs possible. Therefore, the objective function has been has been stated as follows:

Minimise costs:

$$\mathrm{Min}(C(N_{h,i})) = \mathrm{Min} \sum_h \sum_i C_{h,i}(N_{h,i}) \tag{1}$$

3.5. RESTRICTION

$$\sum_h \sum_i N_{h,i} T_{h,i} \leq G \tag{2}$$

This equation poses the restriction on the nitrate emissions, not to exceed 50% of the 1985 load to the North Sea.

Since the restriction can be assumed to be binding, this inequality constraint can be treated as an equality constraint.

This results in the following Lagrange function:

$$L = \sum_h \sum_i C_{h,i}(N_{h,i}) - \lambda(\sum_h \sum_i N_{h,i} T_{h,i} - G) \tag{3}$$

The first derivatives to N and λ have to be equal to zero for optimality:

$$\partial L/\partial N_{h,i} = \alpha_{1,h,i} + 2\alpha_{2,h,i} N_{h,i} - \lambda T_{h,i} = 0 \tag{4}$$

Table I. Initial nitrate emissions by the various sectors per hectare, cow, sow, feeding pig and IE

Source	Initial nitrate emission (kg nitrogen/year)
Arable farms on clay soils	20.34
Arable farms on sandy soils	55.66
Dairy farms on clay soils	7.94
Dairy farms on sandy soils	27.37
Pig breeding farms	7.00
Pig feeding farms	2.84
Sewage treatment plants	1.65

$$\partial L/\partial \lambda = -\sum_h \sum_i T_{h,i} + G = 0 \tag{5}$$

Which can be rewritten and solved by:

$$\begin{bmatrix} N_{h,i} \\ \lambda \end{bmatrix} = \begin{bmatrix} 2\alpha_{2,h,i} & -T_{h,i} \\ -T_{h,i} & 0 \end{bmatrix}^{-1} \begin{bmatrix} -\alpha_{1,h,i} \\ -G \end{bmatrix} \tag{6}$$

This gives the optimal (cost-effective) nitrate emissions by the various sources in the various regions, described by $N_{h,i}$.

4. The Data

Table I shows the initial nitrate emissions by the various sectors per hectare, cow, sow, feeding pig and Inhabitant Equivalent.

In Table II the numbers of hectares of arable farming, cows, sows, feeding pigs and Inhabitant Equivalents are shown for the various regions in the Rhine river basin, together with the transport coefficients. The transport coefficients for Switzerland are relatively high, especially for agriculture. This is caused by the high run-off from the Swiss mountains and the large amounts of water from these mountains that end up in the North Sea. The low transport coefficient for the Dutch agriculture is also remarkable. Often, especially in dry periods, water is taken in from the Rhine. Also, being located in a delta area, the Netherlands discharge into other rivers and lakes. This results in a water balance to the Rhine of almost zero, and therefore low transport coefficients for agriculture. The sewage treatment plants do not have this effect since they are discharging directly into the Rhine.

The cost functions have been assumed to be the same for the various regions. These functions can be written as:

$$C(N) = \alpha_0 + \alpha_1 N + \alpha_2 N^2 \tag{7}$$

Table II. Number of hectares of arable farming, cows, sows, feeding pigs and Inhabitant Equivalents for the various regions in the Rhine river basin

Region	Hectares of arable farming	Number of cows	Number of sows	Number of feeding pigs	Number of IE	NTP*	NTA**
Switzerland	279,506	696,286	171,035	513,104	7,514,268	0.54	0.36
Baden-Wuertemberg	635,297	354,247	225,014	437,686	12,808,174	0.56	0.14
Bayern	658,410	298,018	131,538	435,887	5,173,061	0.59	0.07
Rheinland-Pfalz	415,049	155,350	55,924	176,975	5,104,652	0.61	0.14
Hessen	274,928	95,967	45,632	158,950	6,640,249	0.61	0.11
Belgium	5,138	16,400	160	395	39,768	0.52	0.11
Luxembourg	54,312	69,350	9,262	19,631	519,867	0.52	0.11
France	654,187	329,860	2,624	15,266	3,939,970	0.53	0.12
Nordrhine-Westphalen	479,818	219,728	193,122	814,024	18,066,646	0.66	0.16
The Netherlands	475,729	76,336	5,058	38,665	15,328,979	0.70	0.21

*NTP = Nitrate transport coefficient for point sources. This is the fraction of emissions from point sources reaching the North Sea.
**NTA = Nitrate transport coefficient for agricultural sources. This is the fraction of emissions from agricultural sources reaching the North Sea.

