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Plausibility of an integrated national model for the evaluation of mitigation options on agricultural nitrogen losses

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Plausibility of an integrated national model for the evaluation of mitigation options on agricultural nitrogen losses

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Abstract
In the Netherlands nitrogen control policy is focusing on reducing (i) leaching and runoff of N to ground water and surface water and (ii) emissions of ammonia (NH_3) and nitrous oxide (N_2O) to the atmosphere. An integrated N model is thus crucial to determine the (cost) effectiveness of measures at regional and national scale. Because existing models do not focus on the fluxes of all relevant reactive N compounds to different compartments and are mostly complex, parameter rich and data hungry, they are less suitable for an integrated evaluation of mitigation measures on a regional or national scale. Therefore, the integrated nitrogen model Initiator was developed, representing all crucial processes in the N chain by simple process descriptions. Here we address the plausibility of Initiator and demonstrate how the model can be used for the evaluation of mitigation strategies. Plausibility was judged on a comparison of the crucial model outputs (leaching and runoff of N to ground water and surface water and emissions of NH_3 and N_2O to the atmosphere) with those of the complex national reference models and a comparison between modelled nitrate concentrations in groundwater with available measurements. Results show that the model performance of Initiator is comparable to that of complex references models,, especially when focussing at a national scale. Mitigation measures where evaluated on their environmental benefit and their cost effectiveness as well. Management measures appear to be rather cost effective in order to reduce the N losses to atmosphere and water compartments, but additional expensive technical measures are needed to comply with national and European directives on ammonia emissions to the

atmosphere.

Keywords: ammonia emission, modelling, nitrate leaching, validation

Introduction

In the Netherlands, nitrogen (N) control policy is focusing on reducing (i) leaching and runoff of N to ground water and surface water and (ii) emissions of ammonia (NH_3) and nitrous oxide (N_2O) to the atmosphere. Although several models on N emission do exist for the Netherlands, these are generally focusing on specific N fluxes and not on the fluxes of all relevant reactive N compounds to different compartments. Furthermore, these models are mostly complex, parameter rich and data hungry, thus being less suitable for an integrated evaluation of mitigation measures on a regional or national scale. One possibility is to simplify the model description in such a way that the temporal and spatial resolution is comparable to the resolution of the data. Based on this philosophy, the integrated N model Initiator (De Vries et al., 2003a) was developed, representing all crucial processes in the N chain by simple process descriptions. The model initiator has been used as stand-alone model for the evaluation of the effectiveness of measures (De Vries et al., 2001; Kros, 2003) as well as part of the N decision support system NitroGenius (Erisman et al., 2002). Until now, only little attention has been paid to the plausibility and reliability of the model results.

Here we address the plausibility of the Initiator model and demonstrate how the model can be used for the evaluation of mitigation strategies. Because extensive monitoring data on N emission data at a national scale were generally lacking, a real validation was hardly possible. Therefore, we assessed the plausibility by comparison of the crucial model outputs (leaching and runoff of N to ground water and surface water and emissions of NH_3 and N_2O to the atmosphere) with those of the complex national reference models. In addition, a comparison was made between modelled N concentrations in groundwater with available measurements.

To evaluate the (cost) effectiveness of measures on N fluxes in the Netherlands, 14 measures focussing on improved farming practices (management measures) and structural changes in agriculture (technical measures) were implemented in Initiator. The benefits of these measures were evaluated in view of the current policy aims: (i) a national ammonia ceiling of 100 Gg NH_3 in the year 2010 (i.e. 30% reduction compared to actual (the year 2000) situation); (ii) N in surface waters in view of the: EU standard (< 2.2 mg N L^{-1}) (iii) NO_3 in groundwater in view of the EU standard (< 50 mg NO_3 L^{-1}) and (iii) N_2O emission in view of the Kyoto protocol (a 6% decrease compared to 1990).

The integrated N model Initiator

To gain insight in the magnitude of leaching and runoff of N to ground water and surface water and of emissions of ammonia and nitrous oxides to the air in response to N inputs, it is needed to estimate all N inputs and outputs, including gains and losses within and from the soil (the so-called soil system budget, Oenema and Heinen, 1999). All these fluxes are included in the model INITIATOR (Integrated NITtrogen Impact Assessment Tool On a Regional scale) (De Vries et al., 2003a). A flow chart of the considered N inputs and N transformation processes in the model for both terrestrial and aquatic ecosystems is given in Figure I. This flow chart is a simplified picture of the fate of N that enters the terrestrial system. We have chosen a simple approach to maintain transparency and to be able to apply the model with available data.

The total input at the soil surface is calculated by adding available data on the input by animal manure, fertiliser, atmospheric deposition and biological N fixation. N losses from the terrestrial system are calculated as a function of type of manure, soil type, land use and/or hydrology. Ammonia emissions are calculated by relating them linearly to the amount of animal manure and fertilisers applied to the soil. The parameterisation of the equations for estimating the NH_3 loss from animal manure was done in such a way that it included all NH_3 losses, including those from animal housing systems, manure storage systems, grazing animals and the application of manure to soil. Next, N uptake, N immobilisation/ mobilisation in the soil and N losses via denitrification are calculated.

