Chapter

15

Geographical variation in terrestrial nitrogen budgets across Europe

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Executive summary

Nature of the problem

- Nitrogen (N) budgets of agricultural systems give important information for assessing the impact of N inputs on the environment, and identify levers for action.

Approaches

- N budgets of agro-ecosystems in the 27 EU countries are established for the year 2000, considering N inputs by fertiliser application, manure excretion, atmospheric deposition and crop fixation, and N outputs by plant uptake, gaseous emissions, mineralisation, leaching and runoff.
- Country N budgets for agro-ecosystems are based on the models INTEGRATOR, IDEAg, MITERRA and IMAGE. Fine geographic distribution is depicted with the former two models, which have higher spatial resolution. INTEGRATOR is the only available model for calculating non-agricultural terrestrial N budgets systems.

Key findings/state of knowledge

- For EU-27, the models estimate a comparable total N input in European agriculture, i.e. 23.3–25.7 Mton N yr^{-1}, but N uptake varies largely from 11.3–15.4 Mton N yr^{-1}, leading to total N surpluses varying from 10.4–13.2 Mton N yr^{-1}. Despite this variation, the overall difference at EU-27 is small for the emissions of NH_3 (2.8–3.1 Mton N yr^{-1}) and N_2O (0.33–0.43 Mton N yr^{-1}) but estimates vary largely at a regional scale. The estimated sum of N leaching and runoff at EU-27 is roughly equal to the sum of NH_3, N_2O and NO_x emissions to the atmosphere, but estimates vary by a factor two, from 2.7 to 6.3 Mton N yr^{-1}.
- Trends in N fluxes in agro-ecosystems since 1970 show an increase in N inputs by fertilisers and manure up to 1985, followed by a decrease since 1985 in response to a decrease in crop production and in animal numbers. Actually, livestock decreased since 1970, but in the period 1970–1985 the N input by manure excretion still increased due to an increase in N excretion rates.
- In non-agricultural system (forests and semi-natural vegetation), the estimated total N input is near 3.2 Mton N yr^{-1}, while the net N uptake is near 1.1 Mton N yr^{-1}, leading to a surplus near 2.1 Mton N yr^{-1}. Compared to agricultural systems, the estimated N fluxes in non-agricultural systems are about five times lower for N_2O emissions and 10 times lower for NO_x and NH_3 emissions and for the sum of N leaching and runoff.

Major uncertainties/challenges

- The largest uncertainties in flux values, as estimated from inter-model comparison, concerns N leaching and runoff, followed by N_2O emissions, from agricultural ecosystems.

Recommendations

- Future research should focus on reducing the fluxes with the most uncertainty (N leaching and runoff, followed by N_2O emissions, from agricultural ecosystems), including studies on denitrification.
- To improve model assessments and enable model validation, databases should be set up of: (i) N contents in major crops/vegetation in various regions (to improve estimates of N uptake and N surplus), (ii) NH_3 and N_2O emissions based on inverse modelling approaches

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(to validate N emission calculations) and N concentrations in ground water and surface water (to validate N leaching and N runoff assessments).
- The number of countries with estimated NH_3-N emissions in 2000 exceeding the National emission ceilings for 2010 depends on the model approach and varies between 7 and 18. Exceedance of critical N concentrations in surface water is highly model-dependent. It is relevant that data use, both on activity data and emission or leaching factors is harmonised for models predicting air emissions and N loss to waters for consistent environmental decision-making relevant to air quality, ecosystem deposition and water quality.

15.1 Introduction

The major share of new reactive nitrogen (N_r) is introduced into the environment with the purpose of producing agricultural commodities. Excess N input, however, causes a number of ecological and human health effects, like acidification, eutrophication, elevated N saturation of forest soils, climate change and biodiversity impacts (see also Grizzetti *et al.*, 2011; Moldanová *et al.*, 2011; Butterbach-Bahl *et al.*, 2011, Dise *et al.*, 2011; Velthof *et al.*, 2011, Chapters 17–21, this volume). An indication of the potential impact of N inputs in agriculture can be derived by an overview of all N inputs and N outputs, here referred to as an N budget.