Table III. The coefficients for the cost functions for the various activities*

Source	α_0	α_1	α_2
Arable farms on clay soils	11,112	–25.0489	0.0141
Arable farms on sandy soils	13,879	–7.6132	0.0010
Dairy farms on clay soils	23,651	–92.7476	0.0909
Dairy farms on sandy soils	26,860	–32.0718	0.0096
Pig breeding farms	25,930	–61.7391	0.0367
Pig feeding farms	12,975	–15.8237	0.0048
Sewage treatment plants	9.8168e+8	–56.4964	7.01824e-7

*The coefficients presented here have been adjusted to give minimum costs in the initial situation. This has been done by imposing the restriction that the first derivatives of the cost functions would be equal to zero in the initial situation. These adjusted coefficients are slightly different from the ones calculated based on the methodology used by van der Veeren and Tol (1997); maximum difference was less than 10%. For further reading on applying restrictions on coefficients, see Johnston (1984).

Table IV. The size of average Dutch farms in hectares, cows, sows, and feeding pigs (Leneman et al. 1992)*

Source		Unit
Arable farms on clay soils	43.6	hectares
Arable farms on sandy soils	65.5	hectares
Dairy farms on clay soils	64.2	dairy cows
Dairy farms on sandy soils	61.2	dairy cows
Pig breeding farms	120	sows
Pig feeding farms	576	feeding pigs

*For the recalculation of the cost function for sewage treatment plants, 21,328,801 Inhabitant Equivalents has been assumed. This equals 1.5 times the number of IE treated by municipal sewage treatment plants in the Netherlands in 1985 (CBS 1991). This number includes both municipal sewage and industrial waste water. The recalculation procedures used are similar to the ones for farms and are not repeated here.

Costs are presented in DFL/year, and nitrate emissions in kilograms N/year. The values of the coefficients for the various activities are represented in Table III.

The cost functions have been estimated for average farms, respectively all Dutch Inhabitant Equivalents. When rewriting the cost functions from farm level to hectares arable farming, cows, pigs, the size of the average farms has to be known, as well as the total number of inhabitant Equivalents. These numbers are shown in Table IV.

The recalculation of the cost functions has been done according to Equations (8–11). In Equations (8) and (9) the coefficients of the cost functions are rewritten

Table V. Percentages emission reduction as required for the various sectors in the different regions when emission reductions to the North Sea are allocated in a cost-effective way*

Source	1	2	3	4	5	6	7	8	9	10
Arable clay	60	24	12	23	18	18	18	20	26	35
Arable sand	80	76	38	76	60	60	60	65	80	80
Dairy clay	16	6	3	6	5	5	5	5	7	9
Dairy sand	46	18	9	18	14	14	14	16	20	27
Pig breeding	24	9	5	9	7	7	7	8	11	14
Pig feeding	80	37	18	37	29	29	29	31	42	55
Sewage treatment plants	43	44	47	48	48	41	41	42	52	55

*The numbers for the regions are described in the text.

to represent costs per hectare, cow, pig or Inhabitant Equivalent (α_{animal}). This has been done using the average size of emitting activities (ASA_i) as presented in Table IV. Note that α_1 does not change. In Equations (10) and (11) these costs per hectare, animal, or Inhabitant Equivalent are recalculated to represent costs per activity per region (α_{region}), using the size of the various activities in the various regions ($SAR_{h,i}$) as described in Table II. These α_{region} have been used in the analyses throughout this paper.

$$\alpha_{0,animal} = \alpha_0 / ASA_i \tag{8}$$

$$\alpha_{2,animal} = \alpha_2 * ASA_i \tag{9}$$

$$\alpha_{0,region} = \alpha_{0,animal} * SAR_{h,i} \tag{10}$$

$$\alpha_{2,region} = \alpha_{2,animal} * SAR_{h,i} \tag{11}$$

This results in different cost functions for the various activities in the different regions, depending on the size of various activities, but at source level, the cost functions remain the same in all regions.

5. Results

The cost-effective allocation of nitrate emissions among the various sectors in the respective regions as such, expressed in kilograms nitrogen per year, are not very informative. Therefore, the *percentage* nitrate emission reduction required to fulfil the objective of 50% reduction in load compared to the situation in 1985 have been calculated and are shown in Table V (the numbers of the regions are explained in the section on the indices used).