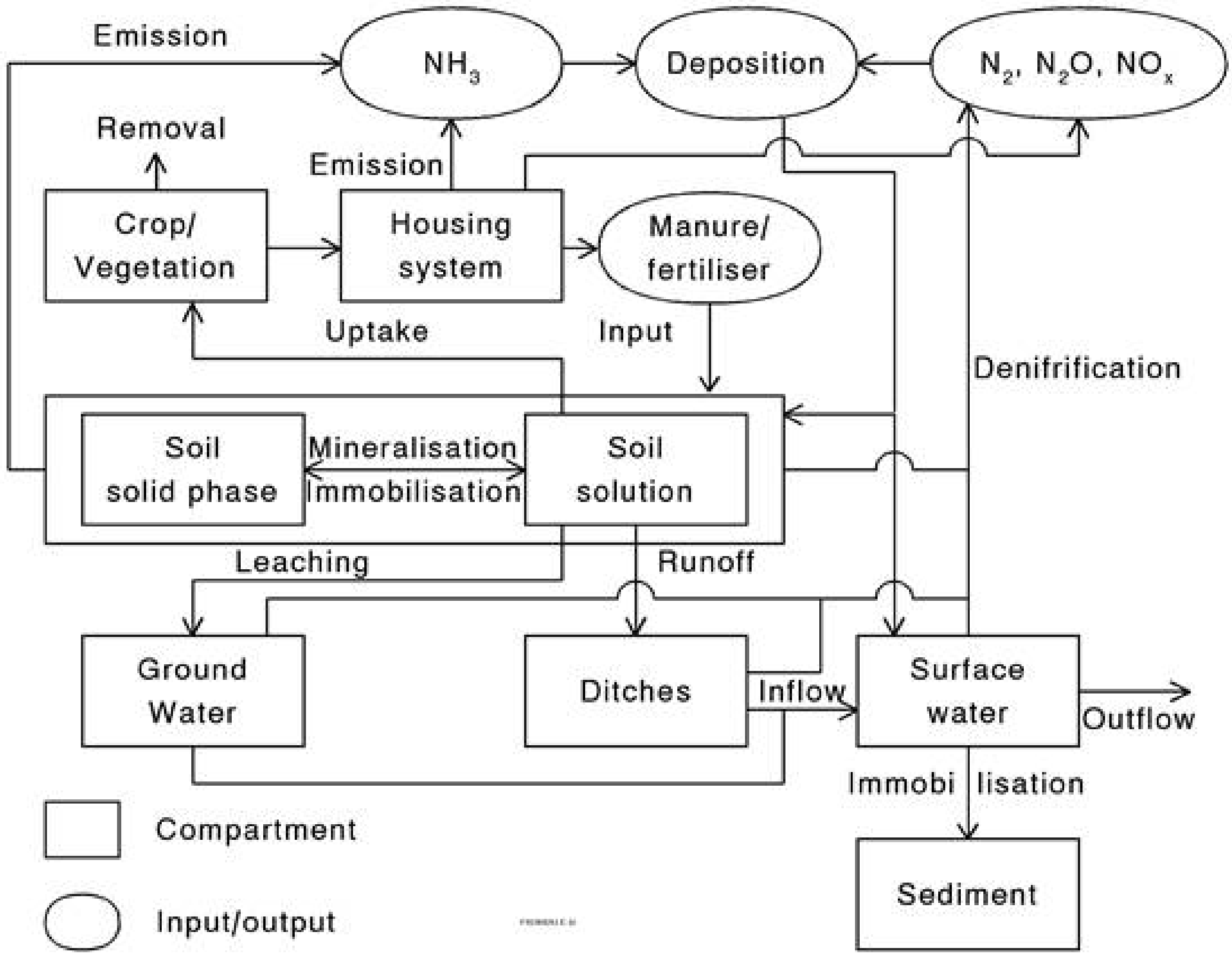


Figure I Schematic representation of the Initiator model

The flux of N leaving the terrestrial system is calculated by subtracting all N losses from the system (emission, uptake, net accumulation and denitrification) from the input to the soil. The leaching loss from the terrestrial system is partitioned to surface water and to groundwater. Since we were interested in the leaching of N to the groundwater at 1-meter depth below the phreatic level (the depth where nitrate concentrations are measured in the Netherlands), denitrification of N in upper ground water was also considered. The processes considered relevant in aquatic systems are N retention in ditches and larger surface waters, retention being distinguished in denitrification and accumulation in the sediment. Denitrification is thus calculated in the soil, upper ground water, ditches and surface water (compare Figure I). We considered runoff from terrestrial systems and direct atmospheric deposition of N to surface waters as an input of N in aquatic systems. The various N outputs from the soil and N immobilisation in the soil, ground water and surface water, are calculated with a consistent set of simple equations while assuming steady state. Mitigation measures are implemented through affecting model inputs and parameters such as, animal manure, fertiliser use, emission fractions, N efficiency and denitrification rate.

The INITIATOR model was applied to the whole of the Netherlands, distinguishing grid cells with unique combinations of soil type, land use, N inputs and hydrology, which determine the parameterisation of N transformation processes. A total number of 4647 so-called STONE plots were distinguished for which N input data by manure, fertiliser and deposition were available, consisting of a multiple of 250m x 250m grid cells with unique combinations of soil use, soil type and ground water table class. Geo-referenced data for the N input via animal manure and fertilisers were based on data statistics at farm and municipal level for the year 2000. Animal manure was divided in cattle, pig and poultry manure and in dung and urine deposited on grassland by grazing animals, because of differences in the ammonia emissions from either housing and manure storage systems or the soil. N deposition data were based on modelled N deposition data at a 1×1 km^2 grid scale for the year 2000. N fixation was estimated as a function of land use. A distinction was made between grassland, maize and arable land. Soils were divided in sand, loess, clay and peat. Furthermore, a distinction was made in different hydrological regimes (wetness classes) based on ground water table classes from the 1: 50 000 soil map. Ranges in model parameters describing the various N transformations were based on literature data, field observations, results from more detailed model calculations and in some cases from expert judgement, as described below. For a detailed description of the model and data used, we refer to De Vries et al. (2003a).

Reference models and observations

An overview of the used reference models and observations is given in Table I. A reference model is defined as a model whose results are used by the government for a particular policy aspect within the N cycle such as ammonia emission, nitrous oxide emission and N leaching to ground and surface waters.

Table I Used models and data for plausibility analysis

Output	Model	Observations
NH_3 emission	MAM; (mam: Groenwold et al., 2003)	-
N_2O emission	IPCC-NL; (Spakman et al., 1996)	-
NO_3 leaching	ANIMO; (Rijtema and Kroes, 1991)	LMM; (Fraters et al., 1998)

For the evaluation of the NH_3 emission policy in the Netherlands, the manure and ammonia model (MAM) has been developed (mam: Groenwold et al., 2003). This model is used to quantify the official yearly national ammonia emission figures. mam calculates NH_3 emission due to animal manure production, distribution, grazing, application, export and processing of animal manure and fertilizer use. It uses data from individual farms throughout the country to calculate ammonia emissions for 31 manure regions.

