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Management, regulation and environmental impacts of nitrogen fertilization in Northwestern Europe under the Nitrates Directive; a benchmark study

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Abstract

Implementation of the Nitrates Directive (NiD) and its environmental impacts were compared for member states in the Northwest of the European Union (Ireland, UK, Denmark, The Netherlands, Belgium, Northern France and Germany). The main sources of data were national reports for the third reporting period for the NiD (2004–2007) and results of the MITERRA-EUROPE model. Implementation of the NiD in the considered member states is fairly comparable regarding restrictions for where and when to apply fertilizer and manure, but very different regarding application limits for N fertilization. Issues of concern and improvement of the implementation of the NiD are accounting for the fertilizer value of nitrogen in manure, and relating application limits for total nitrogen (N) to potential crop yield and N removal. The most significant environmental effect of the implementation of the NiD since 1995 is a major contribution to the decrease of the soil N balance (N surplus), particularly in Belgium, Denmark, Ireland, The Netherlands and the UK. This decrease is accompanied by a modest decrease of nitrate concentrations since 2000 in fresh surface waters in most countries. This decrease is less prominent for groundwater in view of delayed response of nitrate in deep aquifers. In spite of improved fertilization practices, the southeast of The Netherlands, the Flemish Region and Brittany remain to be regions of major concern in view of a combination of a high nitrogen surplus, high leaching fractions to groundwater and tenacious exceedance of the water quality standards. On average the gross N balance in 2008 for the seven member states in EUROSTAT and in national reports was about 20 $kg\,N\,ha^{-1}$ lower than by MITERRA. The major cause is higher estimates of N removal in national reports which can amount to more than 50 $kg\,N\,ha^{-1}$. Differences between procedures in member states to assess nitrogen balances and water quality and a lack of cross boundary policy evaluations are handicaps when benchmarking the effectiveness of the NiD. This provides a challenge for the European Commission and its member states as the NiD remains an important piece of legislation for protecting

drinking water quality in regions with many private or small public production facilities and controlling aquatic eutrophication from agricultural sources.

1 Introduction

The main aim of the Nitrates Directive (1991: Directive 91/676/EEC; hereafter referred to as NiD) is to reduce water pollution caused or induced by nitrates and phosphorus from agricultural sources. The NiD is the most important piece of European (EU) regulation for reducing environmental impacts of fertilizer and manure and for increasing nitrogen use efficiency. The gross nitrogen balance, or nitrogen surplus, (Schröder et al., 2004; Vries et al., 2011) is an important indicator to evaluate the environmental impacts of the Nitrates Directive, particularly for the water compartment. This makes the NiD an important supporting instrument for other EU directives i.e. the Drinking Water Framework Directive (98/83/EC), the Water Frame Directive (2000/60/EC) and the Marine Strategy Framework Directive (2008/56/EC). The NiD legally restricts farm application of manure to 170 $kg\,ha^{-1}$ of nitrogen, or in case of derogation to inputs up to 250 $kg\,ha^{-1}$ (Oenema, 2004). The tenacious problem of regional nitrogen (and phosphorus) surpluses is resolved by manure processing and long distance, sometimes international, transport of manure and manure products.

Agricultural practices in general, and more specifically application rates and management of chemical fertilizers and animal manures, vary greatly between and within EU member states. This makes it interesting to compare nitrogen management and regulation between countries and relate this to the observed states and trends of nitrate concentrations in groundwater and surface water. Since the introduction of the NiD in 1991, most EU member states have implemented four action programs (in the period 1995–2012). The EU Commission obliges member states to report on the results of these action programs. It also charged synthesizing studies on these national reports but these reports are not publicly available. However, the EU Commission did publish summaries of the national data and reports in 2007 and 2011. Also evaluations

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of the effectiveness of environmental monitoring programs for the NiD were published (Fraters et al., 2011). However, overall insight into the effectiveness of the NiD in the EU is still limited and rarely published in peer reviewed journals. With the submission of new action programs ahead, this insight could help to improve implementation of the NiD across the EU.

The combination of environmental directives and the Common Agricultural Policy should provide food security and a healthy natural environment in Europe while maintaining a level playing field for the agricultural entrepreneurs (De Clercq et al., 2001). This is particularly true for agriculture in Northwestern EU member states as they compete to provide food to consumers in the, so-called, “London-Berlin-Paris triangle”. This paper compares, evaluates and benchmarks the implementation of the Nitrates Directive in the northwestern member states of the EU. The objective is to relate differences in implementation to differences in structure, intensity and practices of the agricultural sector and to sensitivity of soil water systems to nitrate pollution. Key issues of the NiD addressed in the benchmark are application rates of N in manure, the balance between applied N and crop requirements and water quality in relation to the nitrate target of $50\,mg\,NO_3^-\,l^{-1}$. The comparison is restricted to Denmark, Germany, The Netherlands, Belgium, the UK, Ireland and the northern part of France. Crop and fodder production potential per hectare on comparable soils in these countries are similar. Note however, that within the UK there are four separate governments and in Belgium two, which implement the Nitrates Directive in differing ways. Moreover, all these countries have regions with high livestock densities causing feed requirements to exceed regional feed production, and manure production to exceed regional crop demands.

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2 Materials and methods

2.1 Data sources

This analysis combines various existing studies on implementation of the Nitrates Directive (Dijk and Berge, 2009; Berge and Dijk, 2009), gross nitrogen balances from Eurostat (2012), monitored nitrate concentrations in groundwater and surface water in synthesizing reports (European Commission, 2007, 2011; Fraters et al., 2011) and various national reports on implementation and evaluation of the Nitrates Directive for the third reporting period (Anonymous, 2008a–d; Desimpelaere et al., 2008; Zwart et al., 2008). A complication when comparing water quality data among EU member states (and sometimes within a single member state) to evaluate the NiD are the large differences in monitoring procedures, e.g. with regard to sampling density (Table 1), monitoring frequency and groundwater sampling depth (Fraters et al., 2011; European Commission, 2011), and data and procedures for calculation of nitrogen balances (Panten et al., 2009). In 2007 the total number of sampling sites for groundwater was 31 000 and for surface water 27 000.

2.2 Nitrogen balance

In this study calculation of the gross nitrogen balance (GNB) was based on the OECD method (OECD, 2007). In addition the soil N balance (SNB) is used which sometimes is confused with the soil surface N balance (SSNB). The GNB represents the total potential loading of nitrogen from primary agricultural production to the environment, but excluding N emissions from fossil fuel combustion for energy requirements for e.g. fertilizer manufacturing, housing, transport and soil and crop management and correcting for export and processing of manure. SNB or soil N surplus represents the total potential loading from nitrogen use on agricultural soil, while SSNB represents the total net nitrogen loading to the soil and water compartment.

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$$\begin{aligned}\text{GNB}: &\ \text{fertilizer} + \text{manure production} + \text{other inputs} - \text{net manure export} - \text{crop removal}\\ \text{SNB}: &\ \text{GNB} - \text{N-loss housing} - \text{N-loss storage}\\ \text{SSNB}: &\ \text{SNB} - \text{N-loss manure application}\end{aligned}$$

Other inputs include N deposition and biological N fixation (BNF), where N deposition is the result of NH_3 and NO_x emissions from both agricultural and other sources, mainly transportation and energy generation. Choosing one of the balance indicators for monitoring and evaluation of NiD effects is determined mainly by data availability. Data requirements for GNB are lowest, but GNB does not correct for environmental measures reducing ammonia emission following from other EU directives like the National Emission Ceilings (NEC) directive (2001/81/EC) and the Integrated Pollution Prevention (IPPC) directive (96/61/EC). However, different calculation procedures, particularly for determining manure input and nitrogen removal by crops, and also inclusion or exclusion of N-losses in during housing and storage (difference between gross and net soil balance) and of smaller input items, may need to be taken into account when comparing national or regional nitrogen balances.

For this reason the use of a model for determining the nitrogen balance is an additional valuable tool to evaluate the effectiveness of the NiD. Model approaches are inherently more consistent regarding calculation schemes, but without sound ground validation, have a risk of not accounting for regional differences in response of crop removal and water quality to nitrogen fertilization. For example, in the UK a model approach is used to estimate nitrogen loading as part of the NiD assessments. Loadings are calculated using the NEAP-N model (Lord and Anthony, 2000) along with an urban estimation model (Lerner, 2000). Leip et al. (2008) coupled the economic model CAPRI and the mechanistic biochemical model DNDC for evaluation of the effects of agri-environmental policies on the European environment, for example on groundwater pollution with nitrate. Here we use the model MITERRA-EUROPE to apply a consistent methodology to all countries.

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2.3 MITERRA-EUROPE

The model MITERRA-EUROPE (referred to as MITERRA hereafter) was used to quantify the nitrogen balances and nitrate leaching from agriculture on both EU-27 level, country level, and regional level. MITERRA consists of an input module with activity data and emission factors, a set of measures to mitigate ammonia and greenhouse gas emission and nitrate leaching, a calculation module, and an output module (Velthof et al., 2009; Lesschen et al., 2011). The database of MITERRA is on national and regional level (NUTS 2, according Nomenclature of Territorial Units for Statistics in the EU) and includes data of N inputs, N outputs, livestock numbers, land use, crop types, soil type, and emission factors for NH_3, N_2O, and NO_x, and leaching factors for NO_3.