Switzerland has relatively high emission reduction percentages, for a region located the furthest away from the North Sea. This is due to the transport coefficient for this country. The large amounts of water coming from this region, running rapidly from the hills, and transporting lots of nutrients from agricultural sources, is the main cause for this phenomenon.[4]

Table V shows that arable farms on sandy soils should be the sector that reduces their nitrate emissions the most; for some regions they have to reduce by 80%. This does not mean that this sector should be shut down almost completely, since this sector has the opportunity to change from manure to fertiliser, which is assumed to result in less nitrate emissions. Pig feeding farms have to reduce their nitrate emissions significantly more than pig breeding farms. This is remarkable, since they have the same opportunities for nutrient abatement as pig breeding farms. The lower initial emissions and the significantly higher costs of applying the same nutrient emission reduction strategies in the latter type of pig farming cause this difference in nitrate emission reduction percentages.

Agricultural activities on sandy soils should reduce their emissions more than agricultural activities on clay/peat soils. This is mainly the result of the fact that nutrients get washed out more easily on sandy soils, and farmers have to apply more nutrients for the same crop yield. Farmers often anticipate on nutrients washing out by applying more nutrients than strictly necessary. This overapplication can be seen as a sort of a risk premium. It is this risk premium that will be reduced at low costs when nutrient emissions have to be reduced.

The total costs when a cost-effective allocation of nitrate emission reductions has been established are 1,089 million DFL/year. The total costs when a flat emission reduction rate of 50% would be applied on all sources are 1,337 million DFL/year. This result shows that a clever allocation of nitrate emission reductions can reduce costs by almost 20%.

Table VI shows the monetary benefits and costs for the various sectors and regions from such a reallocation of emission reductions. Since overall benefits exceed overall costs, opportunities exist for beneficiaries to compensate the sectors that will face costs, and still be better off. This not only applies to the Rhine basin as a whole, but also to the individual regions, except the Netherlands. Such side payments could be installed by regional governments by posing a levy on the beneficiaries, and subsidising those who have to increase their abatement efforts. Table VI shows that such a system can be beneficial, but would require interregional income transfers.

The optimal solution in cost-effectiveness analyses is characterised by the first order conditions (Equations 4 and 5). The marginal costs at source level vary inversely with the transport coefficients. For example, to reduce nitrate loads to the North Sea with one kilogram, Dutch sewage treatment plants would have to reduce their emissions with 1.43 kilogram (= 1 / 0.70), whereas agricultural sources in Bayern would have to reduce their emissions with 14.30 kilograms (= 1 / 0.07). The shadow price of a 1 kilogram nitrate load reduction is DFL 41.52 at the mouth of the river. In the optimal allocation of abatement strategies, the former sources will have to reduce their emissions until their marginal costs equal DFL 29.06,

Table VI. Benefits from changing from a flat rate emission reduction policy to a cost-effective allocation of nitrate abatement strategies (Million DFL/year)*

Source	1	2	3	4	5	6	7	8	9	10	Total
Arable clay	–3.76	15.94	19.91	10.41	7.62	0.14	1.51	17.61	11.03	7.85	88.26
Arable sand	–11.55	–22.40	7.27	–14.64	–3.20	–0.06	–0.63	–12.35	–19.83	–19.66	–97.05
Dairy clay	28.73	16.05	13.67	7.04	4.38	0.75	3.16	15.01	9.91	3.39	102.09
Dairy sand	5.02	16.88	15.81	7.40	4.84	0.83	3.50	16.34	9.99	2.95	83.56
Pig breeding	7.07	11.72	7.04	2.91	2.41	0.01	0.49	0.14	9.95	0.25	41.99
Pig feeding	–4.49	1.15	2.14	0.46	0.60	0.00	0.07	0.05	1.38	–0.05	1.31
Sewage treatment plants**	22.02	29.90	7.24	3.82	4.97	0.14	1.82	12.69	–17.56	–37.50	27.54
Total	43.04	69.24	73.08	17.4	21.62	1.81	9.92	49.49	4.87	–42.77	247.7

*The numbers for the regions are described in the text.
**This includes wastewater treatment plants. The applied technologies are similar.