N budgets of agro-ecosystems are generally constructed (i) to increase the understanding of nutrient cycling, (ii) for use as performance indicator and to raise awareness in nutrient management and environmental policy, and (iii) as regulating policy instrument to monitor and enforce a certain nutrient management policy in practice (Oenema *et al.*, 2003). Sometimes, the term N balance is also used, but this term is consistently used in this chapter to denote the N surplus, defined as the sum of all N inputs minus N removal by feed and food, in line with its use by the Organisation for Economic Co-operation and Development (OECD) (OECD, 2001, 2007). We use the word N budget for a complete N flux assessment.

In this chapter, we present N budgets of agro-ecosystems and non-agricultural terrestrial ecosystems in Europe as performance indicator, illustrating the N use efficiency of agro-ecosystems and the loss of excess N to the environment (air and water). We summarise the present knowledge on European N budgets for terrestrial ecosystems by using a range of different modelling and input data assessment approaches. This way we implicitly assess uncertainties. As a part of the budget approach, the chapter includes key N fluxes, including N inputs by manure, fertiliser, deposition and fixation, N uptake, emissions of ammonia (NH_3), nitrous oxide (N_2O) nitrogen oxides (NO_x) and di-nitrogen (N_2), and the sum of N leaching and runoff, to provide an overall picture of the N status of Europe.

The assessment concentrates at discussing data at the country level with the EU-27 as geographical scope, even though the calculations are performed in many models at much higher resolution in order to cover the nonlinearity of the soil processes. Most data are available around the year 2000 and so most of the data presented are reflecting the situation around this year. However, we include also a discussion of the past trends of important elements in the N-budgets since 1970 onwards.

In Section 15.2, we first describe the modelling approaches and input data that are available to assess terrestrial N fluxes at the European scale. We then present results in terms of farm and land N budgets for agricultural systems, including trends in N budgets in the period 1970–2000 (Section 15.3) followed by land N budgets for non-agricultural terrestrial systems (Section 15.4). An overall evaluation of the results is given in Section 15.5. This includes an evaluation of the validity of the presented model approaches by comparison of model results with independent datasets, whenever available. Furthermore, the relevance of N budgets and their trends with respect to effects on ecosystems and the reliability of N budgets at various geographic scales are discussed. For a complete overview of aggregated N fluxes across media and sectors for countries throughout Europe, we refer to Leip *et al.*, 2011a (Chapter 16, this volume). Details on N sources in deposition are given in Simpson *et al.*, 2011 (Chapter 14, this volume).

15.2 Methodological approaches and input data to assess terrestrial nitrogen budgets at the European scale

15.2.1 Approaches to assess nitrogen budgets at regional scale

While we are interested to obtain N budgets for agriculture on a regional, country or European level, we need to differentiate different budgeting approaches by the respective system boundaries used. We distinguish three basic approaches in regional N budget studies, using the farm, land or soil as the gate at which the N inputs and outputs are quantified (see Table 15.1).

(1) *Farm nitrogen budget* (called farm-gate budget by Oenema *et al.*, 2003); it records the amounts of N in all kinds of products that enter and leave the farm via the farm-gate. Throughputs, as for example uptake of grass by animals, or the application of manure, are not part of the farm N budget. The surplus/deficit, i.e. the difference between inputs and outputs, is a measure of total N losses, adjusted for possible changes in the storage of nutrients in the farming system. Examples of this approach are the now abolished MINAS (Mineral Accounting System) regulatory nutrient book-keeping system in the Netherlands (Oenema *et al.*, 1998; Neeteson, 2000), and the OSPARCOM method (Oslo and Paris Conventions for the Prevention of Marine Pollution) focusing on N and P discharges into the North Sea and Baltic Sea from the surrounding countries (OSPARCOM, 1994). In the simple farm N budget, the N surplus is not further specified, whereas N (NH_3, N_2O, NO_x and N_2) losses from