Crop areas were derived from EUROSTAT at NUTS 2 level and crop yields from FAOSTAT at national level as the EUROSTAT data was incomplete. Grassland yields and N contents of grassland were estimated using the methodology of Velthof et al. (2009), because grassland yields are not available from statistics. The number of livestock in each year was derived from EUROSTAT. Data on annual N fertilizer consumption were collected from FAOSTAT. The N excretion of all livestock categories except dairy cows were obtained from the GAINS model (Klimont and Brink, 2004). A method was developed to estimate the N excretion from dairy cows on regional level based on milk yields, grassland yields, and N inputs (Velthof et al., 2011). The total manure N production was calculated at the NUTS 2 level from the number of animals and the N excretion per animal and then corrected for N losses from buildings and storage. A method was developed to distribute the manure over crops taking account of the maximum manure application of 170 kg N ha^{-1} or higher in case of a derogation. Nitrogen fertilizer was distributed over crops relative to their nitrogen demand, taking account of the amount of applied manure and grazing manure and their respective fertilizer equivalence (Velthof et al., 2009). Further nitrogen inputs include biological N fixation, which is estimated as a function of land use and crop type (legumes) and nitrogen deposition that is derived at NUTS 2 level from EMEP (EMEP, 2010).

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Nitrogen leaching is calculated by multiplying the soil N surplus by a region specific leaching fraction, which is based on soil texture, land use, precipitation surplus, soil organic carbon content, temperature and rooting depth (Table 2). Surface runoff fractions are calculated based on slope, land use, precipitation surplus, soil texture and soil depth (Velthof et al., 2009). The nitrate concentration in leaching water is calculated by dividing the amount of nitrogen leaching from agriculture by the total water flux, which is calculated as the precipitation surplus, derived from the EuroPearl model (Tiktak et al., 2006), minus surface runoff.

3 Results

3.1 Characteristics of agriculture and nutrient use in Northwestern EU

Mean annual temperatures range between 8 and 12 °C, with minimum daily temperatures in January around 0 °C and maximum daily temperatures around 20 °C in July. Mean annual precipitation ranges from values exceeding 1000 mm yr^{-1} in western coastal regions to 500 mm yr^{-1} in Central France, and Eastern UK and Germany (Tiktak et al., 2006). The combination of favorable climatic conditions, good agricultural practices and high inputs of fertilizer and manure allow high yields of cereals, potato, sugar beet, forage grass and maize and of milk, that generally exceed average values for the EU27 (Table 3). Yield differences per hectare in Northwestern EU member states are largest for milk and ruminant meat because of large differences in shares of grazing beef and dairy cattle, areas of marginal grassland, grass in arable rotations (e.g., Denmark) and grazing intensity. Ireland, the UK and France hold large areas of less productive grassland on wet, peaty or mountain soils. All countries considered are net importers of substantial amounts of fodder and feed stuff, in the range of 200–400 kg LSU^{-1} in the period between 2000 and 2007 (FAOSTAT), with the exception of France (120 kg LSU^{-1}). These differences explain a minor part of differences in milk and ruminant meat yield per hectare.

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Mean national livestock densities in the considered member states range between 0.9 LSU ha^{-1} in Northern France, which is near to the average in the EU27, to 3.4 LSU ha^{-1} in The Netherlands (Table 4; using LSU definition according to Eurostat). The share of dairy cows (one dairy cow represents one Livestock Unit; LSU) ranges from 10 % in Denmark to 22 % in Ireland. Regional livestock densities can be much higher, with 8.9 LSU ha^{-1} in the southeastern part of The Netherlands, 6.0 LSU ha^{-1} in Flemish Region-Belgium and 3.7 LSU ha^{-1} in Brittany-France, and are always associated with the presence of a large pig and or poultry sector. Farm sizes per holding in the northwestern member states are much higher than the EU27 average.

Nitrogen from manures constitutes a substantial proportion of total nitrogen fertilization, ranging between 40 % in Germany and Northern France, to 60–65 % in Belgium, Ireland and The Netherlands. In The Netherlands and the Flemish Region the net nitrogen excretion (after subtracting ammonia emission from housing and storage) exceeds the application limit of 170 kg ha^{-1} set by the NiD, by 40 and 12 kg ha^{-1}, respectively, based on MITERRA results. These two countries require a combination of derogation, on the one hand, and export and processing of manure on the other hand, to be able to comply with the NiD at a national level. The sum of nitrogen excretion plus fertilizer use per hectare UAA in the period 2005–2008 ranges between 138 kg ha^{-1} in France to 377 kg ha^{-1} in The Netherlands (Table 5) and exceeds mean values for EU12 (old member states) and EU27.

3.2 Application standards for nitrogen from manure and fertilizer

The most important restriction following from the NiD is the application limit for nitrogen from animal manure. Other restrictions following from the NiD are mandatory minimum manure storage capacities, prohibition periods for nutrient application, restrictions for nutrient application near water courses, on slopes and on frozen, water logged or snow covered soils (Dijk and Berge, 2009; Table 6). These restrictions should facilitate the achievement of the overall objective of the NiD to establish a balance between nutrient application and crop requirements. There are large discrepancies between countries

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regarding the way these restrictions are translated into national law and applied in practice. For example methods of estimation of N emissions by livestock (including volatilization coefficients for ammonia), definitions of periods when and areas where manure application is restricted, procedures for enforcement of regulations can be very different and hamper a strict comparison of environmental impacts of the NiD between countries.

With the exception of France, all member states have negotiated with the EU Commission an extension of the application limit in the NiD of 170 $kg\,N\,ha^{-1}$ for manure from ruminants (a so-called derogation; Table 7). These derogations are based on proof that this extension will not increase the risk for exceeding the critical nitrate limit of 50 $mg\,NO_3^-\,l^{-1}$ in groundwater and surface water. Derogations are granted at farm level (except the Flemish Region) and mostly apply to farms with at least 70–80 % of farm land in use for grassland. The Flemish Region has a derogation at field level which depends on the cultivated crop, including for some arable crops. For grassland and forage maize followed by one cut of grass or cut rye the application limit is 250 $kg\,N\,ha^{-1}$ as cattle manure or treated pig manure and 200 $kg\,N\,ha^{-1}$ for beet and winter wheat followed by a catch crop (Table 7). Denmark has implemented a maximum application limit for arable land of 140 $kg\,ha^{-1}$ of nitrogen from pig manure and on organic farms (Kronvang et al., 2008), which is beyond the requirements of the NiD. The Netherlands has the largest derogation both regarding the extension of the application limit itself and regarding the area where this extension applies.

Only the NiD action programs of The Netherlands, Denmark and the Flemish Region have introduced crop and soil type dependent applications standards for total N inputs, from manures and mineral fertilizers (Dijk and Berge, 2009). Application standards in The Netherlands and Denmark apply to fertilizer equivalent (FE) N (Table 8). In Denmark, Ireland, The Netherlands and the UK standards are for some crops differentiated with actual yield level and target. For cereals different standards may apply to baking, malting and fodder qualities, for potato to cultivars for use as ware, french fry, starch and seed. In the Flemish Region farmers can choose between a fixed total nitrogen amount or FE N values for organic fertilizers per crop. This new system with some new limits has been introduced in 2011 (Anonymous, 2011). In Denmark, Ireland and the UK application standards also depend on the soil N status and cropping history.

Differences between total FE N application standards for the Flemish Region, The Netherlands and Denmark are quite considerable. The differences between the highest and the lowest values are in the order of about 20 % for forage maize and go up to about 75 % for potato (Table 8). As a whole, the standards are the highest in The Netherlands for most crops mentioned in Table 8. For grassland without clover, standards are highest in Denmark, however, grass with clover is predominant in Denmark, and has lower standards. Standards for winter wheat and, to a lesser extent, for forage maize in Denmark and the Flemish Region are comparable. On the other hand, the standards for potato and sugar beet are lower for Denmark compared to the Flemish Region while this is the reverse for grassland. One would expect application standards in Denmark to be lower than in the Flemish Region in view of a lower yield potential (Table 3) and taking into account that in Denmark the fertilization limits are set at 90 % of the optimum N-fertilization.

The consequence of having a legal system of application standards based on total FE nitrogen is the introduction of fixed statutory values for the fertilizer equivalency of manures. This is the case in the NiD action programs of Denmark, the Flemish Region, Germany and The Netherlands. Statutory FE values may provide an incentive to farmers to raise their actual FE values (low with poor management) up to at least the statutory value. A legal system based on FE is more comparable to the system for N recommendation than a system based in total N, and provides the farmer direct insight into whether he needs to improve his N management to meet the recommended N requirement. The statutory FE values do not always correspond to those used in fertilizer recommendations (Berge and Dijk, 2009). Low fertilizer equivalencies for manure are caused by gaseous losses of ammonia, N oxides and di-nitrogen, and the slow release of organic manure N. For slurry FE's range from about 20 % in the UK to 75 % in Denmark. The small values quoted for the UK assume the manures are not applied

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using reduced ammonia emission techniques. For solid poultry manure FE's range from 30 % in the UK, the Flemish Region and Germany to 55–65 % in Denmark and The Netherlands (Webb et al., 2012; Table 9). In France FE reference values vary with crops (spring versus winter) and application period (COMIFER, 2012). In Ireland maximum fertilizer equivalencies for manure of 40 % have been reported (Hoekstra et al., 2011). Equivalencies can be increased by using low emission manure application techniques and by improved management of manure and soil (Dalgaard et al., 2011), such as replacing autumn application of manure by spring application. Increasing legal FE provides a strong incentive to apply these techniques and to improve management of manure.