Table VII. Marginal costs at source level for point sources and agricultural sources (DFL/kilogram nitrate ∗ year)

Region	Marginal costs point sources	Marginal costs agricultural sources
Switzerland	22.42	14.95
Baden-Wuertemberg	23.25	5.81
Bayern	24.49	2.91
Rheinland-Pfalz	25.32	5.81
Hessen	25.32	4.57
Belgium	21.59	4.57
Luxembourg	21.59	4.57
France	22.00	4.98
Nordrhine-Westphalen	27.40	6.64
The Netherlands	29.06	8.72

whereas the latter will continue until their marginal costs equal DFL 2.91. Table VII shows the marginal abatement costs for point sources and agricultural sources in the various regions in the optimal solution for a 50% reduction of nitrate loads to the North Sea.

Whereas, at source level, different cost functions have been used for alternative nitrate emitting activities, they have been supposed to be similar for the various regions. Therefore, differences in required emission reduction percentages between regions are due to the transmission coefficients used. If different cost functions would have been used for the various regions, this might also have been the case. However, as Gren et al. (1997) showed, this is no longer a triviality. If upstream activities have low abatement costs compared to the downstream ones, the differences in costs may overcome the differences in impacts on the receiving water body. This would result in more emission reduction in the upstream areas than in case costs related to emission reduction strategies are (supposed to be) similar.

Dutch agriculture is very intensive. High nutrient emissions per hectare are caused by high production rates. This has lead to increased policies aimed at reducing environmental pressure from this sector. As a result of this, in 1990 Dutch agriculture had reduced their nitrogen surpluses by about 17% compared to 1986 (Olsthoorn 1992), and in 1993 this reduction was about 23% (Fong 1995). Since the manure problem has often been regarded as a typical Dutch problem, other regions in the Rhine river basin can be assumed to have less strict nutrient policies. This could mean that farmers and sewage treatment plants outside the Netherlands still have some cheap options for nutrient emission reductions; options already used in the Netherlands. On the other hand, the Rhine river basin is located in the western, industrialised part of Europe. Therefore, differences in production patterns can be

Table VIII. Percentages emission reduction as required for the various sectors in the Netherlands, Nordrhine-Westfalen, and Switzerland when emissions are allocated in a cost-effective way, when abatement costs outside the Netherlands are assumed to be 10% lower than in the Netherlands

	Switzerland	Nordrhine Westphalen	The Netherlands	Switzerland	Nordrhine Westphalen	The Netherlands
Source	0%	0%	0%	10%	10%	10%
Arable clay	60	26	35	61	27	32
Arable sand	80	80	80	80	80	80
Dairy clay	16	7	9	16	7	9
Dairy sandy	46	20	27	48	21	25
Pig breeding	24	11	14	25	11	13
Pig feeding	80	42	55	80	43	51
Sewage treatment	43	52	55	44	53	51

assumed to be relatively limited, as opposed to situations in which western and eastern European regions are compared (e.g. Gren et al. 1997). Therefore, the cost effectiveness analysis has also been performed assuming 10% lower costs for the regions outside the Netherlands. Table VIII shows some shifts in the percentages emission reduction for the Netherlands, Nordrhein-Westfalen, and Switzerland. The benefits of changing from a flat rate emission reduction policy to a cost effective allocation of nutrient abatement is lower in absolute terms, but the total cost reduction is still about 20%.

The North Sea Commission wanted to reduce loads of all kinds of substances to the North Sea by 50% as a first step to enhance the rehabilitation of the North Sea ecosystems. In this paper, the 50% emission reduction policy of the International Rhine Committee is seen as a reaction to reduce the loads of the Rhine to the North Sea by 50%. The cost-difference between the two cost-effective emission reduction schemes – that is, the one that reduces loads to the North Sea by 50%, and the one that does that and in addition achieves the same water quality as the flat rate emission reduction – can be interpreted as the willingness to pay to improve the water quality of the Rhine. The International Rhine Committee has made the return of the salmon the prime visible objective, which should be possible when water quality in the Rhine has been improved. Costs related to water quality improvement could thus be stated as the costs for the rehabilitation of the salmon. Table IX shows the three types of costs for the ten regions in the Rhine river basin.

The price for the easy implementation of a flat rate policy is apparently very high, since the same water quality of the Rhine can be achieved at almost 20% lower costs.