For the calculation of national N_2O emissions the Netherlands uses a country specific version of the IPCC-method (Spakman et al., 1996). Contrary to the NH_3 emission and NO_3 leaching, for N_2O emission no detailed model approach is used in the Netherlands, which hampers the plausibility test. Ideally, a comparison should be made with a more detailed process-oriented N_2O model, such as the DNDC model that has been applied successfully at a national scale in various countries, e.g. for China (Li et al., 2001).

For the quantification of the official N and P leaching towards surface waters and ground water the integrated nutrient emission model stone (Wolf et al., 2003). is used in the Netherlands. and to evaluate the impacts of the ‘Animal manure law’ (RIVM, 2004). This model is used to evaluate the impacts of the Dutch ‘Animal Manure Law’. Within the stone framework the soil model animo (Rijtema and Kroes, 1991) calculates the transport of water and nutrients (N and P) in the soil-water system on a daily time scale for all stone plots (the same as used for initiator). The inputs of those nutrients as fertilisers and animal manure are derived with the model clean (Van Tol et al., 2002).

For the validation of modelled concentrations in ground water, observations from the national manure monitoring network were used (Fraters et al., 1998). This monitoring network consists of 250 farms distributed over the whole of the Netherlands. For the validation farm averaged observations, sampled in the period April 1998 to February 2001, where used. At each monitoring point the best available farm specific input data, such as soil type, crop type, manure and fertilizer use, were derived. Initiator was run for each monitoring farm, while using these farm specific input data.

Mitigation measures and Cost-Benefit analysis

In order to investigate the benefit of N mitigation possibilities, 14 measures were evaluated (see Table II), divided in management measures (mostly related to farming practise) and technical measures (mostly related to structural changes in the farming system). Management measures were performed first because they do not need expensive technical adaptation of the farming system. Most management measures were focussing on either reducing the N inputs or an increase in N efficiency. The technical measures focussed mainly on mitigating the emission of ammonia to the atmosphere through adapting the farming system and manure processing. The parameterisation of the mitigation measures was mainly based on expert judgement (see Kros et al., 2003 for more details). The distinction between technical (T) and management (M) measures was made in order to get insight into what can be achieved relatively simple by the farmers themselves without large investment and what can be achieved with more costly structural measures.

In order to gain more insight into the relevance and practise of these measures, a rather simple cost-benefit analyses was performed. Here we report, the costs of applying each measure to the Netherlands as whole. As with the parameterisation of the mitigation, the costs of the measures were based on expert judgement (see Kros et al., 2003). The cost-effectiveness was expressed by dividing the achieved reduction percentage of a particular N emission by the total cost of the concerned measure.

Table II Evaluated measures

Nr	Measure	Type[1)]	Explanation[2)]
1	Decrease livestock intensity	M/T	Ongoing process due to actual policy
2	Improving animal feeding	M	Enhancing the N efficiency in all animal categories
3	Reducing fertiliser use	M	Due to a better use of fertilisers and animal manure through precision agriculture
4	Apply cover crops	M	On arable land more N will be taken up.
5	Optimal drainage	M	Irrigation of dry soils and draining very wet soils will result in higher uptake of N.
6	Low emission application of animal manure and cover manure reservoirs	M	Results in a lower NH_3 emission fraction for application and storage

7	Reduce grazing time	M	Leads to moving of manure from stable to pasture. Reduces NH_3 emission (if low emission housing is applied) and N leaching
8	Low emission housing	T/M	Lower NH_3 emission fractions from stables and manure storage systems within pig and poultry husbandry, according to Dutch policy rules
9	Extremely low emission pig and poultry husbandry	T	Apply lowest NH_3 emission fractions possible for pig and poultry farms.
10	Extremely low emission housing for dairy farms	T	Apply lowest NH_3 emission fractions possible for dairy farms
11	Manure processing	T	Processing the manure surplus without any emission losses; the nutrients and metals are extracted from the Dutch agriculture
12	Improving workability factor of animal manure	T	Increase N efficiency and causes a lower N and P input by fertiliser
13	Buffer strip	T	Manure and fertiliser free zones along drainage canals. Reduces runoff of N but increases leaching.
14	Emission free pig and poultry husbandry	T	Remaining pig and poultry are staying in NH_3 emission free stables and all manure is processed and transported (national target for 2030)

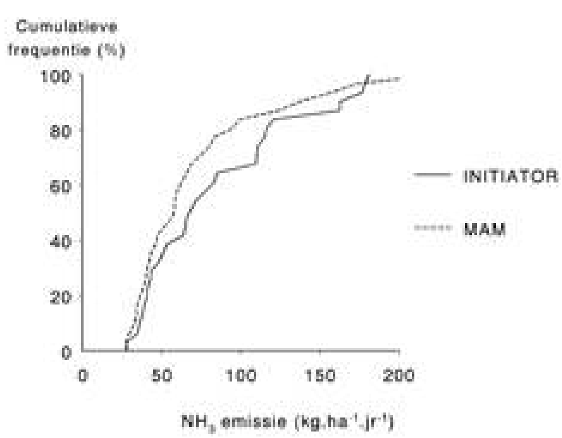
1) M: management measure; T: technical measure
2) Background information is given in (Kros et al., 2003).

Results and Discussion

Plausibility

Ammonia emissions

Compared with the reference model mam, initiator calculates a national total of NH_3 emission which is about 10% higher for the year 2000, 141 vs 155 Gg NH_3 (Table III). Concerning the relative high uncertainty in ammonia emission this deviation is considered acceptable. The 95% uncertainty interval for mam calculations ranges from 131 to 180 Gg NH_3 (Gijlswijk et al., 2004). Results show that initiator calculates higher emissions for housing and application than mam, whereas for grazing it calculates lower emissions. Deviations were mainly due the use of (slightly) different emission fractions and the amount of animal manure production (Kros et al., 2004).