Table 15.1 Definition of N inputs, N outputs and N surpluses in regional farm, land and soil nitrogen budgets for agricultural systems

System boundary	Budget		N Inputs	N Outputs	N Surplus[a]
	Simple	Detailed			
Farm	*Farm N budget*	*Agricultural system N budget*	Fertiliser, feed (concentrates), external organic N sources, N fixation and N deposition, net N manure import, and withdrawals	Sold animal (meat, milk, etc.) and crop products	N (NH_3, N_2O, NO_x and N_2) emissions and N leaching/runoff from housing and manure storage systems and soil; soil N stock changes
Land	*Gross N budget (OECD approach)*	*Land system N budget*	Fertiliser, manure excretion, external organic sources, crop residues returned on soils, N fixation, N deposition, net N manure import/ export, and withdrawals	Harvest of crop products (in arable land) or above ground removal of grass, crop residues	N (NH_3, N_2O, NO_x and N_2) emissions and N leaching/runoff from housing and manure storage systems and soil; soil N stock changes
Soil	*Soil N budget*	*Soil system N budget*	Fertiliser, manure application, grazing inputs, external organic sources, crop residues returned on soils, N fixation and N deposition	Removal of crop products (in arable land) or above ground removal of grass; crop residues, soil N stock changes	N (NH_3, N_2O, NO_x and N_2) emissions and N leaching/runoff from soil

[a] N surplus is specified in the detailed N-budgets

the housing and manure storage systems and from soil to the air and to aquatic systems are specified in a detailed *agricultural system budget*, as illustrated in Figure 15.1. An example of this approach is the CAPRI-DNDC model (Leip *et al.*, 2009).

(2) *Land nitrogen budgets* (called gross N balances by the OECD). It records all N that enters a farm land (including housing and manure storage systems) and leaves the farmland by crop products. Nitrogen inputs include fertiliser, animal manure production/excretion, biological N fixation and N deposition. This approach is used for example by the OECD as environmental performance indicator for agriculture (OECD, 2001, 2007). In the simple approach, called gross N budget (gross N balance by the OECD), the N surplus is not further specified, whereas N losses from the housing and manure storage systems and from soil to the air and to aquatic systems are specified in a *detailed land system budget*. This approach is used in this chapter.

(3) *Soil nitrogen budget* (*called soil surface budget by Oenema et al., 2003*). It records all N that enters the soil and that leaves the soil via crop uptake, including nutrient gains and losses within the soil. Nitrogen inputs via animal manure are adjusted for losses of N emissions in housing and manure management systems; all other N inputs are the same as for the land N budget. Nitrogen output (defined here as output of 'useful product') is corrected by the changes of N storage; accumulation of N in organic matter is regarded as useful because it improves soil quality and can potentially contribute to crop growth in following years. Soil N surplus (see Table 15.1) is then a measure for the total N loss from the soil to either the atmosphere (NH_3, N_2O, NO_x and N_2 emissions) or the hydrosphere (N leaching to ground water and N runoff to surface water). In the soil N budget, this N surplus is not further specified, whereas in the *soil system budget* all N inputs and outputs, including N gains and losses within and from the soil are specified. It should be noted, that in the literature the soil N budget mostly differs from our definition, as the NH_3 emission from soils is often already corrected for while the soil N changes are included in the calculation of the surplus (Oenema *et al.*, 2003).