In Germany there are no maximum N application limits for total or FE nitrogen. Instead, there is a restriction on net N surplus at farm level. The farmers have the responsibility to plan fertilization in such a way that the three year average of the N surplus does not exceed 60 $\mathrm{kg\,N\,ha^{-1}}$ from 2009 onwards. This surplus constraint has been introduced stepwise since 2006 (Wolter et al., 2011).

France does not prescribe application standards in its action program for NVZ's. Total N inputs are limited only in areas where nitrate concentrations in ground or surface water are high and where that water is used for drinking water. This limit is 210 $\mathrm{kg\,N\,ha^{-1}}$ in parts of Brittany, while in some watersheds with nitrate in surface water exceeding 50 $\mathrm{mg\,l^{-1}}$ total N inputs are restricted to values as low as 140 $\mathrm{kg\,N\,ha^{-1}}$ (Dijk and Berge, 2009). Restrictions for use of fertilizers, and other agrochemicals like pesticides, in drinking water abstraction areas are common in Europe, also before the introduction of the NiD.

3.3 Nitrogen balance

Complete official reports to the EU of the effect of the national action plans for the NiD are available for the 3rd (1999–2003) and 4th (2004–2008) reporting period and summarized by the European Commission (2011). A high gross nitrogen balance (GNB) is always associated with high gross inputs of manure (Table 5). In all countries

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considered, the GNB decreased between 2000 and 2008 (Fig. 1). The decrease of GNB between 2000 and 2004 is larger than between 2004 and 2008. The decrease in The Netherlands was 80 kg ha^{-1} and largest, but the GNB in 2008 is still higher than for other countries. The relative decreases of the GNB between 2000 and 2008 in Belgium (31 %), Ireland (25 %) and the UK (23 %) are comparable to the decrease in The Netherlands (30 %). The major cause for a decrease of the GNB is the decrease of the use of chemical fertilizer. In Denmark and The Netherlands this decrease was instigated to a large extent by increased utilization of manure N (Mikkelsen et al., 2010; Dalgaard et al., 2012).

Nitrogen balance calculations using MITERRA provide insight in soil inputs and outputs underlying the differences in the N balance (Table 10). MITERRA results for N removal (R^2 = 0.92), GNB (R^2 = 0.94) and even more so SNB (R^2 = 0.96) are significantly correlated with total N input from manure and fertilizer but results for individual countries may deviate from the average relation. This is the case for Ireland in view of dominant grazing sector. In The Netherlands and the Flemish Region the difference between total N excretion and actual manure application is larger than for other countries because of substantial net export and processing of manure from pigs and poultry, amounting to 18 kg N ha^{-1} and 54 kg N ha^{-1} in 2008, respectively. Flemish pig manure is mostly processed by waste water treatment where N is removed by denitrification. In The Netherlands the five provinces with an intensive pig and poultry sector export on average 127 kg N ha^{-1} to the other seven provinces and a small part (10–20 kg N ha^{-1}) abroad, mainly to Germany.

Comparing nitrogen surpluses at national level for the Northwestern EU member states is not very informative because of large differences in agricultural structure and livestock intensity within these countries (Table 4). Therefore, nitrogen use and balance by MITERRA model at NUTS 2 level. were recombined to generate results for regions with similar UAA (Fig. 2). Eleven regions had an SNB exceeding 100 kg N ha^{-1}. In addition to The Netherlands, Belgium, Brittany in France is standing out while several regions in the UK and single regions in Germany, Ireland and France

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have an SNB modestly exceeding 100 kg N ha^{-1}. Zooming further into MITERRA results for The Netherlands and Belgium, we find greatest surpluses for 2008 in the Province of Antwerp (241 kg N ha^{-1}), and the Southeast of The Netherlands (mean value 191 kg N ha^{-1} and maximum value of 197 kg N ha^{-1} in the province of Noord Brabant). These regions with the greatest N surplus are also most sensitive to nitrate leaching with MITERRA leaching fractions of 18 % in Brittany, 22 % in the Flemish Region (26 % in Province of Antwerp), 24 % in Southeast of The Netherlands (33 % in the province of Noord Brabant). GNB by MITERRA for the seven considered countries in 2008 is on average 19 kg ha^{-1} higher than GNB in Eurostat and fairly well correlated ($R^2 = 0.74$). Major outliers are Belgium and Ireland with differences of 38 and 58 kg ha^{-1}, respectively, the possible causes of which will be addressed in the discussion.

3.4 Water quality

In view of different monitoring procedures and differences in hydrology, geology and soils in the considered member states, reports to the EU Commission of nitrate concentrations in groundwater exceeding a policy target (in this case the nitrate limit for drinking water) do not provide direct insight in the effectiveness of NiD action programs or in the impact of differences of nitrogen balances. This is perhaps most strikingly illustrated in The Netherlands where mean nitrate concentrations in groundwater are low (Fig. 3) while the GNB is highest (Figs. 1 and 2). In part differences between reporting periods and countries are artifacts of different monitoring procedures and data selections. For example the apparent increase of nitrate concentrations in Denmark and The Netherlands between 2000–2003 and 2004–2007 in the EU dataset (European Commission, 2011) is an artifact of inclusion of observations in the uppermost groundwater in the 2004–2007 EU dataset. But differences also have hydrogeochemical causes like the presence of relatively deep soils, high groundwater tables and high organic matter contents (in part as peaty soils) promoting denitrification. Some areas in the UK

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have deep unsaturated extents through which the travel time for nitrate may be several decades (Wang, 2012). Analysis of lag times required for improvements of groundwater nitrate levels in Ireland showed that the achievement of good water quality status for some water bodies may be too optimistic but improvements are predicted within subsequent 6- and 12-yr cycles (Fenton et al., 2011). Analyzing a 50 yr time series of SNB and nitrate concentration in groundwater in Denmark, Hansen et al. (2011) found that nitrate concentrations are decreasing since 1980. They found that the frequency of downward nitrate trends in groundwater samples clearly increased with lower recharge age, providing proof that younger groundwater responds fastest to decreasing trends of SNB. Hansen et al. (2012) further found that nitrate concentration decreased significantly more in areas with a high livestock density. Reported nitrate concentrations in Germany are higher than in the other Northwestern EU member states because sampling is restricted to agricultural soils and focused on polluted regions. Changes in monitoring procedures and densities do not allow solid conclusions on nitrate trends between the 3rd and 2nd monitoring period. The overall picture is that of stable nitrate concentrations in the total population of groundwater observations. In shallow groundwater, which responds most directly to NiD action programs, 60 % of all samples in the EU27 were below 25 mg NO_3 l^{-1}, and 20 % above the NiD target of 50 mg NO_3 l^{-1} (European Commission, 2011).

When selecting data for shallow phreatic groundwater from official national NiD reports for The Netherlands (Zwart et al., 2008), the Flemish Region (Desimpelaere et al., 2008), Walloon region, Ireland, Germany and Denmark (Anonymous, 2008b–e), differences between countries (Fig. 4). appear to be more in accordance with differences of the nitrogen balance (Fig. 1).

In countries with a long running monitoring network for nitrate in the upper, sometimes shallow, groundwater in sandy phreatic aquifers (Fig. 5) a slow to moderate decrease of nitrate concentration can be observed. The mean decrease of the nitrate concentration in the monitoring period is largest in The Netherlands (6 mg NO_3 l^{-1} yr^{-1}), followed by Denmark (2 mg NO_3 l^{-1} yr^{-1}), Germany (0.6 mg NO_3 l^{-1} yr^{-1}), Flemish

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Region ($0.7\,mg\,NO_3\,l^{-1}\,yr^{-1}$) and finally the Walloon region with a small increase ($0.3\,mg\,NO_3\,l^{-1}\,yr^{-1}$). These trends do not only reflect the effect of the measures from implementation of the NiD, but also on changes in agricultural practices and effects of implementation of other policies, e.g., measures for reducing ammonia emission. Trends further depend on sampling depth and travel time of infiltrating water which differ spatially within countries and between countries.

Observed nitrate exceedance in the period 2004–2007 (Fig. 4) and nitrate concentration concentrations between 2005 and 2010 (Fig. 5), both in upper levels of phreatic groundwater, agree fairly well with modeled nitrate concentrations in leaching water in 2008 using MITERRA (Figs. 6 and 7). Some level of disagreement is to be expected considering that nitrate concentrations in leaching water will to tend be higher than in groundwater, and that monitoring data are not always representative for nitrate concentration in total UAA. In Germany, observed concentrations are higher than MITERRA results in view of the intended focus of the monitoring program on areas with high nitrate concentrations (Anonymous, 2008d).