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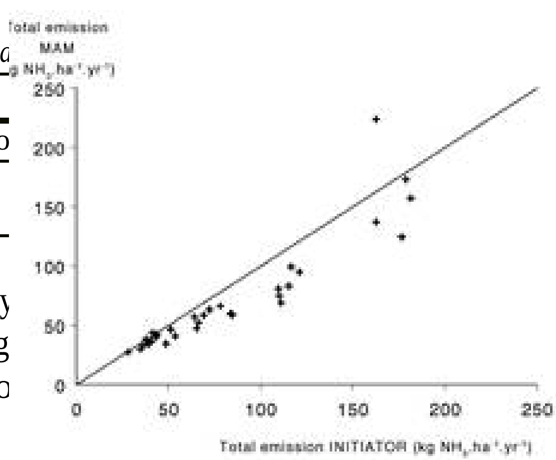

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Figure II CDF of the NH_3 emissions for the year 2000 as calculated with initiator and with mam (left) and the corresponding X-Y-plot (with a 1:1 line) for each of the 31 manure regions within the Netherlands (right)

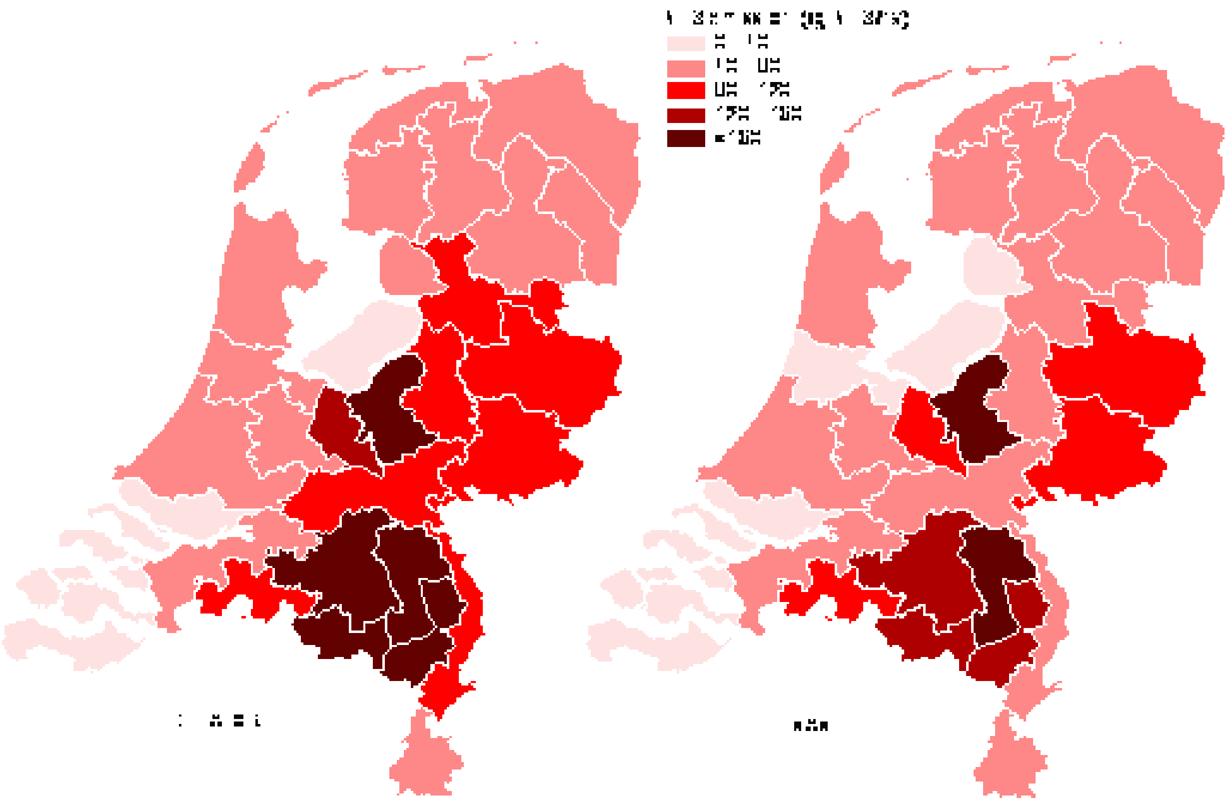

Figure III Comparison of the NH_3 emissions for the year 2000 as calculated with initiator (left) and with mam (right) for each of the 31 manure regions within the Netherlands

The spatial distribution of NH_3 emissions as calculated with initiator corresponds reasonably well with that of mam (Figure III). Apparently, the overestimation by initiator does not disturb the spatial pattern. Figure II shows that both models calculate relatively high emissions in the Central and Southern part of the country, which corresponds with the occurrence of intensive animal husbandry in this part of the country. NH_3 emissions in other parts of the country were mainly caused by dairy farms and the application of transported manure from the Central and Southern part of the country.

Nitrous oxide emission

An overview of the calculated yearly national N_2O emissions (Gg N yr^{-1}) from agriculture calculated with initiator and the IPCC method for the year 2000 is given in Table IV. As with NH_{3},, the total emission flux as calculated with initiator was clearly higher (*c* 20%) than the values calculated with the reference method. The correspondence, however, is considered remarkable, especially when the large uncertainties in reported IPCC values (Olivier et al., 2002) are concerned. A range of 7 to 36 Gg N.yr^{-1} was calculated with initiator for the year 2000, while for the standard IPCC method a range from 12 to 25 Gg has been calculated when using the same input uncertainty (De Vries et al., 2003b). Results show that relatively large deviations do occur for peat soils, which is due to a difference is process description. The ipcc method do assume a constant N_2O emission by mineralisation (5-10 kg.ha^{-1}.yr^{-1}) whereas initiator calculates the N_2O emission as a function of the ground water level, influencing the mineralisation, nitrification and denitrification rates.

Table IV Calculated yearly N_2O-N fluxes from agriculture in the Netherlands as calculated with initiator (De Vries et al., 2003b) and a spatial explicit version of the official IPCC method [1)] for the year 2000

Method	N_2O fluxes from agriculture soils (Gg N yr^{-1})				
	Sand dry	Sand wet	Clay	Peat	Total
initiator	2.6	3.3	5.4	10.2	21.5
ipcc	4.0	3.2	5.5	5.4	18.1

1) Method applied to the schematisation used in this study and the officially reported value for the Netherlands (Olivier et al., 2002)

A comparison of the spatial distribution of N_2O emissions by initiator and the spatial explicit version of the ipcc method is given in Figure IV. From this figure it is clear that both methods result in comparable spatial distributions and that the spatial variation in N_2O emissions can be rather high. The relatively high emissions from peat soils, as calculated by both methods, can be recognised in the Central and Northern part of the country.

