Water fluxes and diffuse nitrate pollution at the river basin scale – Interfaces for the coupling of agroeconomic models with hydrologic models

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Abstract: The REGFLUD-project, commissioned by Germany’s Federal Research Ministry (BMBF), addresses the problem of reducing diffuse pollution from agricultural production. The objective of the project is the development and application of multi-criteria scientific methods, which are able to predict diffuse pollution in river basins subject to economic feasibility and social acceptability. The selected river basins (the entire Ems basin and sub-catchments of the Rhine) cover a variety of landscape units with different hydrological, hydrogeological and socio-economic characteristics. This paper focuses on the analysis of the effects of certain policy measures to reduce diffuse pollution by nitrogen. For this purpose a model system consisting of an agricultural sector model, a water balance model and a residence time/denitrification model was combined and applied. First results indicate a wide range of annual nitrogen surpluses for the rural areas between less than 10 N ha⁻¹·a⁻¹ up to 200 kg N ha⁻¹·a⁻¹ or more, depending on the type and intensity of farming. Compared to the level of nitrogen surpluses the level of nitrogen inputs into the surface waters is relatively moderate because of degradation processes during transport in soil and groundwater. Policy impact analyses for a nitrogen tax and a limitation of the livestock density stress the importance of regionally adjusted measures.

Key words: Diffuse water pollution, river basin management, multicriteria assessment, agro-environmental policy evaluation, denitrification, nitrate leaching

1. INTRODUCTION

In Germany, considerable progress has been achieved towards the improvement of water quality. However, diffuse water pollution, a source largely attributed to agricultural production continues to be of concern. As described by Goemann et al. (2003), a wide range of problems concerning nutrient pollution of water bodies are prevalent in the Ems basin and subcatchments of the Rhine. It is to be expected that political measures towards a solution of these problems will have different effects on the reduction of nutrients in the different water bodies. Thus, the efficiency of measures has to be evaluated, taking into account both socio-economic conditions as well as natural site conditions. On one hand the different historically evolved and partly established socio-economic conditions in the study area such as agricultural farm structures or the structure of water protection as well as water supply and sewage disposal are an important prerequisite for the development of effective nitrogen reduction measures. On the other hand, natural conditions, which determine pathways and transport of diffuse nutrient surplus into surface waters, have to be considered. The Linkage of the agricultural sector model RAUMIS (Henrichsmeyer et al. 1996) with the hydrological model GROWA (Kunkel & Wendland 2002) and the reactive nutrient transport model WEKU (Kunkel & Wendland 1997) represents a consistent link-up of the environmental pressure indicator “agricultural nutrient surplus” with the environmental state indicator “nutrient loads of water bodies” and the environmental response indicator “nutrient reduction measure”. This paper focuses on the application of the integrated agro-economic / hydro(geo)logic model system for the management of diffuse nitrogen fluxes exclusively.
2. METHODOLOGICAL APPROACH

Combining agroeconomic and hydro(geo)logical models is a scientific challenge. The most efficient way to homogenize and adjust models from different scientific disciplines is the development of a common model interface for data exchange. This interface has to guarantee a uniform definition (e.g. scope of representation, spatial and temporal dimension) of variables being exchanged within the model network. Figure 1 shows the integration of the agricultural economic model RAUMIS with the hydrological models GROWA and WEKU.

A central interface between RAUMIS and GROWA/WEKU are regional nutrient surpluses and land use patterns. According to requirements specified above, it has to be considered that the two models are using different regional resolutions: raster cells in the hydrological models and administrative units in the RAUMIS model. This is due to the different data sources: while GROWA/WEKU uses land use maps, RAUMIS employs agrarian statistical data. For this reason, regional nitrogen balances calculated by RAUMIS as averages for the agricultural areas (AA) in the individual administrative units cannot be directly used as input variables in GROWA/WEKU. As a first step, these nitrogen surpluses are disaggregated and geographically referenced on raster cells as required by GROWA/WEKU.