MITERRA results for NUTS 2 regions with mean area weighted nitrate concentrations exceeding $50\,mg\,NO_3\,l^{-1}$ are found only in The Netherlands, the Flemish Region, the western part of Germany and in Brittany (Fig. 7). SNB values exceeding $100\,kg\,N\,ha^{-1}$ in regions in the UK and Ireland (Fig. 2) do not lead to exceedance of the nitrate target of the NiD as a result of relatively low nitrate leaching fractions in these regions. However, the risk of exceedance of ecological limits for nitrate or nitrogen in surface water will be higher in regions with high SNB.

The EU Water Framework Directive gives room to member states to define and differentiate national standards for good ecological status or potential. A nitrate limit concentration of $10\,mg\,NO_3\,l^{-1}$ ($2\,mg\,N\,l^{-1}$) was used as a proxy for the nitrate limit in fresh waters (Cardoso et al., 2001). Surface waters with mean nitrate concentration greater than $10\,mg\,NO_3\,l^{-1}$ ranged from 20 % in Ireland to 60 % in Germany (Fig. 8). Between 2000 and 2007 the percentage of surface water samples exceeding $10\,mg\,NO_3\,l^{-1}$ shows a small decrease, when looking to the total population of fresh surface water samples

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reported to the EU Commission (Fig. 8). Differences between countries do not seem to have a clear relation with observed exceedance in groundwater. Again, in part these differences reflect different response mechanisms and response times and nitrate attenuation during transport from groundwater to surface water (Fenton et al., 2010). However, differences in response time will be less than for deeper groundwater bodies. In particular response of surface water nitrate to restrictions on how and when to apply manure and fertilizer (Table 6) should be faster, due to the shorter transport pathways compared to deeper aquifers, while full response to restrictions on application levels may take decades.

4 Discussion

4.1 Application standards

The theoretical or empirical basis of differences between nitrogen application standards in national regulations for NiD implementation in northwest European countries is not always clear (Table 8). Differences between standards to a large extent derive from differences in fertilizer recommendation in the northwestern members states (Table 11). One may expect more comparable fertilizer recommendations in view of the similar yield potentials. However, it is difficult to compare fertilizer recommendations as different countries apply different systems (Berge and Dijk, 2009). The Flemish Region, Denmark and The Netherlands use systems based on dose-effect trials, while Germany and France use a balance approach. All countries use calculation schemes to correct N recommendations for yield level and N deliveries from soil, and cropping history and manure application. These schemes are not standard, and may depend on the local advisors, which leads to significant variability in the recommendations. In general nitrogen application standards in NiD action programs for Denmark and The Netherlands tend to be lower than the N-fertilizer recommendation. With the new introduced standards, this is partly also the case for the Flemish Region.

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The overall effects of these differences are difficult to judge as standards are implemented at farm level and arable crops are cultivated in rotations. Denmark has far less permanent grassland than The Netherlands and grassland contains more clover while temporary grassland is part of the crop rotation. Rotations may to some extent level out environmental effects of differences between standards. Comparison of recommendation systems could perhaps be improved by applying country or regionally specific system to derive N recommendation for more narrowly defined combinations of crop, rotation, soils and weather characteristics.

4.2 Nitrogen balance

There are considerable differences between estimates of GNB in EUROSTAT, by MITERRA and in national reports (Table 12). Precise comparison of results for GNB was difficult because results were not always available for the same years and because data underlying GNB for a specific year are regularly modified. GNB for 2008 calculated by MITERRA is on average 19 $kg\,N\,ha^{-1}$ higher than reported to the EU Commission (EUROSTAT) and to a lesser extent than reported by the OECD (Velthof et al., 2009). Differences are most marked for Belgium and Ireland. N removal and N excretion (not shown) are the major source of difference between GNB estimates. National use of chemical fertilizer in general is fairly accurate, but values for specific years in national reports, e.g. Belgium, show quite some variation, and in part reflect the absence of reliable registration systems for fertilizer purchase. Different estimates of UAA play a minor role.

On average, estimates of N removal in MITERRA for the seven member states are 22 $kg\,N\,ha^{-1}$ lower than estimates for EUROSTAT and could fully account for the mean difference of GNB (Table 12). Estimates in national reports for some countries tend to be somewhat higher than values reported to EUROSTAT, but this in part may be due to comparing different periods. The uncertainty of N removal in crops is further illustrated by results from Leip et al. (2008), that were on average nearly 28 $kg\,N\,ha^{-1}$ higher than in EUROSTAT, using a more deterministic European model approach.

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N removal from grassland for fodder likely is the major source of difference in estimates of total N removal (Velthof et al., 2009).

For the Flemish Region, Lenders et al. (2012) estimate N removal at about 320 kg N ha^{-1} based on grassland yields of 10.5 t ha^{-1} for permanent grassland and 11.5 t ha^{-1} for temporary grassland, and a N content of 3 %. MITERRA estimates N removal from permanent grassland at about 220 kg N ha^{-1}. Differences are caused by lower estimates of effective dry matter yield for mixed system of grazing and cutting, and of lower N contents. Estimates of mean N removal from grassland in The Netherlands, with practices and N intensity comparable to that in the Flemish Region, are around 260 kg N ha^{-1}. So overestimation of N removal from grassland (36 % of UAA) could explain a major part of the difference between GNB estimates by MITERRA and national reports.

GNB in 2008 by MITERRA for Brittany in France is more than twice the regional estimate for 2006 (Agreste, 2009). Again this can be largely (> 50 %) explained by a much higher regional estimate of N-removal, and to lesser extent by lower estimates of manure input (about 20 %) and chemical fertilizer (about 10 %). Regional data would suggest an overall nitrogen use efficiency (N-removal over total N input from fertilizer and manures) of 80 %, which does not seem realistic. Nitrogen use efficiency in Brittany by MITERRA is about 40 %, as compared to 60 % for EU27.

For Ireland, total N removal in MITERRA in 2008 is 23 kg N ha^{-1} lower than the average N-removal between 2005 and 2008 in EUROSTAT and national reports. In Ireland 3.9×10^6 ha of UAA (95 %) is grassland. Mean N-removal on grassland is estimated for EUROSTAT at 155 kg N ha^{-1}, while MITERRA calculates about 130 kg N ha^{-1}. Part of this difference may be due to different assumptions on reduction of yields and N removal for grazing as compared to cutting, and to different assumptions on shares of intensively and extensively managed grassland. Differences in N removal per hectare between intensive and extensive grassland can amount to a factor of two (Velthof et al., 2009). Another major source of discrepancy for Ireland between MITERRA results and national reporting is a higher gross input of N in manure. In Ireland almost 90 % of N

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production in manure is from cattle. Irish national reports use an N-excretion value of 85 kg N per dairy cow (Anonymous, 2010), while MITERRA uses a value of 105 kg N per dairy cow (Velthof et al., 2011, Annex 1). The high value is based on a more dynamic approach accounting for regional differences in milk yields, grassland yields, and N inputs, while the low value is mainly a function of milk yield. Estimates of N removal for fodder and N excretion are related, as fodder is the major N input and manure N is the major output. For Ireland N-removal in EUROSTAT (and national reports) is more than 30 % higher than N excretion. Even when taking into account N removal in milk and meat and N imports of feed concentrates, the large difference between N removal and N excretion may be an indication that either N removal is overestimated or N excretion is underestimated. On the other hand excretion estimates by MITERRA do not seem to match with a relatively modest average milk yield in Ireland around 5000 kg per cow per year.

Germany is the only country that has established targets for the surplus of N (90 kg ha^{-1} for 2006–2008) and phosphate (20 kg ha^{-1} in a six-year-average); and managed to achieve these targets in 2008. The stricter targets of 60 kg N ha^{-1} as a three-year-average from 2009–2011 onwards may also be achieved, but some intensive livestock farms and other farms with higher N surplus still have to increase their N-efficiency. Infringements of these restrictions are not directly subject to fines, but will lead to administrative procedures with increasing obligations for farmers to adapt to the maximum surplus levels.

Recent national census data indicate that since 2008 the use of chemical fertilizer in Denmark, Germany and The Netherlands is still decreasing, and along with that, probably also the soil surplus of nitrogen. The decrease of the purchase of chemical N fertilizer coincides with the increase in fertilizer price since 2008 (Fig. 9). This price increase is not compensated by an increase of prices of agricultural commodities. In Germany the ratio of the price indices of nitrogen fertilizer and wheat increased more than twofold between 1990 and 2011. In view of the high fertilizer prices farmers may tend to reduce or postpone fertilizer purchases. The latter hypothesis is supported by

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an increase of purchase of chemical N fertilizer in Germany and The Netherlands in 2011. On the other hand, in Denmark purchase of N fertilizer was hardly affected. So changes of nitrogen use and surpluses since 2008 in part can be price effects which interfere with effects of the NiD.