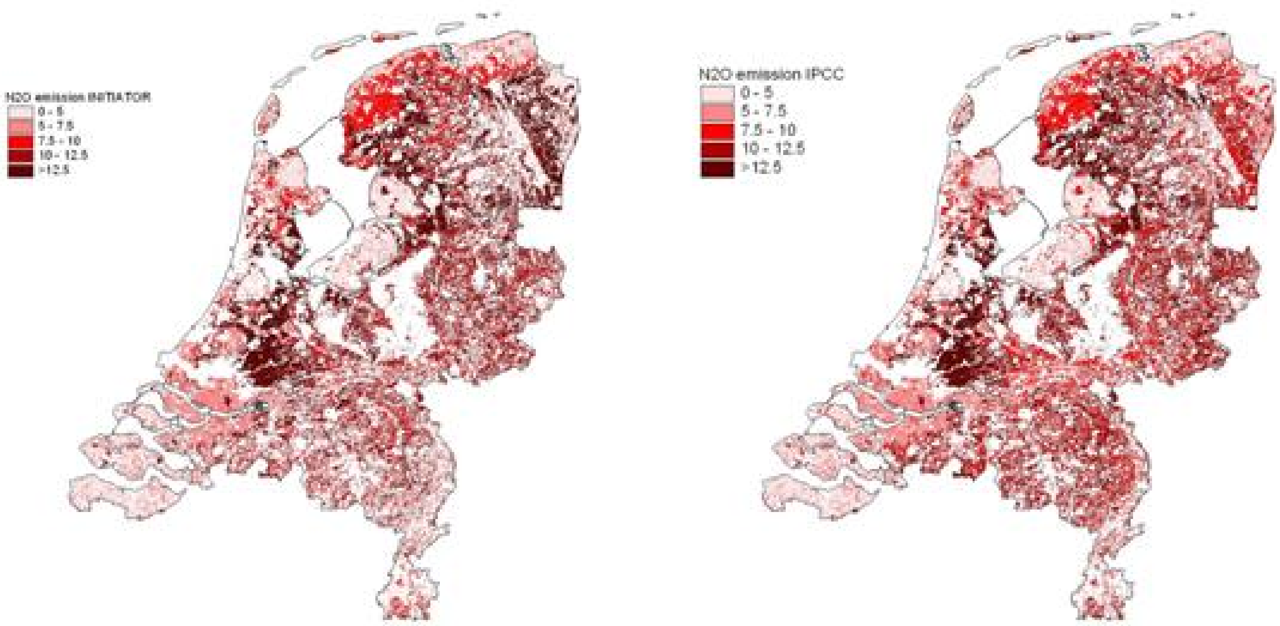

Figure IV Comparison of the N_2O emissions (kg N ha^{-1} yr^{-1}) for the year 2000 as calculated with initiator (left) and with a spatial explicit version of the ipcc method (right) for a 250 m grid

Nitrate concentration in ground water

Nitrate concentrations in groundwater as calculated with initiator were compared with results calculated with the reference model stone for the year 2000 (Figure VII). Both models were run with the same input as calculated with the model clean (Van Tol et al., 2002) for the year 2000.

Results show that the spatial pattern as calculated with initiator is quite comparable with that from stone (Figure V and Figure VI). initiator calculates higher concentrations for some grid cells in the Northern and Southern part of the country. This indicates that initiator calculates a lower N loss due to either denitrification, emission, immobilisation or uptake for these plots. Further examination is needed to check whether this is caused by a difference in parameterisation or process formulation.

From the comparison of the initiator NO_3 concentrations (reference run) with the observation from the national monitoring network (LMM), an underestimation was found for all soil types (Table V). initiator calculated an overall mean which was about 30% lower than the observed mean. The highest deviation was found for clay soils (about 65%) and the lowest for sandy soils (about 10%). Most likely, this was due to an over estimation of the used reference denitrification rate, since the NO_3 concentration is very sensitive for the rather uncertain denitrification rate parameter (De Vries et al., 2003a).