Another feature from Table IX is that most regions will face higher costs when policy is shifted from a reduction of nitrate loads to the North Sea to

Table IX. Costs per region, for three nitrate emission reduction strategies; All sources reducing 50%, all regions reducing 50%, or the loads to the North Sea reduced by 50% (million DFL/year)*

Region	All sources 50%	All regions 50%	North Sea 50%
Switzerland	180.15	135.09	137.10
Baden-Wurthemberg	226.27	186.50	157.02
Bayern	134.17	93.96	61.08
Rheinland-Pfalz	99.26	81.24	81.83
Hessen	101.05	89.35	79.43
Belgium	2.40	1.43	0.59
Luxembourg	16.42	11.22	6.50
France	114.43	84.63	64.93
Nordrhein-Westfalen	260.98	233.10	256.11
The Netherlands	202.06	185.40	244.82
Total	1,337	1,101	1,089

*the costs for all German Lander together are 822, 684, and 635 million DFL/year respectively.

a reduction in nitrate loads within each region. The Netherlands, Nordrhein-Westfalen, Rheinland-Pfalz, and Switzerland will have to reduce their abatement efforts. These are the regions with high nitrate transport coefficients for either the agricultural sector (e.g. Switzerland) or for both sectors (e.g. the Netherlands, Nordrhein-Westfalen, and Rheinland-Pfalz). These regions reduce their emissions by more than 50% when the reduction objective is aimed at the North Sea. The benefits these four regions get from this shift in policy is almost equal to the increase in costs incurred by the other regions. It is remarkable that the total costs of a 50% reduction in nitrate loads to the North Sea is almost similar to those of a 50% reduction per region. This is caused by the limited retention in the mainstream of the Rhine. Differences in transport coefficients to the North Sea are therefore largely determined by differences at a regional scale, and to a lesser extent by the place where the nutrients enter the mainstream of the Rhine.

The International Rhine Committee wants to improve the water quality in the Rhine river basin. This can be done by imposing a 50% flat rate emission reduction on all sources. Such a policy is easy to implement, and sounds fair since everybody has to reduce by the same amount. However, the same objective can apparently be achieved at lower costs, when emission reductions are allocated in a cost-effective way. Such a cost-effective allocation implies a more tailor made emission reduction policy and is therefore more difficult to implement. The difference in costs between a flat rate emission reduction and a cost-effective allocation with the same water quality is an estimation of the costs for easy implementation of an emission reduction policy. The extra costs compared to a cost-effective allocation put the

Table X. Transport coefficients used in the sensitivity analysis on nitrate transport (ranges as described in Schuttelaar, 1998)

Region	NTP low	NTP average	NTP high	NTA low	NTA average	NTA high
Switzerland	0.38	0.54	0.62	0.16	0.36	0.59
Baden-Wuertemberg	0.40	0.56	0.63	0.03	0.14	0.29
Bayern	0.45	0.59	0.66	0.03	0.07	0.21
Rheinland-Pfalz	0.47	0.61	0.67	0.03	0.14	0.28
Hessen	0.47	0.61	0.67	0.03	0.11	0.23
Belgium	0.33	0.52	0.60	0.03	0.11	0.19
Luxembourg	0.33	0.52	0.60	0.03	0.11	0.19
France	0.34	0.53	0.61	0.03	0.12	0.21
Nordrhine-Westphalen	0.56	0.66	0.71	0.06	0.16	0.32
The Netherlands	0.64	0.70	0.73	0.08	0.21	0.34

argument of fairness in another dimension. Not many people will agree that asking more money from society than necessary to achieve the same objective(s) is fair.

For such a shift in policy, interregional side-payments become necessary, since some regions will benefit and others will face higher costs. However, a shift from a flat rate emission reduction to a cost-effective allocation is still beneficiary even when the same water quality levels in the Rhine need to be achieved.

Table X shows the ranges of the nitrate transport coefficients as described by Schuttelaar (1998). The coefficients used to arrive at the results presented in the previous tables are the average values as presented in this table. Hydrological variations have important consequences on run-off, discharge, and retention. In general, nitrate emission reductions by agricultural sources become relatively more important when hydrological circumstances relate more to a year with high rainfall. This is according to intuition; in humid years, more nutrients run off, therefore, agricultural nutrient management becomes more important. This result could already be predicted by comparing the upper and lower bounds of the transport coefficients (see Table X). For sewage treatment plants they differ less than 100%, whereas for agricultural sources, the coefficients can be almost ten times as high.