The N surplus gross N budget includes the sum of all nutrient emissions from agriculture into soil, water, and air (OECD, 2007) and is thus often used as the indicator of agricultural pressure on water quality (EEA, 2005), as it allows identifying areas with high risk of N leaching. Detailed budgets are able to resolve the individual pathways of N as presented in Table 15.1. It is important to remember that different accounting methods cover different N flows. Animal housing and manure management systems are not included in the soil budgets, while they are accounted for in farm and land budgets. In the land N budgets, the N excreted in the manure is considered, while in the soil N budget only the N in applied manure, corrected for losses in housing and manure management systems, is accounted for. Manure used for other purposes (e.g. burning) is not considered in both approaches. With respect to 'mineral N fertiliser', the farm N budget considers fertiliser *purchases*, while mineral fertiliser *applications* are relevant for the land and soil N budgets.

While for soil budgets the system boundaries are usually the top soil layer (surface to rooting depth), and covers thus only land-based agricultural production, farm and land budgets include also the livestock sector. As for the farm (and agricultural systems) budget, the boundary is the farm, they don't consider manure and animal intake of N in fodder produced in the farm as input or output. However, if data are available, they are often quantified as N *throughput*. The difference in farm, land and soil budgets is illustrated further in Leip *et al.* (2010).

15.2.2 Modelling approaches

There are several operative activities that estimate N budgets for the European Union and for Europe at various spatial resolutions. Table 15.2 gives an overview of main model approaches that have been used for assessing total agricultural emissions of different forms of reactive N for various parts of Europe (from EU15 to whole Europe), at various geographic resolutions (from grid to country) and for different time periods. The approaches included in Table 15.2 are: (i) complete land system N budget models for agriculture, using yearly time steps (INTEGRATOR, CAPRI, IDEAg, MITERRA, IMAGE), (ii) emission factor approaches for both agricultural and total annual NH_3, N_2O and NO_x emissions to the atmosphere (GAINS, EMEP, EDGAR, UNFCCC-IPCC) and (iii) N loss models to either surface water (GREEN) or ground water (EPIC).

In the supplementary information to this chapter (Supplementary material, Chapter 15 & 16), a description of the various models mentioned above and the meaning of their abbreviations is given. In short, the complete land system N budget models are able to calculate all N fluxes to and from a land system, as defined in Table 15.1. First of all, these models are able to assess the N surplus or gross soil N budget according to (see Table 15.1): N surplus = input (mineral fertilisers + livestock manure excretion corrected for transport + other organic sources + left crop residues + biological fixation + atmospheric deposition) – total crop removal – total forage uptake. The models are also all able to simulate the fate of the N surplus in terms of NH_3, N_2O, NO_x and N_2 emissions from housing and manure storage systems, N accumulation in or release from the soil (not in all models) and N losses by leaching and runoff. The emission factor approach models are limited to atmospheric emissions, but unlike the land system N budget models they include all sectors, including traffic and industry. Similarly N loss models are limited to estimates of N losses to surface water and/or ground water, but they generally include all N sources, including human sewage and direct deposition inputs to surface water.

In this chapter, we focus on complete N budgets for agriculture, as derived with INTEGRATOR, IDEAg (CAPRI based model), MITERRA and IMAGE. More details on these models is given in the supplementary materials at Chapter 15 and 16 and in De Vries *et al.* (2010b). We also include a comparison of results of NH_3, N_2O and NO_x emissions with the emission factor approaches (GAINS, EMEP, EDGAR, UNFCCC-IPCC), while results of the model GREEN are shown to illustrate the impact of diffuse sources versus point sources.

There are also detailed ecosystem models available that provide process-level descriptions for either daily NH_3, N_2O and NO_x emissions, such as the DNDC model (Li *et al.*, 2000) or N leaching, such as the EPIC model (Bouraoui Aloe, 2007; Van der Velde *et al.*, 2009) that have been applied to derive N fluxes at regional scale in Europe.

The DNDC model has for example been used to assess N_2O and NO_x emissions for both forests (Kesik *et al.*, 2005) and agricultural land (Butterbach-Bahl *et al.*, 2009) at a fixed 10 km × 10 km grid, while the EPIC model that has been applied to study the effect of agricultural practices and biofuel cultivation on N leaching (Bouraoui and Aloe, 2007; Van der Velde *et al.*, 2009). However, these models do not include emissions from housing systems and in case of EPIC also not explicitly from soils, and are therefore not included in the model comparison presented in this paper. Some results are, however, shown in the Supplementary material (Chapter 15 and 16).