![Figure 1. Integrated agroeconomic / hydro(geo)logic model system](image)

In the agricultural sector model RAUMIS (Henrichsmeier et al., 1996), a set of agro-environmental indicators is linked to agricultural production. Currently, the model comprises indicators such as fertilizer surplus (nitrogen, phosphorus and potassium), pesticides expenditures, a biodiversity index, and corrosive gas emissions. These indicators help to evaluate direct and indirect environmental impacts of policy driven changes in agricultural production. Regarding diffuse water pollution the indicator “nitrogen surplus” is of particular importance. Agricultural statistics with data, e.g. on crop yields, livestock farming and land use, were used to balance the nitrogen supplies and extractions for the agricultural area. The long-term nitrogen balance averaged over several vegetation periods is calculated considering the organic nitrogen fertilization, the mineral nitrogen fertilization, the symbiotic N-fixation, the atmospheric N-inputs and the N-extractions with the crop substance. As a rule, the difference between nitrogen supplies, primarily by mineral fertilizers and farm manure, and nitrogen extractions, primarily by field crops, leads to a positive N-balance (Goemann et al. 2003).

The displacement of N-surpluses into surface waters is coupled to the runoff components. Against the background of a long-term treatment for the hydrological period 1961-1990, runoff was distinguished into direct runoff and groundwater runoff. Whereas direct runoff reaches the surface waters within short time periods (within about a week), groundwater run-off needs much more time.
(years) to percolate into surface waters. The runoff components were quantified area-differentiated as a function of climate, soil, geology, topography and land use conditions using the GROWA model (Kunkel & Wendland 2002). The ratio between groundwater recharge and total runoff was taken as a measure for the extent diffuse nitrogen surpluses, which are displaced from soil to groundwater (Wendland et al. 2002).

During transport through the soil and the groundwater nitrogen surpluses may be denitrified to molecular nitrogen. According to a review by Köhne & Wendland (1992) denitrification losses in the soil occur mainly in the root zone in case of low oxygen and high water contents as well as high contents of organic substances. In a Michaelis-Menten kinetics approach these denitrification conditions were combined with the nitrogen surpluses given by RAUMIS and the residence times of the percolation water in the root zone calculated as a function of average field capacity and the percolation runoff level (Behrendt et al. 2003). Reactive nitrate transport in groundwater was modelled using the stochastic WEKU model (Kunkel & Wendland 1997) on the basis of a first order reaction depending on the nitrogen inputs into the aquifer, denitrification conditions in groundwater and groundwater residence times.

In the first step groundwater velocities are calculated according to Darcy’s law from hydraulic conductivity, effective yield of pore space of the aquifer and the slope of groundwater surface (hydraulic gradient). The calculation of the residence times of the groundwater runoff is performed in a second step. Based on groundwater contour maps, a digital relief model of the groundwater surface is generated. This is analysed paying attention to information on the water network as well as the groundwater discharge or transfer areas with respect to lateral flow dynamics and groundwater-effective recipients. The residence times of the groundwater runoff are then obtained for each initial grid by summation over the individual residence times in the grids resulting from the groundwater velocities and individual flow distances along the flow path until they enter a surface water.

The WEKU model was extended by a module for the quantification of nitrate degradation in groundwater. According to extensive field studies by Böttcher et al. (1989) and Pätsch (2004) in catchment areas in the North German Lowlands and van Beek (1987) for a site in the Netherlands a first order denitrification kinetics has been assumed with a reaction constant in the range of 0.17 to 0.56 a\(^{-1}\). This corresponds to a halving of the nitrogen leached to the groundwater after a residence time between 1.2 and 4 years. Rather simple indicators, such as the presence of Fe (II), Mn (II) and the absence of O\(_2\) and NO\(_3\) can be used to decide whether a groundwater province has hydrogeochemical conditions in which denitrification is possible or such transformation of nitrogen can be neglected (Wendland & Kunkel 1999, Kunkel et al. 1999).