4.3 Implications for the NiD

Monitoring and evaluation of the implementation and effects of NiD is crucial for its success. At a national level it is a requirement to maintain support from farmers and their local advisors, as the main actors involved, and for national governments to optimize policies. The main activities for monitoring and evaluation are registrations of farm resources and activities (fertilizer, livestock, UAA), monitoring of water quality and using calculation procedures and models to assess environmental loads and relate this to farm measures and water quality. These evaluation activities take place at the national level, with varying levels of detail and sophistication, and in a more harmonized and generalized manner at the European level. For the latter, the European Commission uses institutes like the European Environment Agency (EEA) and the Joint Research Centers (JRC) and has initiated various service contracts, to improve datasets of agricultural activities, and develop and apply models to relate activities to N emissions and water quality (RAINS, GAINS, CAPRI, MITERRA). In spite of recent progress it is difficult to judge to what extent national implementation and evaluation of the NiD benefits from joint activities and what are major caveats in data and knowledge about the effects and effectiveness of the NiD.

A typical conclusion from national evaluations is that the NiD has made a major contribution to reduction of the N surplus. Evaluation of the Danish Aquatic Plan II concluded that between 1998 and 2004 the reduction of N-application standards contributed 13×10^6 kg (32 %) to the total reduction of the soil N-surplus (SSNB) of 80×10^6 kg, while increasing legal FE for N in manure contributed 10×10^6 kg (26 %) and reduced N in feeding 4×10^6 kg (10 %) (Mikkelsen et al., 2010). Evaluation of the Dutch second action program concluded that between 1998 and 2004 the Mineral Accounting

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System (MINAS) led to an overall reduction of the net SSNB by 78×10^6 kg N (Grinsven et al., 2005). Here the combination of reducing N-loss standards, and more efficient N management by better insight from keeping mineral accounts at farm level, contributed about 100×10^6 kg (67 %), while reduced N in feeding contributed 14×10^6 kg (19 %) and reducing livestock and increasing manure export 11×10^6 kg (14 %). In The Netherlands the dairy sector contributed most to reduction of the use of chemical fertilizer, and this reduction was both a learning effect of applying mineral accountancy at farm level and of enforcement of N loss standards.

In spite of various efforts at the European level to harmonize procedures for monitoring and evaluation of the NiD, differences in implementation and insight into the effectiveness still vary considerably. A major source of difference among member states is the way nitrogen recommendations, and application standards for total FE nitrogen, account for nitrogen in manure. Nitrogen emissions from agricultural sources, particularly manures, are a major source of environmental pollution and welfare loss (Sutton et al., 2011). A logical next step for improving harmonization and effectiveness of the NiD is to demand stricter accounting of nitrogen in manures, e.g. by imposing a compulsory time path and increasing nitrogen fertilizer equivalencies for different types of manures in application limits (Csathó and Radimszky, 2009). However, such steps require knowledge sharing, e.g. in defining codes of Good Agricultural Practice and adopting techniques to improve nitrogen efficiency in manures. Without that, a too fast and too strict regulation of nitrogen in manures may decrease the willingness of arable farmers to accept manure from livestock farmers, because of fear of insufficient N supply. In the future increasing prices of nitrogen fertilizer may provide an additional economic incentive to reduce the use of chemical fertilizer and to increase the efficiency of manures.

The NiD and the national implementation of restrictions on where, when and how much nitrogen in fertilizer and manure can be applied to agricultural land, will remain a major instrument to reduce nitrogen pollution in waters. However, we should also recognize that agricultural sources of nitrate are only part of the nitrogen burden. On average in the EU non point agricultural sources contribute 65 % to the N load to fresh

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water, with national values ranging from about 50 % in Germany to 75 % in The Netherlands (inferred from Leip et al., 2011). So even when all the measures under NiD have taken hold it is unlikely that nitrate concentrations in surface water, and to a lesser extent in groundwater, will return to pre-industrial levels (Howden et al., 2011). For the immediate future the importance of the NiD for protecting drinking water may be best seen in those areas with private or small public drinking water facilities, using groundwater from shallow aquifers, as is the case in Denmark (Grinsven et al., 2010). In order to protect their coastal waters member states in deltas or estuaries of large cross boundary rivers, like The Netherlands and Romania, depend on the NiD, particularly when national implementation of the Water Framework Directive is limited to reducing non-agricultural sources of N. A problem when implementing the NiD for this purpose is that the limit value of 50 $mg\,l^{-1}$ does not apply to fresh waters and coastal waters (Nimmo Smith et al., 2007). Nonetheless, the NiD requires Member States to protect such bodies at risk of eutrophication. The lack of a single standard along with the range of influences that bear on eutrophication can cause some confusion. For control of coastal eutrophication, e.g. in Brittany, a limit value around 5–10 $mg\,NO_3\,l^{-1}$ would be more appropriate.

5 Conclusions

The most significant effect of the implementation of the NiD since 1995 in the northwest of the EU is a major contribution to the decrease of the nitrogen soil N balance and by that of the gross N load to the aquatic environment. This effect of the NiD has not yet manifested in a convincing decrease of nitrate concentrations in EU monitoring in groundwater and fresh surface waters since 2000. However before 2000, introduction of Good Agricultural Practices for fertilization has decreased median and extreme nitrate concentration in many surface water systems in e.g. The Netherlands, Demark and the Flemish Region. Only countries that operate long running monitoring programs in

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shallow groundwater in agricultural areas, viz. Denmark, the Flemish Region and The Netherlands, can detect a convincing decrease of nitrate concentrations.

Without good opportunities to evaluate the effectiveness of NiD, it is difficult for the EU community to improve the NiD and implementation in member states may lose momentum. This benchmark study indicates that differences in calculation and data procedures between member states in Northwestern EU for determining the nitrogen balances are such that comparison of effects of NiD on the N balance between countries is not yet possible. In particular the calculations methods for N excretion and N removal vary considerably among countries. Regarding compliance with application limit for N in manure also the definition of farm area differs between countries ranging from total farm area to the area where manure actually is applied. Improved guidelines and procedures for monitoring water quality, registration of fertilizer use and harmonization of fertilizer recommendation systems are needed. Better selections and availability of the collective monitoring results in EU synthesis reports and data facilities can help to increase the efficiency of our monitoring effort to evaluate the NiD.

Implementation of the NiD in member states in the northwest of the EU is fairly comparable regarding restrictions for application of fertilizer and manure, but very different regarding application standards for total N fertilization. Nitrogen application standards in national implementations of the NiD are closely linked to national nitrogen fertilizer recommendations. However, differences in national systems for nitrogen recommendations are substantial and resulting recommendations for specific combination of crops and soils and do not bear a clear relationship with differences in yield per hectare.

At some point in the future, when the first and relatively easy environmental improvements by the present implementations of NiD are achieved, the NiD may need to be adjusted. Perhaps through more specific regulation of nitrogen in manure and differentiation of water quality limits. However, there is an immediate need to improve our data procedures to allow evaluation and benchmarking of adequacy and effectiveness of NiD implementation.

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Table 1. Density of groundwater and surface water sampling for the whole land surface in monitoring programs for the NiD (European Commission, 2011).

	density of groundwater sampling stations (points/1000 km^2)	density of surface water sampling stations (points/1000 km^2)
Belgium	99	38
Germany	3	1
Denmark	34	5
France	5	3
Ireland	1	3
The Netherlands	33	13
UK	13	33

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Table 2. Precipitation surplus and fraction of nitrogen surplus leaching to groundwater and small surface waters, the fraction leaching to large surface waters and the runoff fraction of N in applied fertilizer and manure, used in the MITERRA model.

	Precipitation surplus (mm)	Groundwater + small surface leaching fraction (%)	Large surface water leaching fraction (%)	Runoff fraction (%)
Belgium-Flemish	396	23	9	3
Belgium-Walloon	479	11	12	4
Denmark	280	24	6	2
Northern France	356	13	10	5
Germany	295	13	10	4
Ireland	554	10	8	3
The Netherlands	420	17	7	3
UK	450	11	10	3

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Table 3. Mean yields in northwestern member states of the EU for cereals, forage maize, potato and sugar beet (Sources: FAOSTAT mean crop data are for the period 2000–2007; EFMA, 2008, mean data for 2006–2009), and the sum of ruminant meat +0.1× total milk production as a proxy for ruminant productivity per hectare of permanent grassland (Sources: production from FAOSTAT, data 2008, and grassland areas from Eurostat, 2011, data 2007).

	FAO	FAO 2000–	FAO 2007	FAO	FAO 2008	EFMA 2006–	EFMA 2009	EFMA
	Wheat	Forage maize	Potato	Sugar beet	0.1× Meat + Milk	All cereals	Potato	Sugar beet
	$t ha^{-1}$	$t ha^{-1}$	$t ha^{-1}$	$t ha^{-1}$	$t ha^{-1}$ grassland	$t ha^{-1}$	$t ha^{-1}$	$t ha^{-1}$
Belgium	8.2	11.1	43.4	67.9	1.09	8.8	46.0	65.0
Denmark	7.1		39.5	57.3	1.67	5.9	44.7	55.7
France	6.9	8.6	41.4	76.5	0.50	7.2	45.7	82.5
Germany	7.3	8.8	40.9	59.1	0.85	6.5	40.1	58.0
Ireland	8.9		35.2	48.6	0.36	7.0	32.8	
The Netherlands	8.2	11.2	43.5	61.6	1.85	8.2	46.3	63.2
UK	7.7		41.6	54.7	0.25	7.1	41.6	61.7
EU27					0.43	5.0	29.0	62.1

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Table 4. Main characteristics of agricultural sector in northwestern member states of the EU in 2007 (Eurostat, 2011).