15.2.3 Data sets to estimate nitrogen inputs and outputs

In order to understand the operation of models, an overview of internationally coherent datasets used by the models is given. In addition to these international datasets, often national information also exists, but in general this cannot be assessed by activities operating on a European scale.

Inputs of N to agricultural systems include N fertiliser, N manure due to application and grazing, N deposition and N fixation. Data sets that are relevant for the assessment of N uptake are crop yields and element contents in crops, while N and C pools are relevant for the assessment of N emission fluxes. The assessment of N fluxes to the air (emissions of NH_3, N_2O, NO_x, and N_2) and water (N leaching to ground water, N surface runoff and subsurface flow to surface water) requires data on emission and leaching parameters in the various models to make such predictions. An overview of the data used by all the four complete N budget models is given in De Vries *et al.* (2011). More information on the datasets that are used to calculate the amount of fertiliser and manure N applied to soil is given in Supplementary material Chapter 15 and 16.

In biogeochemistry models, soil C and N contents often strongly determine the N_2O flux. Maps of present concentrations and pools of C and N in the soil and C/N ratios in the soil distinguishing between agricultural soils and non-agricultural soils can be based on various databases, i.e. WISE/SOTER, European Soil Data Base (ESDB2) and ICP forests database. More information on approaches and results is given in the Supplementary material (Chapter 15 and 16).

15.3 Farm and land nitrogen budgets for agricultural systems

In the following sections, data on farm and soil N budgets are presented focusing on two recently developed model systems, i.e. IDEAg and INTEGRATOR. IDEAg consists of three

Table 15.2 Overview of available models approaches for assessing emissions of different forms of N, for various parts of Europe at various geographic resolutions and for various time periods

Model approach	Element flux considered	Method	Sectors considered	Area involved	Geographic resolution	Time
Complete land N budget models						
INTEGRATOR (De Vries *et al.*, 2010)	N_2O, NO_x and NH_3 emission, N leaching, N runoff	Adapted MITERRA approach for agricultural systems. Statistical model for terrestrial systems	Agriculture, terrestrial systems	EU-27+3	NCU[a]	1970–2000
MITERRA (Velthof *et al.*, 2007, 2009)	N_2O, NO_x and NH_3 emission, N leaching, N runoff	Emission and leaching factor approach for agricultural systems	Agriculture	EU-27	NUTS2	2000
CAPRI (Britz, 2005; Britz *et al.*, 2005; CAPRI 2010)	NH_3, N_2O, N surplus	Mass-budget model using an emission-factor approach	Agriculture	EU-27	NUTS2	Base year currently 2002 projections up to 2012
IDEAg, (Leip *et al.*, 2008)	N_2O, NO_x and NH_3 emission, N leaching	Economic model for agriculture, linked to mechanistic model to simulate soil N budget	Agriculture	EU-27	HSMU[a]	2000
IMAGE (Alcamo, 1994; Leemans *et al.*, 1998; MNP, 2006; IMAGE, 2010)	N_2O, NO_x and NH_3 emission, N leaching, N runoff	Extended emission factor approach with consideration of mitigation technologies	All sectors	Europe Global	Country	Present, projections
N emission models to atmosphere						
GAINS (Höglund-Isaksson and Mechler, 2005; Winiwarter, 2005) http://gains.iiasa.ac.at/gains/EU/index.login?logout=1	N_2O, NO_x and NH_3 emission	Extended emission factor approach with consideration of mitigation technologies	All sectors	Europe Global	Country	Present, projections
EDGAR (Van Aardenne, 2002) http://edgar.jrc.it	NH_3, N_2O and NO_x emission	Extended emission factor approach with consideration of mitigation technologies	All sectors	Global	1 × 1 degree. The latest version (released 11/2008) is 0.1 × 0.1 degree	Past and present
EMEP (Simpson *et al.*, 2003, 2006; EMEP, 2010a)	NO_x and NH_3 emission N deposition	Emissions (disaggregated from official national inventories) and Atmospheric dispersion model	All sectors	Europe	50 km × 50 km; 5 × 5 km possible (e.g. Vieno *et al.*, 2009)	Past, present and projections up to 2030