3. CASE STUDY RIVER BASINS

Two German river basins, namely the Ems basin (12,900 km\(^2\)) and several Rhine sub-catchments, comprising the river basins of the Sieg, Wupper, Ruhr and Erft, (in total 12,100 km\(^2\)), have been selected as study areas in order to cover a wide range of different landscape units with different hydrological, hydrogeological and socio-economic characteristics. The administrative bodies that correspond to RAUMIS regions (“Landkreise”) cover an area of 32,700 km\(^2\) in total and thus over-extend the catchment areas of about 30%.

The river Ems basin is located in the North-German Plain. Agriculture plays an important role in comparison to the German average: Agricultural area (AA) accounts for about 62 % of total area and production is dominated by intensive animal husbandry which is more competitive on the prevailing less fertile sandy soils than cash cropping. Farmers typically grow fodder crops, such as silage maize and corn-cob-mix on arable land. These generate higher yields than permanent grassland and enable a higher livestock production. This production structure explains the visible correlation between shares of arable land and livestock densities (LD) that are displayed in for the regions within the Ems catchment.
The situation is quite different in the Rhine sub-catchments mentioned above. A striking socio-economic difference is the population density being three times higher than in the Ems basin. Settlements, traffic, and industries, in addition to forests, play an important role such that agricultural area amounts to 30% of total area.

Eastern parts of the Rhine sub-catchment are located in consolidated Palaeozoic rock areas with high total area runoff levels, dominated by fast (direct) runoff components. These conditions hamper tilling of soil so that permanent grassland dominates land use. Farmers specialise in cattle and milk production on a fairly extensive level. All these regions can be classified as areas with a high risk of surface water pollution, e.g. of reservoirs. On the other hand it can be expected that nutrient reduction measures will improve surface water quality in these areas rapidly. Western parts of the Rhine sub-catchment are located in the unconsolidated quaternary rock area of the lower Rhine bay with considerable ground water recharge levels. Because of the very fertile loess soil, intensive cash cropping is the main agriculture production activity. These regions feature a share of arable land of more than 90 % of AA and low live-stock densities.

4. RESULTS AND DISCUSSION

4.1. Diffuse nitrogen surpluses

The nitrogen surpluses calculated with the RAUMIS model were calculated for a projection of the development under the current Common Agricultural Policies (Agenda 2000) of the European Union for the year 2010. This surplus is used as reference scenario instead of the actual situation, as comparative static policy impact analyses for a future target year require a scenario of reference because various parameters are changing in the long-run in addition to the variations of policy measures being investigated. Typically the scenario of reference is a projection of the development
under “business as usual”. Thus, nitrogen surpluses indicate the amount of nitrogen that potentially leaches into groundwater and surface waters. Deviating from the reference scenario, alternative policies and regulations are imposed on the model keeping all other parameters and constraints constant. Comparison to the actual situation would lead to a convolution between the effects of these already implied policies and the effects of the investigated reduction measures.

On average, the calculated nitrogen surpluses for the agricultural acreages based on this reference scenario amounts to about 130 kg N ha\(^{-1}\)a\(^{-1}\) in the Ems basin, whereas the average for the investigated sub basins of the Rhine basin is much less (74 kg N ha\(^{-1}\)a\(^{-1}\)), due to the generally less intensive agriculture. In Figure 3 the nitrogen surpluses from agriculture are plotted for the agricultural areas within each district. Especially in regions with area-independent animal processing (intensive animal production) nitrogen surpluses result from both animal excretions and mineral fertilizers as well. This kind of land use management occurs mainly in the north-western part of the Ems basin. In addition, the western sub basins of the Rhine basin, dominated by fertile loamy soils and favourable climatic conditions display significant nitrogen surpluses because of intensive growing of commercial and specialty crops. Low nitrogen surpluses are calculated for regions with mostly forage crops production, which is typical for the eastern parts of the Rhine basin.