	Agricultural area (UAA) 10^6 ha	Livestock density LSU ha^{-1}*	Permanent Pasture % of UAA	Farm size ha UAA/ holding
Belgium	1.4	2.8	37	29
Denmark	2.7	1.7	8	60
France	27.5	0.8	29	53
North-Central**	17.8	0.9	21	–
Germany	16.9	1.1	29	46
Ireland	4.1	1.4	76	32
The Netherlands	1.9	3.4	43	26
UK	16.1	0.9	62	65
EU27	172.5	0.8	33	13

* In the EUROSTAT definition one LSU corresponds to the feed requirement of one adult dairy cow producing 3000 kg of milk annually.
** All departments above the line "Nantes-Dijon".

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Table 5. Average inputs, crop removal and gross balance of nitrogen in 2005–2008 in north-western member states of the EU (Eurostat, 2012).

	Inorganic fertilizer	Gross manure	Other inputs $kg\,N\,ha^{-1}$	Removal	Gross N balance
Belgium	101	168	41	191	119
Denmark	75	100	24	101	98
France	76	62	26	112	52
Germany	103	74	42	125	93
Ireland	78	117	15	155	55
The Netherlands	140	236	28	194	210
UK	94	87	31	111	101
EU-15	67	63	26	98	58
EU-27	61	54	25	89	50

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Table 6. Restrictions for application of fertilizer and manure in national implementations of the Nitrates Directive (adapted from Dijk and Berge, 2009).

	DK	BFL	F	GE [1]	UK	NL	IRL
Farm measures							
fertiliser planning							
keeping records	yes	yes	yes	yes	yes	yes	yes
soil analysis	yes	yes		yes		yes [2]	
fertilisation							
closed periods for manure/fertilisers [3]	yes	yes	yes [4]	yes	yes	yes	yes
low emission application	yes	yes				yes	
no manure application on frozen, snow covered and waterlogged land	yes	yes	yes [4]	yes	yes	yes	yes
unfertilised zones along surface water [6]	yes [7]	yes	yes [4]	yes	yes	yes	yes [8]
post-harvest measures							
catch crops	yes		yes [4]			yes	
no tillage in autumn	yes						yes [5]
Other Policy Measures							
Max limit for livestock	yes						
Maximum limits on N and P use							
manure	yes	yes	yes	yes	yes	yes	yes
total N (manure+fertilisers)	yes	yes	yes [4]		yes	yes	yes
Maximum N and P surpluses				yes			
Maximum soil mineral N autumn		yes	yes [9]	yes [1]			

DK = Denmark, BFL = Belgium Flemish Region, F = France, GE = Germany, UK = United Kingdom, NL = The Netherlands, IRL = Ireland

[1] Implementation varies between states (Länder) of Germany, e.g. maximum soil mineral N autumn only in Baden Wurttemberg.

[2] In case farm has derogation.

[3] For liquid manures generally between September/October and February.

[4] In some departments within the NVZ's. E.g. catch crops in western regions (Brittany and Normandy); Anonymous (2008a).

[5] Ploughing between July and November if green cover emergence of planted crop within 6 weeks of ploughing.

[6] With large variation in width and length of unfertilized zones.

[7] Increased from 2 m to 10 m from 2012 onwards.

[8] No fertiliser within 2 m of a surface water.

[9] In small highly sensitive areas (e.g. coastal areas with green tides).

Table 7. Overview of area in Nitrate Vulnerable Zones and derogations for grassland (mostly dairy) farms in 2009 (European Commission, 2011).

	Nitrate Vulnerable Zones area (%)	Application limit for manure ($kg\,N\,ha^{-1}$)	Share of agricultural land (%)	Share of farms (%)
Belgium	68			
Flemish Region	100	250/200 [1]	12	10
Walloon Region	42 [2]			
Denmark	100	230	4	3.2
France	45	170	0	0
Germany	100	230	< 1	< 1
Ireland	100	250	8	8
The Netherlands	100	250	45	32
UK	39	250	1.5	1.3

[1] Also a derogation for some arable crops.
[2] Situation in 2007 (Anonymous, 2008b).

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Table 8. Nitrogen application standards for some major crops in the 4th action programs for the NiD expressed either as fertilizer equivalent N (FE) or total N.

		soil	Grass: graze and cut	Forage maize	Winter wheat	Potato ware	Sugar beet
The Netherlands	FE	sand	260	150	160	245	145
	FE	clay	310	185	220	250	150
Denmark [1,2]	FE	sand	310 [5]	150	150 [3]	140	110
	FE	clay	330 [5]	155	180 [4]	140	120
Flemish Region	FE [8]	sand	235	135	160	190	135
	FE [8]	clay	245	150	175	210	150
	total	sand	350	205	200	260	205
	total	clay	360	220	215	280	220
UK	total	all	330	150	220	270	120
Ireland [6]	total	all	306 [7]	140	180	145	155

[1] 0–5 % clay, not irrigated.
[2] > 15 clay, not irrigated.
[3] Fodder quality.
[4] Baking quality.
[5] For grass with clover 62–227 kg N ha^{-1}, depending on % clover.
[6] Soil nitrogen index 2 for arable crops.
[7] For stocking rate between 170 and 210 kg ha^{-1} N per year.
[8] Valid from 2011 and without catch crop.

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Table 9. Nitrogen fertilizer equivalency (%) for application of most common manure types (after deduction of gaseous losses from buildings and storage; taken from Webb et al., 2012).

	Cattle slurry	Pig slurry	Layer solid manure	Broiler solid manure
The Netherlands	60	60–70	55	55
Flemish Region	60	60	30	30
Denmark	70	75	65	65
France	50–60	50–75	45–65	45–65
Germany	50	60	30	30
UK	20/35	25/50	20/35	20/30
Ireland	40	50	50	50

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Table 10. N inputs, removal an soil N balance in 2008 in northwestern member states of the EU according to MITERRA ranked with SNB.

	UAA	total N excretion	applied manure	grazing	applied fertilizer	Total N soil input	N removal	SNB
	10^6 ha				kg N ha^{-1}			
The Netherlands	1.9	264	140	67	110	356	179	176
Belgium	1.3	187	76	54	107	272	149	124
Flemish R.	0.7	281	109	63	107	314	166	147
Walloon R.	0.7	114	51	47	107	240	135	105
Ireland	4.1	138	46	81	81	228	132	94
North. France	17.8	65	29	24	75	154	87	66
UK	14.3	70	23	35	64	143	72	66
Denmark	2.5	95	67	11	69	170	106	65
Germany	16.7	79	49	13	93	186	122	64
France	30.1	57	24	23	67	137	80	56
EU27	172.5	57	27	19	61	127	67	59

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Table 11. Ranges of N recommendations in different regions for sandy to loamy soils with no effect of previous crop and a medium level of soil nitrogen supply (SNS). Relatively high N-recommendations are found in The Netherlands and Denmark, relatively low values in France and the UK (sources: Dijk and Berge, 2009; for FL Bodemkundige Dienst van België, 2012; for UK DEFRA, 2010; for IRL Coulter and Lalor, 2008).

	NL	DK	FL	GE	FR	UK	IRL[1]
				$kg\,N\,ha^{-1}$			
Grass	285–385	365–405	250–300	200–300	185–285	180–340	40–306[2]
Fodder maize	150–175	160–190	150–175	150–160	110	50	110–180
Winter wheat	190–230	180–210	150–190	130–220	170	70–120	120–210[3]
Potato ware	245–250	155–180	200–225	70–140	120	60–160	120–170
Sugar beet	150	125–150	130–160	90–150	120	80	120–195

[1] Rates shown for non-grassland correspond to a soil N Index range of 1 to 3.
[2] Rates of N application on grassland vary depending on stocking rate and usage for grazing and/or cutting.
[3] Assuming $9\,t\,ha^{-1}$ yield of winter wheat (additional N is recommended for higher yields).

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Table 12. N removal, and gross N balance (GNB) by MITERRA in 2008, compared to values in Eurostat and national reports in the period 2004–2009.

		MITERRA 2008		EUROSTAT 2005–2008		National 2004–2009	
	UAA 10^6 ha	removal	GNB	removal kg N ha^{-1}	GNB	removal	GNB
EU27	172.5	67	70				
Belgium	1.4	149	156	191	118	191 [1]	117 [1]
Flemish R	0.7	166	200			213–223 [2]	57 [2]
						220 [1]	63 [1]
Walloon R	0.7	135	122			163 [1]	57 [1]
Denmark	2.5	106	82	101	93	115 [3]	79 [3]
France	30.1	80	67	112	49	120 [4]	50 [4]
North France	17.8	87	79				
Brittany	1.6	89	215			157	91 [5]
Germany	16.7	122	81	125	92	131 [6]	91 [6]
Ireland	4.1	132	108	155	50	155	53
The Netherlands	1.9	179	213	194	188	209 [7]	178 [7]
UK	14.3	72	84	111	93	137 [8]	91 [8]

[1] Gybels et al. (2009), for period 2004–2006.
[2] Lenders et al. (2012), for period 2007–2009.
[3] Grant et al. (2010), period 2006–2008.
[4] Anonymous (2008a), period 2004–2006; GNB inferred from SNB using gaseous N loss by MITERRA.
[5] Agreste Bretagne (2009); for 2006. SNB value converted to GNB using gaseous N loss by MITERRA (48 kg N ha^{-1}).
[6] Anonymous (2008c), period 2004–2006.
[7] CBS statline, http://statline.cbs.nl, downloaded January 2012.
[8] Fernal and Murray (2009), period 2005–2007.