Table 15.2 *(cont.)*

Model approach	Element flux considered	Method	Sectors considered	Area involved	Geographic resolution	Time
UNFCC/IPCC (IPCC, 2006; UNFCCC, 2010)	N_2O (and NO_x) emission	Emission factor approach on activity data	All sectors	Europe and other 'Annex-I' countries (industrialised)	Country	1990–present
N loss models to hydrosphere						
GREEN (Grizzetti *et al.*, 2005, 2008; Bouraoui *et al.*, 2009)	Total N diffuse emissions to waters and total N runoff	Geospatial empirical regression model	Agriculture and Point Sources	Europe	Sub-catchments (average size 180 km^2)	1985–2005
EPIC (Bouraoui and Aloe, 2007; Van der Velde *et al.*, 2009)	NO_3, NH_4, total N, soluble and particulate N runoff, N leaching	Detailed mechanistic model	Agriculture, terrestrial systems	EU-27 + Swiss	10 km × 10 km grid (including multiple crops)	1985–2005

[a] HSMU = Homogeneous Spatial Mapping Units; NCU = NitroEurope Calculation Units. Units refer to clusters of 1 km^2 grid cells that are characterised by similar environmental and/or agronomic conditions

elements: (i) the CAPRI-SPAT downscaling model (Leip *et al.*, 2008); (ii) the DNDC-CAPRI meta-model (Britz and Leip, 2009b); and (iii) an interface combining results of the DNDC-CAPRI meta-model with elements of CAPRI-SPAT, yielding a database with environmental indicators that are inherently consistent and operating at the level of individual crops. These models use the most detailed geographically explicit input data currently available, thus allowing the best way to map the various N fluxes included in the N budget. In particular, the DNDC-CAPRI meta-model is based on detailed spatial information, partly based on biophysical model simulations. A special feature of INTEGRATOR is that it includes historical data up to 1960, thus allowing the assessment of trends in N budgets. Despite the high spatial resolution of the data available in these model systems, results presented in this chapter are mainly restricted to model comparisons at the Europe-wide scale (tables of complete N budgets) and at the national scale (scatter plots of N fluxes). Detailed maps are limited to N input by manure and fertiliser and to NH_3 and N_2O, emissions from the agricultural system (both housing systems and soil) as derived by IDEAg and INTEGRATOR. Detailed maps of total N emissions divided in various sectors are further presented in Leip *et al.*, 2011a (Chapter 16 this volume).

15.3.1 Farm nitrogen budget

The IDEAg model system can be used to provide an updated picture of a farm N-budget for Europe. In IDEAg, a combination of the farm budget (animal and crop production in relation with the EU and global market) and soil N budget has been implemented (see Figure 15.1). As explained above, the farm N budget comprises as inputs feed intake and as output animal products, both driven by the economic situation of the farm (i.e. region). The N surplus is exported to manure management systems and finally applied to crops or excreted on grassland by grazing animals (other uses of manure are not significant in Europe and are not considered in IDEAg). IDEAg also calculates the fate of animal and crop products and distinguishes human consumption, processing by the industry to generate feed concentrates, biofuels or other products and, in- and export for each commodity considered. Also, losses at the market (and at the farm) are estimated. As a result, the IDEAg system is able to depict a detailed picture of N-flows of the agriculture sector at the European scale.