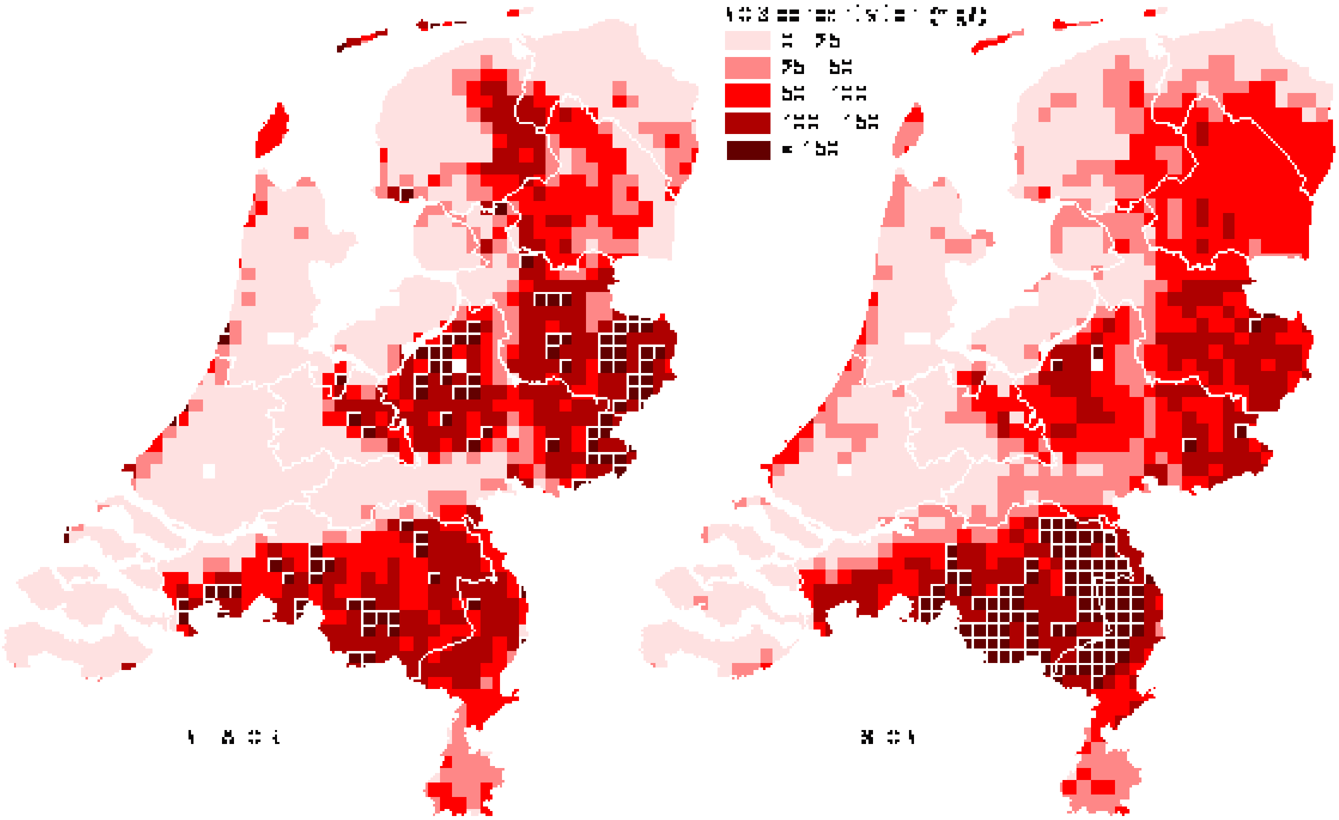

Figure V Comparison of the NO_3 concentration (mg NO_3 l^{-1}) as calculated with initiator (left) and with stone (right) for a 5×5 km^2 grid for the year 2000

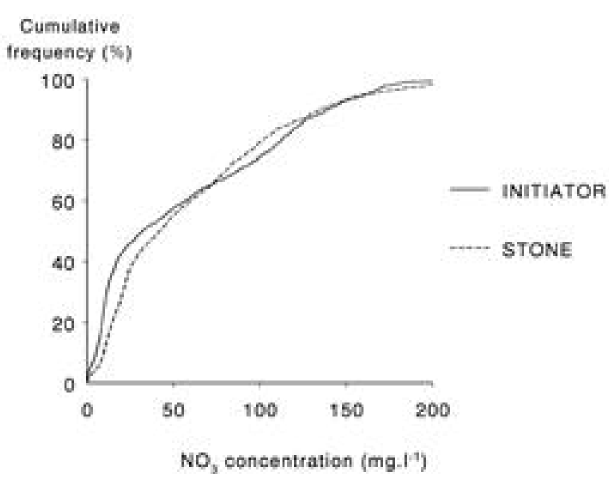

Figure VI CDFs of the NO_3 concentration (mg NO_3 l^{-1}) as calculated with initiator and with stone

In order to get an impression of the uncertainty due to denitrification, Table V also presents mean initiator results obtained while using the 25 (P25) and 75 percentile (P75) of the uncertainty range in the denitrification fraction (see De Vries et al., 2003a). For sandy soils and peat soils, the observed mean fits within the bounds corresponding with the high and low denitrification fractions although it is near the upper boundary for peat. For sandy soils, this is also illustrated with the corresponding cumulative distribution functions (see Figure VII). For clay soils, however, the calculated value exceeds the upper limit. However, the LLM dataset for clay soils contains several observations from soil water samples collected in relatively shallow drainage tubes. For these samples denitrification might be lower than those collected in shallow ground water. Nevertheless, this finding gives rise to adjust the denitrification parameters for clay and peat soils used in initiator.

Table V Overall comparison of mean NO_3 concentrations in the upper ground water as calculated with initiator and the LMM observations for the years 1998, 1999 and 2000. initiator runs were performed with the mean (Reference run), the 75 percentile (P75) and the 25-percentile of the uncertainty range in denitrification parameters

	Mean Nitrate concentration (mg l^{-1})			
	Sand (N=130)	Clay (=126)	Peat (N=76)	All (N=332)
initiator Reference run	88.7	15.2	5.8	41.8
initiator P75, high denitrification .	67.4	10.2	4.0	31.2
initiator P25, low denitrification	135.9	27.7	10.2	66.1
LMM observations	95.5	43.9	22.2	59.1