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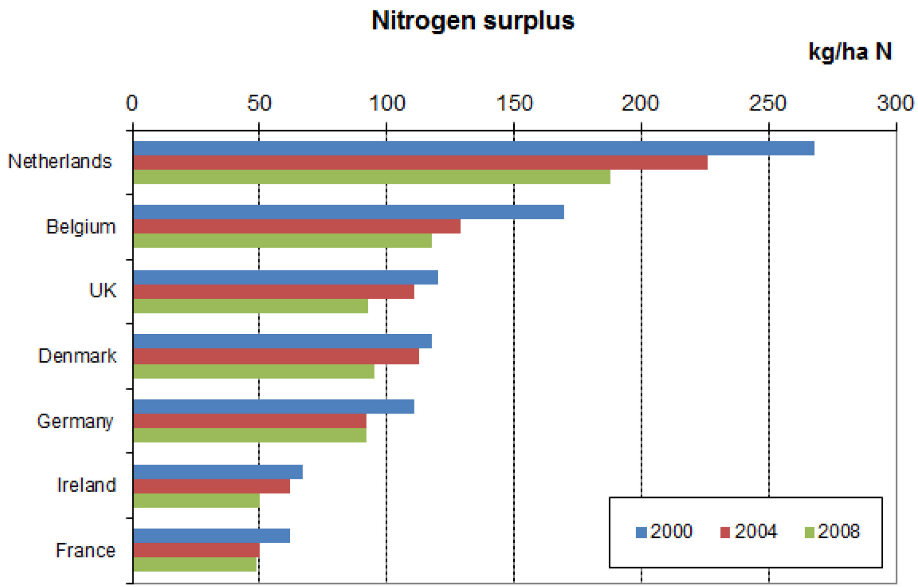

Fig. 1. Gross nitrogen balance between 2000 and 2008 (Eurostat, 2011).

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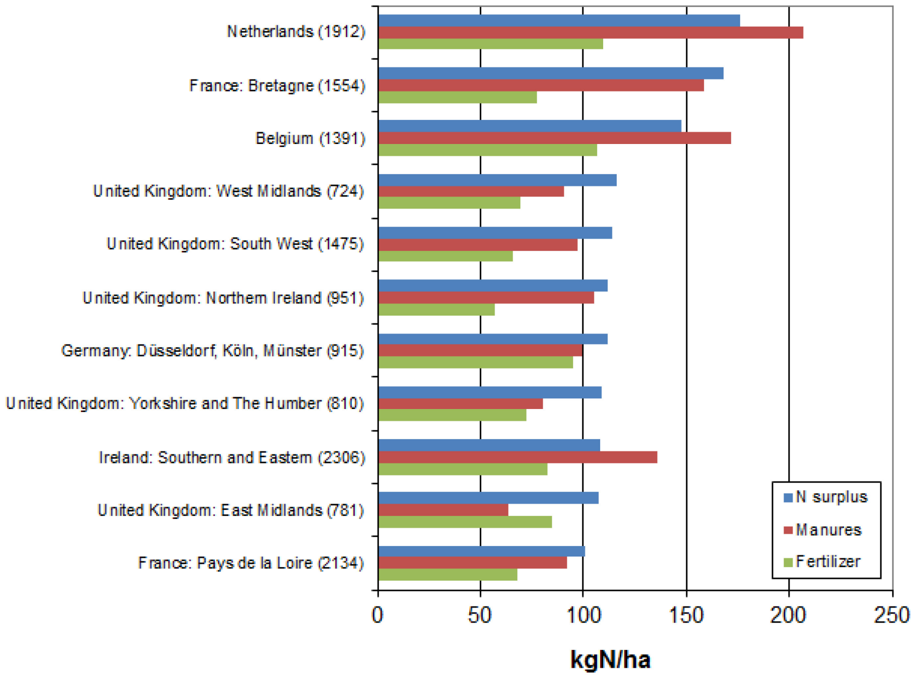

Fig. 2. Soil N balance (soil N surplus) and N inputs from manure and fertilizer in 2008 by MITERRA for regions in Northwestern Europe of comparable UAA and N surplus exceeding 100 kg N ha^{-1} (NUTS 1 level or clusters of NUTS 2; UAA in 1000 ha in between brackets).

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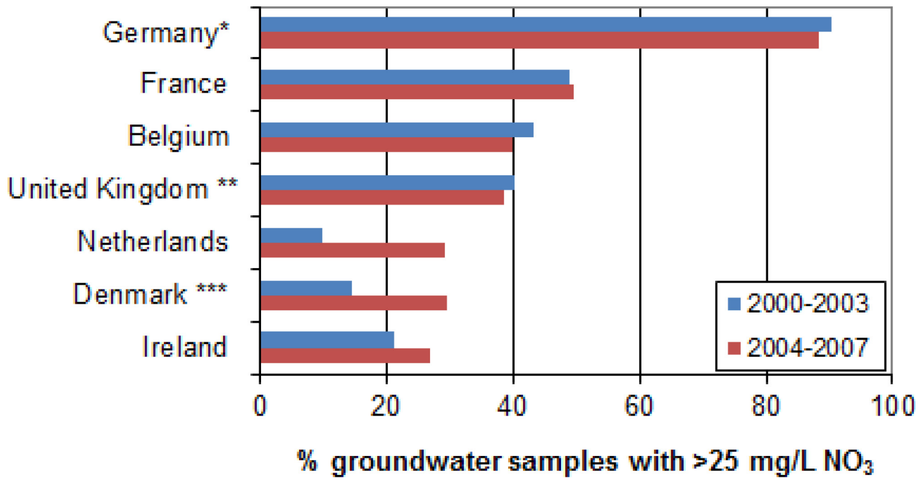

Fig. 3. Percentage of groundwater samples in monitoring programs for the Nitrates Directive exceeding 25 mg NO_3 l^{-1} for the 2nd and 3rd reporting period (European Commission, 2011).
* For Germany only data for the agriculture monitoring network.
** For the reporting period 2000–2003 UK reported only stations within England.
*** For the reporting period 2000–2003 Denmark provided aggregated results.

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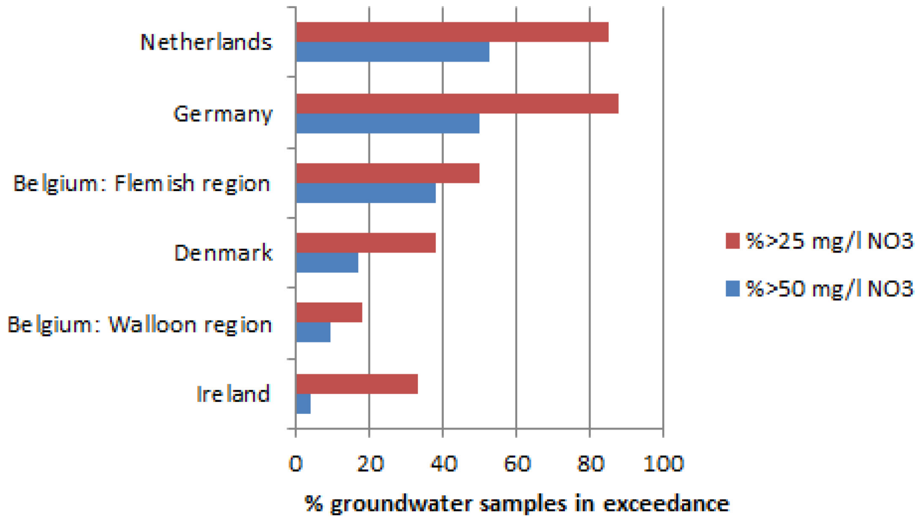

Fig. 4. Percentage of shallow phreatic groundwater samples in monitoring programs for the Nitrates Directive for the 3rd reporting period (2004–2007) exceeding 25 or 50 mg NO_3 l^{-1}.