![Figure 3. Annual nitrogen surpluses from agriculture.](image)

The nitrogen surpluses from agriculture, calculated as averages on a district level, are disaggregated with respect to the current land use. For this purpose the CORINE land cover land use classes arable land and pasture are used as disaggregating criteria. In addition, atmospheric nitrogen inputs of 30 kg N ha\(^{-1}\)a\(^{-1}\) and an asymbiotic nitrogen fixation of 1.4 kg N ha\(^{-1}\)a\(^{-1}\) have been considered as lump sum amounts. For areas representing non agricultural regions in the REGFLUD study areas, urban areas and forests, only the atmospheric inputs and asymbiotic nitrogen fixation were considered. Denitrification in the soils has been modelled using a Michaelis-Menten kinetics. In this way the nitrogen surpluses given by RAUMIS were reduced in some areas by up to 50%, e.g. in areas where loamy soils with a high water storage capacity a high organic carbon content occur. The remaining nitrogen leaching from the root zone is transported to the surface waters either by direct runoff or leaches into groundwater according to the calculated base flow ratio. In the north-western part of the river Ems basin or in the mountainous regions in the eastern part of the
Rhine basin, groundwater runoff is not more than 20 to 40 % of the total runoff. In these regions direct runoff is the dominant pathway of nitrogen input into surface waters. In other areas, e.g. the central part of the Ems basin, groundwater runoff is the main pathway for nitrate entries into surface waters.

4.2 Nitrogen inputs into surface waters via direct runoff and into groundwater via groundwater recharge

The results of coupling nitrogen leaching from the root zone with runoff values are shown in Figure 4 and Figure 5.

Figure 4 shows the corresponding nitrogen input into surface waters via direct runoff. In this case no further denitrification in the unsaturated zone is considered. It becomes clear, that N-inputs to surface waters from direct runoff are important especially in the marshy areas of the Ems basin and the mountainous regions in the Rhine basin.

![Figure 4. Nitrogen inputs into rivers from direct runoff for the reference situation.](image)

Figure 5 shows the nitrogen inputs into the aquifer via groundwater recharge. High nitrogen leaching to the groundwater is calculated for regions with a high groundwater runoff portion and high nitrogen surpluses, which is important in particular for the central part of the Ems basin. In the sub-basins of the Rhine the nitrogen leaching to groundwater is less important due to the low nitrogen surplus level (Figure 3) on one hand and the large portion of direct runoff in the mountainous regions of the Rhine basin.

4.3 Nitrogen inputs into surface waters via groundwater runoff

Transport and denitrification in the aquifer is calculated using the WEKU model taking into account groundwater residence times and natural nitrate degradation in the aquifers. Calculated groundwater residence times range between less than 1 year and more than 150 years. Long residence times result both from small groundwater velocities as well as from long flow paths up to the recipient, pointing at the long time periods, after which nitrate inputs into the aquifer can contribute to the pollution of surface waters in
some regions. Short residence times generally result for areas in the vicinity of rivers and/or regions with high groundwater velocities.

![Figure 5. Nitrogen leaching into the upper aquifer for the reference situation.](image)

The quantification of the parameters of denitrification kinetics in groundwater was done separately for the groundwater bearing formations occurring in the river basins. In total, about 1050 groundwater samples were evaluated and classified with respect to nitrate-degrading capacity. From this analysis the groundwater bearing formations glaciofluvialite sands and moraine deposits, both occurring in the river Ems basin, were classified as nitrate degrading. In contrast, most aquifers in the investigated sub basins of the river Rhine, predominantly consolidated rocks (e.g. shists and limestones), showed usually non-nitrate degrading conditions.

The remaining nitrogen outputs to surface waters from groundwater were calculated by combining the N-leaching into the aquifers and the reactive N-transport in the aquifers. The result is shown for the reference status in Figure 6 for the initial cells for which the inputs into the soil have been calculated.