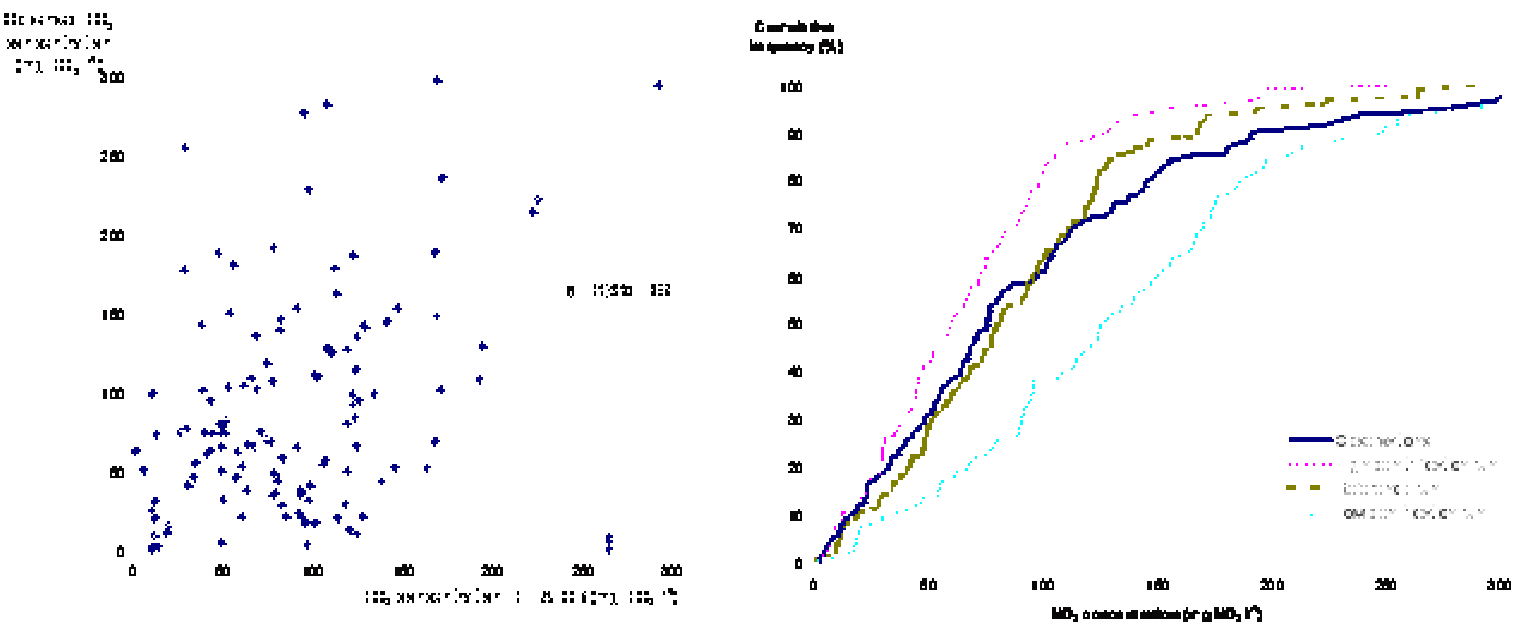

Figure VII Comparison between observed Nitrate concentration (mg NO_3 l^{-1}) and calculated with initiator for 130 sampling locations on sandy soils throughout the Netherlands, using a X-Y plot (left) and a cumulative distribution function (CDF) (right). The CDF also shows the results for a run with low denitrification rate constants and high denitrification constants

Based on the mean (Table V) and on the CDF (Figure VII, right panel) it can be concluded that for sandy soils the initiator results correspond rather well with the LLM observations. However, from the comparison at an individual farm level using a X-Y plot (Figure VII, left panel), it appears that the deviations can be very high. This implies the need for obtaining more accurate site specific input data, especially on hydrology and calibration of the denitrification rate at the farm level. Together with the adjusting of the denitrification rate parameters for clay and peat soils, this will be a subject of future validation studies.

Evaluation of measures

Although the national emission ceiling for ammonia (100 Gg NH_3) is seriously exceeded at present situation (the year 2000), implementation of the management measures results in such a considerable reduction that this ceiling can be achieved (Figure VIII). The most effective measures were improving animal feeding (2), low emission application (6) and low emission housing (8 and 9). The long-term national ceiling of 50 Gg NH_3 could only be achieved in combination with the technical measures. For N_2O, the management measures were by far more effective than the technical measures. The evaluated management measures were by far enough to achieve a 6% decrease compared to 1990 emissions, which are comparable with those for the year 2000 (see Table IV).

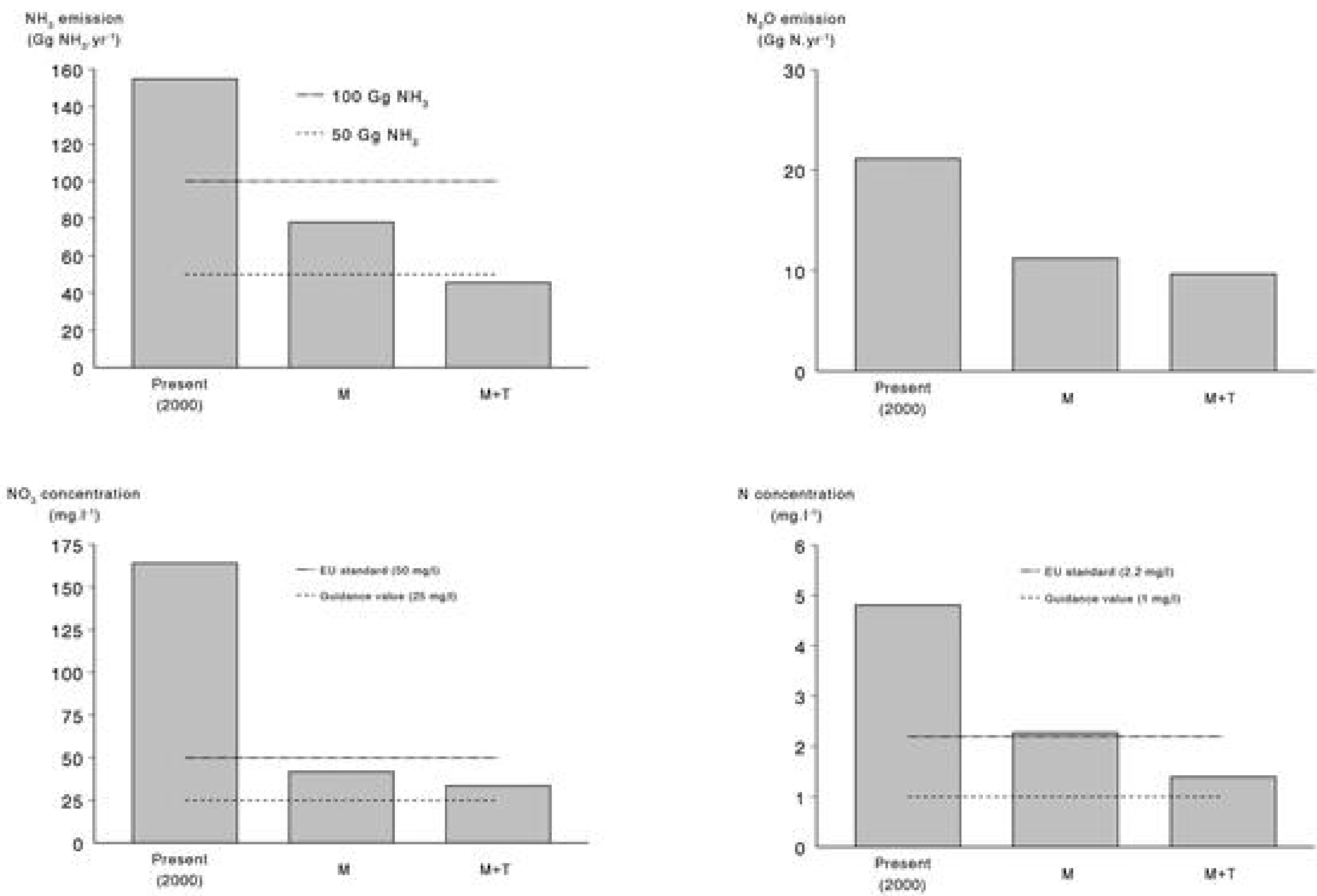

Figure VIII Effects of mitigation measures on the total national ammonia emission (top left), total national nitrous oxide emission (top right), the average NO_3 concentration in upper ground water below dry sandy soils (bottom left) and the national average N concentration in surface water (bottom right)