15.3.2 Land nitrogen budgets

Detailed land nitrogen budgets at European level

An overview of a detailed European (EU27) field scale (land) N budget is presented in Table 15.3. The table compares results derived with INTEGRATOR (De Vries *et al.*, 2010) with information from IDEAg (Britz and Leip, 2009a), MITERRA (Velthof *et al.*, 2007, 2009) and IMAGE (De Vries *et al.*, 2009). Furthermore, the sum of the officially submitted data to the UNFCCC secretariat by the 27 EU countries, as reported in the Annual European Community greenhouse gas inventory, are presented (EEA, 2008). Results include N (NH_3, N_2O, NO_x and N_2) emissions from housing systems to give complete emission estimates from the agricultural system. Consequently, we include manure excretion instead of manure application as input to the system. For EU27, the four models estimate a total N input in European agriculture of 23.3–25.7 Mton N yr^{-1}, which is mainly due to fertiliser and animal manure inputs and to a lesser extent caused by atmospheric deposition and N fixation. The N uptake varies from 11.3–15.4 Mton N yr^{-1} leading to total N surpluses (N input not used by the plants) varying from 10.4 to 13.2 Mton N yr^{-1} at EU27 level. The lowest surplus is calculated by INTEGRATOR, as it assesses the highest uptake. The various models give in general very similar results

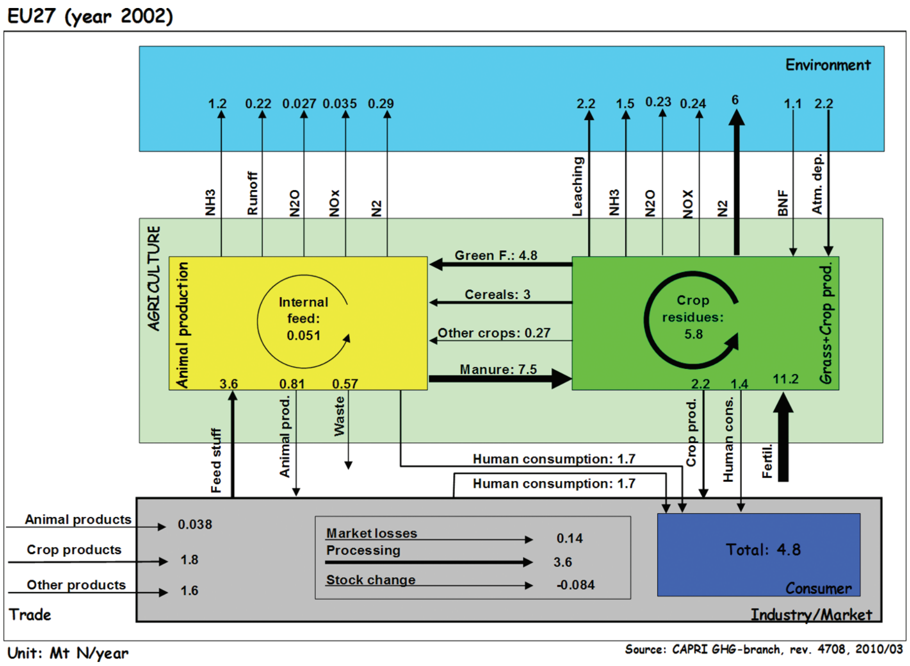

Figure 15.1 N budget for the agricultural sector in EU27 for the year 2002 as calculated by the IDEAg model.

for the emissions of NH_3 (2.8–3.1 Mton N yr^{-1}). Comparable estimates are also derived for the direct N_2O emissions (0.33–0.43 Mton N yr^{-1}), but NO_x emissions vary by a factor 10 (0.02–0.22 Mton N yr^{-1}). The sum of N leaching and N run-off also varies largely. The estimates by IDEAg and IMAGE are nearly twice as large as the estimates by INTEGRATOR and MITERRA, causing a much lower estimated N_2 emission by IDEAg and IMAGE as compared to INTEGRATOR and MITERRA (Table 15.3).