It can be seen that nitrogen intakes in the vicinity of surface waters and high nitrogen leaching levels contribute considerably to the groundwater-borne nitrate inputs to the surface waters. Even with good conditions for a complete degradation of nitrate in the aquifer, the brief residence times are not sufficient for an adequate degradation of high nitrate inputs. There is, furthermore, a hazard potential in many regions where high nitrate inputs are associated with relatively short residence times of the groundwater, as well as restricted and/or insignificant degradation conditions in the aquifer. These regions include almost the whole Rhine catchment area. The loose rock aquifers in the northern part of the Ems basin show an opposite behaviour. There, even high nitrogen inputs into the groundwater systems result only in very slight nitrate inputs to surface waters after transport through the aquifers. Long groundwater residence times and good denitrification conditions cause high denitrification of up to 90% of the inputs into the aquifer systems. As a consequence, groundwater is almost nitrate-free when it enters the rivers after transport through the aquifer systems.
The observed N-loads in rivers represent the sum of all N-inputs by the different diffuse and point source intake pathways. The residence times of direct runoff and groundwater runoff differ significantly not only between the different input pathways but also from intake site to intake site. Thus, the input to surface waters from a certain intake location via direct runoff refers to an input from of less than 2 years ago in general, whereas the inputs via groundwater for the same location refers to an input from some decades ago. Hence, for the calibration and verification of the integrated RAUMIS-GROWA-WEKU model calculations of nitrogen river loads concerning the past inputs have to be considered as well. This has been done using nitrogen surpluses calculated with RAUMIS for the reference periods of the last decades.

The validity of modelled groundwater-borne nitrogen inputs into surface waters was checked following a procedure suggested by Behrendt et al. (2002). At first the measured N-loads were corrected by the point N-inputs (Behrendt et al. 2000). In order to avoid effects of the N inputs by direct runoff, only observed nitrogen concentrations at low flow conditions were considered. Additionally only observed values at temperatures below 5° C were taken as reference in order to avoid effects of nutrient retention in rivers. Following this procedure the comparison of the modelled groundwater-borne nitrogen inputs into surface waters with the observed river load data of 54 sub-catchment areas show only relatively small differences to the observed values (about 10-20 %).

4.4 Evaluation of N reduction measures

Two measures aiming at a reduction of agricultural nitrogen use have been calculated and compared to the simulation of the intakes into surface waters according to the input pathways on the basis of the nitrogen surpluses for the references status. The first scenario reflects to nitrogen reduction by rising a tax on the price of mineral fertilizer of 200 v%. The implementation of a nitrogen tax has been repeatedly discussed since the mid-eighties. The second scenario reflects on the limitation of the livestock density (LD) to not more than 1.0 livestock units (LU) per ha. The introduction of such a measure is proposed by environmental associations (BUND 2004, BBU 2001) and also discussed in a study of USDA-ERS (2000). The effects of these scenarios on the
potential reduction of N-surpluses from agriculture compared to the reference status (Figure 3) are shown in Figures 7 and 8.

It can be seen easily that the measures affect the nitrogen surpluses in the two selected study areas differently. The nitrogen tax (scenario 1, Figure 7) noticeably reduces nitrogen use in all arable farm regions such as the western part of the Rhine sub basin and the southern part of the Ems basin by at least 10-25 kg N ha$^{-1}$·a$^{-1}$. It has also an effect on the N-surpluses in the central parts of the Ems basin. This area is characterized by intensive animal production and animal food production on associated animal crop food production areas (e.g. maize), where not only farm manure is applied but mineral fertilizers as well. In regions with intensive livestock production the introduction of a tax on mineral fertilizers alone does not mitigate the hot-spot problems since the manure supply is not affected. The impact of a measure aiming at the limitation of live-stock density (LD) to not more than 1.0 livestock units (LU) per ha is shown in Figure 8. It becomes obvious, that in large parts of the Ems basin most of all this measure would be effective and would lead to a potential reduction of the N-surpluses to at least 25-50 kg N ha$^{-1}$·a$^{-1}$. In contrast however, scenario 2 has almost no effect in regions with extensive animal production like in major parts of the Rhine sub basins.