The decrease in costs, compared to flat rate emission reduction policies (costs flat rate: 1,337 million DFL/year), are for a dry year 356 mln DFL/year, which is a reduction of 27% compared to a flat rate emission reduction policy. For an average year this is 248 mln DFL/year (see also Table VI), which is almost 20%, and for a humid year this is 295 mln DFL/year, or 22%. From this, it appears that the hydrological circumstances have a significant impact on the results.

If biochemical processes in the mainstream of the Rhine take place to a higher degree than assumed by Schuttelaar (1998), retention will be higher, and therefore, transport coefficients would be lower. For this purpose, analyses have been performed, assuming a low (0%) retention in the mainstream of the Rhine, a high retention rate (50%), and the 23% as used by Schuttelaar (1998), and which has been used throughout this paper as well. Table A1 (Appendix) shows the transport coefficients used to analyse the importance of this effect.

Tables A2 and A3 (Appendix) describe results for this part of the sensitivity analysis. For some regions, increased uptake of nitrates by biochemical processes (= retention) in the mainstream of the Rhine reduces the relative importance of nutrient abatement by point sources, since their impact on total loads to the North Sea is reduced, compared to the impact of agricultural sources (e.g. France, The Netherlands). Therefore, in these regions, the latter sources can be expected to increase their abatement activities, whereas emission reductions for sewage treatment plants can be expected to stay the same. In contrast, for some other regions, point sources become relatively more important (e.g. Switzerland, Nordrhine-Westphalen).

The changes in emission reduction percentages shown may look rather insignificant; however this is not true for the impact on the benefits of a reallocation of nitrate abatement strategies, as can be seen from Table A3. It appears that Rheinland-Pfalz and Hessen would significantly benefit from reduced nitrate uptake in the mainstream of the Rhine, allowing these regions to reduce their abatement activities. Since the objective of a 50% reduction of nitrate loads to the North Sea has to be met, reduced abatement efforts by one region should result in increased efforts by other regions and/or sectors.

6. Conclusions

This paper has shown a cost-effectiveness analysis for nitrate emission reduction to the North Sea. It includes regional aspects by means of the transport coefficients, and offers the possibility to implement different cost-functions for different regions, when available.

The 50% emission reduction target as set by the North Sea Conference has been the starting point of this analysis. A flat emission reduction rate, as proposed by the International Rhine Committee, is likely not to be the most cost-effective way. This paper shows that various sectors would have to reduce their emissions to different degrees if cost-effective nitrate abatement is to be achieved. As is also found by Gren et al. (1997) and by Schleich et al. (1996), agriculture will have to reduce their emissions quite substantially, but, in contrast to these two studies, this paper shows that agriculture should not be treated as one homogeneous sector. Some agricultural activities will have to reduce their emissions by only a few percent, whereas other sectors, with relatively low abatement costs, will have to impose drastic measures. With this model it is possible to estimate the cost-effective allocation of nutrient abatement strategies also with respect to other receiving waters than the North Sea, e.g. recreation areas along the Rhine, thus enabling a cost-benefit analysis of improvements in water quality. This has not been done in this paper, but this option will be explored in future papers.

Another useful and interesting extension of this model would be the implication of phosphate emissions as well.

Nutrient abatement in the Rhine river basin has serious implications with respect to environmental quality, spatial equity and economic impacts. The analysis in this paper is but one of the elements in an integrated framework to evaluate nutrient abatement policies in the Rhine river basin for sustainable development (cf. Van der Veeren et al. 1998; Gilbert et al. 1999).

The distribution of the benefits of a reallocation of emission reduction targets is far from uniform. The least benefiting regions may not be in favour of the implementation of a more cost-effective allocation. Furthermore, if regions are compensated to apply more than 50% emission reduction, they may have an impetus to cheat. This type of issue can be analysed using game theoretical concepts. Van der Veeren and Tol (1999) present a game theoretical analysis on the impacts of cooperative and non-cooperative behaviour with respect to nitrate abatement policies in the Rhine river basin, based on the analyses presented in this paper.

The implementation of regional/sectoral emission reduction targets can take place by imposing standards or economic instruments (e.g. taxes, transferable discharge permits; cf. Pearce and Turner 1990; Pearce and Markandya 1989). Presently, the application of standards is common practice. However, this can be challenged in the same way as this paper has challenged the application of a uniform emission reduction target. A thorough analysis of the consequences of different ways to implement emission reduction targets falls outside the scope of this paper and is left for future research.