It was calculated that for the present (year 2000) situation, the average NO_3 concentration in ground water below dry sandy soils exceeds the EU standard of 50 mg NO_3 l^{-1} by far. Implementation of management measures yields a serious decrease; the average concentration drops below the EU standard. Additional technical measures results in only slightly lower average NO_3 concentration. The guidance value is not achievable even with the combined package of measures. Most effective measures were reducing fertiliser use (3) and optimal drainage (5).

For surface waters management measures resulted in a considerable decrease in the average N concentration, but were not sufficient to achieve even on average the EU standard. Contrary to ground water, for surface water technical measures showed to be rather beneficial. This applies to the application of buffer strips that resulted in considerable reductions. The combination of management and technical measures resulted in concentrations below the EU standard.

Generally, N_2O emissions and NO_3 in ground water could be mitigated with management measures only, whereas to mitigate NH_3 emissions and N in surface water both management and technical measures were necessary.

It must be noted that measures leading to national total emissions or national average concentrations below target criteria, as presented in this paper, may still cause exceedances of the used criteria at a smaller spatial scale. This is especially relevant for ammonia emission (exceeding of critical deposition for a particular nature or forest area) and the NO_3 concentration in groundwater.

Cost-benefit of measures

In addition to the quantification of the environmental benefit, the cost-effectiveness of the mitigation measures was also quantified (Table VI) for various types of N emission. For all considered outputs, the technical measures were by far more expensive than the management measures, whereas the calculated reduction percentage is larger for the management measures. As a result the cost-effectiveness for the management measures was substantial higher for all considered outputs. Note, however, that these results are biased because the cost effectiveness of technical is determined after the management measures were already effectuated. On the other hand the chosen sequence of first applying the management measures, that are relatively easy to implement and cheap, is the most rational sequence.

Table VI Cost benefit analysis for the effect of the set of management measures (M) and technical measures (T) on the total ammonia emission ($NH_{3\ tot}$), mean NO_3 concentration in ground water ($NO_{3\ gw}$), and the mean N concentration in surface waters (N_{sw})

	Reduction percentage with respect to the previous situation		Reduction percentage per 100 M€	
	M	T	M [1]	T [2]
NH_3	50	41	10.6	3.2
$NO_{3\ gw}$	65	0	13.9	0.0
N_{sw}	52	39	11.1	3.3

[1] Total costs of all management measures: 471 M€

[2] Total costs of all technical measures: 1284 M€

Management measures were most cost effective in reducing the NO_3 concentration in ground water, whereas the technical measures were very cost ineffective for this output. With technical measures the NH_3 emission and N leaching to surface water could be reduced further, but with a cost effectiveness that is about three times as low compared to the management measures.

Conclusions

The many interactions in the N cycle and the cascade of multiple environment and public health effects make it necessary to have a simple, accurate instrument to show all consequences of different N abatements options targeted at decreasing environmental impacts. A simple and transparent integrated system such as initiator will provide insight into various emissions to soil, air and water.

Plausibility

initiator produced results on NH_3 emission and N_2O emission at a national scale that are comparable with results from existing reference models. Uncertainty in manure production and distribution and emission coefficient are of more relevance than simplifications in model structure. It is therefore advised to put more emphasis on getting reliable emission coefficients rather than more sophisticated model descriptions.

NO_3 concentrations calculated with initiator were comparable with the results of the reference model stone. This is also true when comparing the results with observations for sand and peat soils. However, for clay soils the concentrations were underestimated. This give rise to adapt the denitrification rate parameters.

Although this plausibility study identified differences between the results of initiator and the corresponding results of the reference models, generally the differences are within acceptable ranges. Nevertheless, some results give rise to adjust either the model parameterisation or the process formulation. However, more emphasis must be put on adjusting the current parameterisation rather than extending the process description. Especially when concerning the relatively large uncertainties which are inherent within this domain. However, it has been illustrated that initiator produced plausible results which makes it an acceptable, quick, flexible and verifiable tool for integrated analysis and the evaluation of mitigation measures on a national scale. For this type of analysis initiator can serve as a substitute for a whole collection of separated complex reference models.

Mitigation and cost effectiveness

With the implementation of the evaluated management measures the short term (2010) national goals for NH_3 emission and N_2O emission are achievable. The long-term goal (2030-205) of 50 kton for NH_3 is only achievable with additional technical measures, which would require serious investments. For nitrous oxide emission reduction management measures are more promising that the technical measures.

For the reduction of NO_3 the concentrations in ground water management measures are most promising. For surface waters management measures are effective, but less effective than the are for ground water. For technical measures it is the other way around. On average the EU ground water standard is achievable with only management measures, whereas for surface water it is necessary to implement both management and technical measures. It must be noted, however, that achieving the EU standard on average does not imply that achievement is guaranteed in all areas. The management measures are most cost-effective for all considered outputs. For NO_3 leaching technical measures are very cost ineffective.

Implications for Southeast Asia

The experience and knowledge on parameterisation and application of an integrated N model like initiator might also be beneficial for South East Asia. High cost-effectiveness for management options might be rather relevant information for China and South-East Asia. Good support and assisting farmers and their organisation in changing their management strategies could be rather promising.

At the moment we are working at an integrated research proposal on developing an N Decision Support System for two counties in the Yangtze Delta region in China that will be based on the system developed for the Netherlands. Using this system and the embedded experience as a starting point will allow a much faster progress rather than starting from scratch.

Acknowledgement

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