In the Figures 9 to 12 the effects of the two scenarios on the N-inputs into surface waters are shown differentiated according to the input pathways direct runoff and groundwater runoff. The impact of the implementation of a nitrogen tax (scenario 1) is shown in Figure 9 and 10.

Compared to the reference situation this measure would reduce the total N-input into surface waters by about 25 % in both catchments. The impact of the implementation of a limitation of livestock density (LD) to not more than 1.0 livestock units (scenario 2) is shown in Figures 10 and 12.

Because scenario 2 affects only areas with intensive animal production, significant N-reduction can be achieved only in the Ems basin. Compared to the reference situation this measure would reduce the total N-input into surface waters in Ems basin by about 35%. In the Rhine sub basins almost no reduction (less than 5 % of the total N-inputs to river) can be obtained with scenario 2.
Most of all the N-inputs to surface waters derive from direct runoff. Due to the (in general) short residence times of percolation water in the soil and the corresponding low denitrification rates in the soils the N-inputs into surface waters via direct runoff are the more important input pathway. As could be shown with the two scenarios, a reduction of the N-surpluses in the study areas has direct impacts on the N-inputs into surface waters via direct runoff.

The N-inputs into surface waters via groundwater runoff are in general less important and not significantly reduced by the reduction measure. Displaying no significant denitrification conditions in the groundwater bearing units, in the sub-basins of the Rhine this is only due to the relative low N-input level into groundwater. The large N-inputs into groundwater calculated for the Ems basin however, are denitrified to a large extent independent from the input level due to the generally reduced aquifers in the Ems basin. Thus the groundwater borne N-inputs into rivers for the reference situation as well as for the N-reduction scenarios have a minor important impact on the
groundwater-borne N-inputs to surface waters. Only for areas in the vicinity of surface waters the reduction measures may lead to a reduction of the nitrogen input via groundwater runoff.

5. CONCLUSION AND OUTLOOK

In Germany, the water pollutions caused by diffuse nitrogen from agriculture are regionally different. Using the nitrogen surpluses as an indicator to detect or classify “hot-spot” regions, the Ems catchment seems to be quite endangered by N-inputs from agriculture. However, a direct inference from the risk indicator “nitrogen surplus” being calculated with the agricultural sector model RAUMIS to actual depositions of nitrogen into water bodies is limited since natural site conditions (e.g. nitrogen degradation capacities, residence times, etc.) vary considerably among regions.

These natural conditions are accounted for in the hydrological and hydrogeological models GROWA and WEKU, which were used to quantify the nitrogen inputs into the surface waters from the different transport pathways. From the results of this study we conclude that in the groundwater systems of the river Ems basin about 90% of the diffuse nitrogen input into the ground water is degraded in groundwater due to a long groundwater residence time and favourable denitrification conditions. There, groundwater borne nitrate input into the surface waters turned out to be relatively low even if the region were addressed as a “hot-spot” in terms of total nitrogen surplus from agriculture.

The agricultural policy impact analysis pointed out that nitrogen reduction measures affect the regional nitrogen use differently. In particular, the investigated effects of a tax on mineral nitrogen or a limitation of the regional livestock density highlight that the mitigation of the diffuse water pollution problems require regionally adjusted measures. However, it could also be shown that in areas where natural degradations capacities are missing only very drastic measures lead to a significant improvement of the nitrogen load of surface waters.

The networking of the agro-economic model RAUMIS with the hydro(geo)logic models GROWA and WEKU has shown, that the very complex interactions between the driving-force indicator “diffuse nitrogen surpluses” and the state indicators “nitrogen loads in surface waters and groundwater” can be analysed in a consistent and regionally differentiated way. The synergetic effects shows the potential of interdisciplinary model networks for the implementation of political measures aiming at the sustainable management of nitrogen fluxes in river basins.

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