There are a number of assumptions that have been made in the analysis presented in this paper that can be questioned. Most of them are caused by a lack of useful data. Probably the most important one is that cost-functions have been assumed to be the same in the various regions. Dutch agriculture may well be more nutrient efficient than average. Different cost-functions then need to be estimated for the various activities and regions. A first attempt was made to analyse the effect of 10% lower costs for nutrient emission reductions outside the Netherlands. Results showed that this has only limited consequences on the cost-effective allocation of nitrate abatement strategies.

Also the values used for the transport coefficients may be less accurate. The sensitivity analysis showed that changes in these coefficients, depending on the hydrological circumstances, have significant impacts on both the allocation of abatement strategies and on the costs involved.

Another important aspect is the relatively low percentage of the French and Belgian population connected to sewage treatment plants. In the calculations presented above, the contribution of people not connected to sewage treatment plants has not been accounted for.

Appendix

Table A1. Transport coefficients used in the sensitivity analysis on retention in the mainstream of the Rhine

Region	NTP* Low retention	NTP Mid retention	NTP High retention	NTA** Low retention	NTA Mid retention	NTA High retention
Switzerland	0.70	0.54	0.35	0.47	0.36	0.23
Baden-Wuertemberg	0.73	0.56	0.36	0.18	0.14	0.09
Bayern	0.77	0.59	0.38	0.09	0.07	0.05
Rheinland-Pfalz	0.79	0.61	0.40	0.18	0.14	0.09
Hessen	0.79	0.61	0.40	0.14	0.11	0.07
Belgium	0.68	0.52	0.34	0.14	0.11	0.07
Luxembourg	0.68	0.52	0.34	0.14	0.11	0.07
France	0.69	0.53	0.34	0.16	0.12	0.08
Nordrhine-Westphalen	0.86	0.66	0.43	0.21	0.16	0.10
The Netherlands	0.91	0.70	0.45	0.27	0.21	0.14

*Nitrate Transport coefficient for Point sources (sewage and waste water treatment plants).
**Nitrate Transport coefficient for Agricultural sources.

Table A2. Percentages emission reduction by various sources in the Rhine river basin for various degrees of retention in the mainstream of the Rhine*

	Switzerland			France			Nordrhine Westphalen			The Netherlands		
Source	Low	Mid	High	Low	Mid	High	Low	Mid	High	Low	Mid	High
Arable clay	60	60	59	20	20	20	27	26	26	34	35	36
Arable sand	80	80	80	67	65	67	80	80	80	80	80	80
Dairy clay	16	16	16	6	5	6	7	7	7	9	9	10
Dairy sandy	47	46	46	16	16	16	21	20	20	27	27	28
Pig breeding	24	24	24	8	8	8	11	11	10	14	14	15
Pig feeding	80	80	80	32	31	32	42	42	40	54	55	57
Sewage treatment	43	43	43	42	42	42	52	52	53	55	55	55

*Low, Mid and High apply to NTP and NTA simultaneously.

Table A3. Benefits from changing from a flat rate emission reduction policy to a cost-effective allocation of nitrate abatement strategies per region for different levels of retention in the mainstream of the Rhine (Million DFL/year)*

Region	Low	Mid	High
Switzerland	43.01	43.04	44.29
Baden-Wuertemberg	69.80	69.24	71.49
Bayern	73.01	73.08	70.94
Rheinland-Pfalz	18.53	17.4	16.54
Hessen	22.69	21.62	20.33
Belgium	1.82	1.81	1.81
Luxembourg	9.99	9.92	9.93
France	47.64	49.49	47.95
Nordrhine-Westphalen	3.81	4.87	2.92
The Netherlands	–42.43	–42.77	–40.38
Total	247.9	247.7	245.8

*Low, Mid and High apply to NTP and NTA simultaneously.

Notes

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[2] These "free" or sometimes even "profitable" emission reductions have been integrated in the numbers shown in the various tables in the results section.

[3] The high production levels per hectare in the Netherlands offset the relatively low emissions per kilogram production, resulting in high emissions per hectare. Nitrogen surpluses in this country are the largest in Europe (Brouwer et al. 1995).

[4] Schuttelaar (1998) indicates that the transport coefficients for the Netherlands and Switzerland, as described in her paper, are possibly less reliable than those presented for the other regions. Therefore, transport coefficients for the Netherlands have not been taken from this study, but recalculated figures (Schuttelaar, pers. com.) have been used. However, since these numbers are the best presently available, they are used throughout this paper.

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