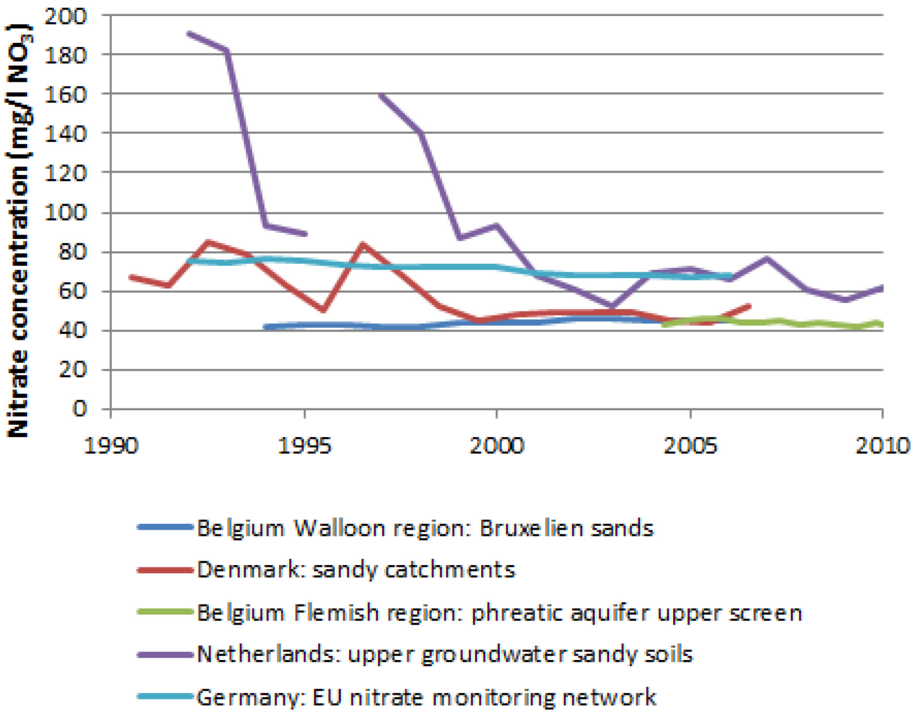

Fig. 5. Trend of nitrate concentrations in upper levels of phreatic groundwater in sandy soils, catchments or aquifers in monitoring programs for the Nitrates Directive (data taken from Fraters et al., 2011).

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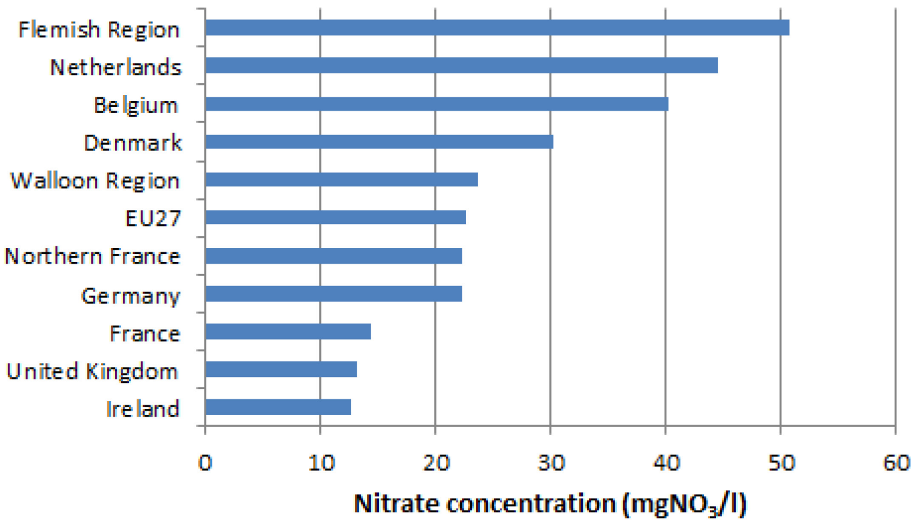

Fig. 6. Mean nitrate concentration (UAA and precipitation surplus weighted) in leaching water from agricultural soils in Northwestern EU in 2008 by MITERRA model.

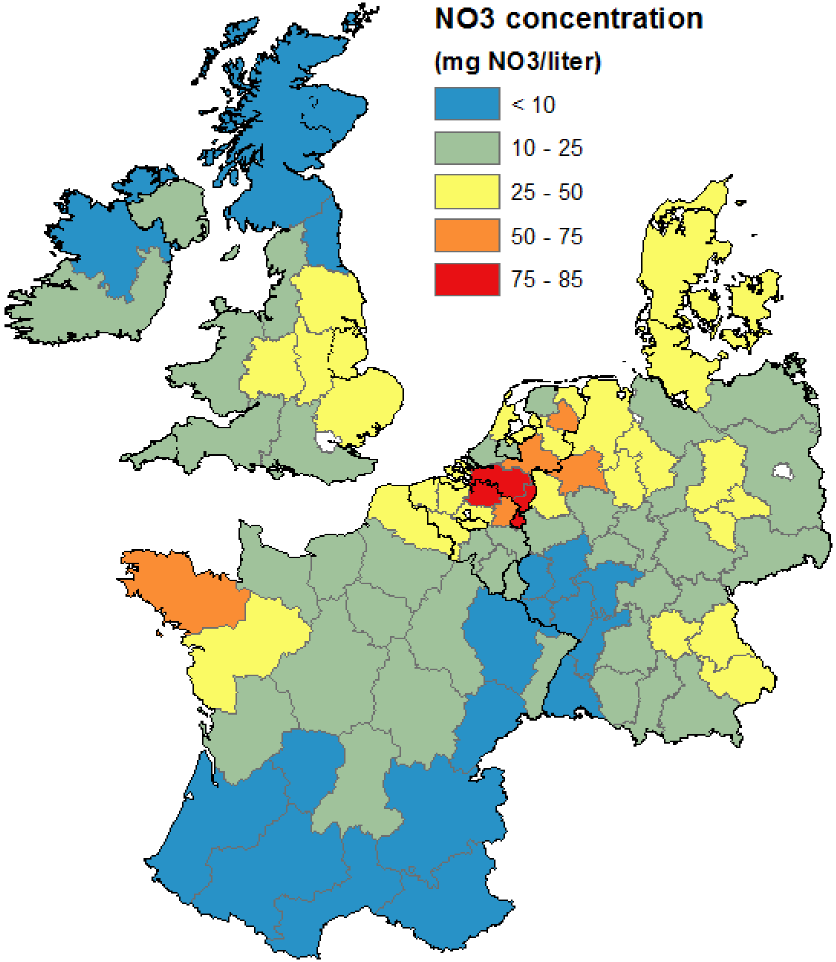

Fig. 7. Mean nitrate concentration in 2008 at NUTS 2 level by the MITERRA model.

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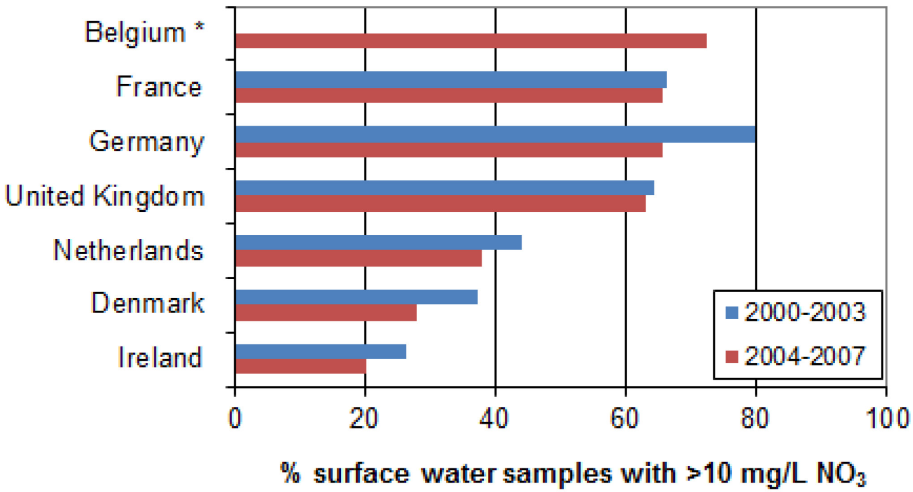

Fig. 8. Percentage of surface water samples in monitoring programs for the Nitrates Directive exceeding 10 mg NO_3 l^{-1} for the 2nd and 3rd reporting period (European Commission, 2011). *NO_3 data for 2000–2003 were not available.

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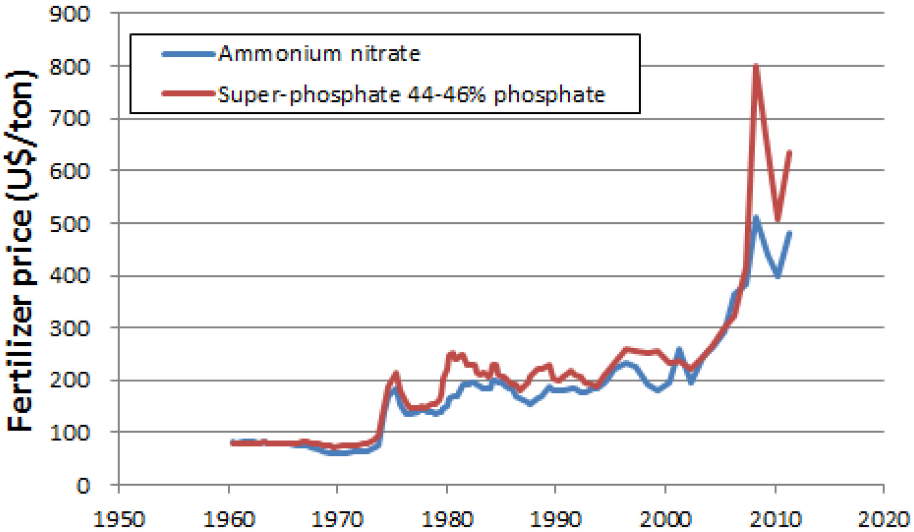

Fig. 9. Average US farm prices of common nitrogen and phosphate fertilizer (USDA, 2012; http://www.ers.usda.gov/Data/FertilizerUse/).

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