An important difference in this context is also that both INTEGRATOR and IDEAg include mineralisation estimates, whereas this input term is neglected in MITERRA and IMAGE. In INTEGRATOR, the net release is mainly determined by the N mineralisation in drained peat soils. In IDEAg, mineralisation of all soils is obtained from the DNDC meta-model and then scaled in two steps (the second jointly with N_2 flux estimates) to close the N budget.

More details on the N emission sources calculated by the various models are given in Table 15.4. Results show that the difference in NH_3 emissions between IDEAg versus the other three models is the result of the higher emissions from housing and manure storage systems. Another notable difference is the much higher N_2O and NO_x emission from grazing by IMAGE as compared to the other models (Table 15.4).

Reasons for the various similarities and differences can be summarised as follows.

- All model give similar results for the N inputs by fertiliser as they use the same FAO data regarding fertiliser rates.
- Deviations between inputs by manure application are larger due to different sources for animal numbers, but specifically due to deviating N excretion rates.
- Differences in biological N fixation mainly follow from the different values used to derive N fixation of pulses/legumes as a fraction of the harvested N amount, as summarised in the supplementary material chapters 15 & 16 (see also De Vries *et al.* (2010b).
- The NH_3 emissions by INTEGRATOR, IDEAg and MITERRA are comparable as they are based on the same GAINS dataset. There are however differences in N manure and fertiliser distribution and this affects the N leaching that is affected by soil type, land use, etc.
- The difference in N_2O emissions is limited on a European wide scale, considering the differences in N_2O emission factors used. In INTEGRATOR, these emissions are determined as a function of soil type, land use, manure type, etc. In IDEAg, results are based on the DNDC-CAPRI meta-model, whereas MITERRA uses standard emission fractions based on IPCC. These differences do, however, affect the spatial variation in N_2O emissions (see Section 15.3.3).
- The higher estimated sum of total N leaching and runoff by IDEAg and IMAGE are mainly due to higher leaching and runoff fractions. In IDEAg, N leaching is based on the DNDC meta-model whereas N leaching by the other models depends on various environmental factors as described in detail in De Vries *et al.* (2011). Apparently, the difference in parameterization of the factors and in geographic resolution leads to strongly different results.

Land nitrogen inputs and nitrogen surplus at country level

Land N budgets at country-scale for agriculture for the year 2000 calculated by INTEGRATOR, IDEAg, MITERRA and IMAGE for the various EU countries are presented in the Supplementary materials (Chapter 15 and 16). A scatter diagram of the N inputs as calculated with the INTEGRATOR model compared to IDEAg, MITERRA and IMAGE is given in Figure 15.2. The four approaches generally agree for fertiliser input and N inputs by manure, which is logical as it has the same

Table 15.3 Annual N budgets of agricultural land in Europe in 2000, including N (NH_3, N_2O, NO_x and N_2) emissions from housing systems and from soil. Output terms in italic are summations of more detailed N fluxes and should not be added in the calculation of the total N output

	N budget (Mton N yr^{-1})				
Source	**INTEGRATOR EU 27–2000**	**IDEAg EU25–2002**	**MITERRA[a] EU 27–2000**	**IMAGE[a] EU 27–2000**	**UNFCCC[b] EU27–2002**
Input to land					
Biological fixation	1.3	1.0	0.8	1.4	1.1
Manure excretion	10.3	8.8	10.4	9.8	9.1
Synthetic fertiliser	11.5	11.4	11.3	11.3	10.6
Atmospheric deposition	2.7	2.1	2.0	2.8	—
Total	25.7	23.3	24.5	25.3	20.8
Output from land					
Plant removal	15.4[c]	12.5	11.3	13.5	—
N accumulation	−3.3	−3.5	—	—	—
Emissions of					
NH_3	2.9	3.1	2.9	2.8	3.1
N_2O[d]	0.40	0.43	0.33	0.43	0.4
NO and NO_2	0.21	0.11	0.02	0.22	—
N_2	7.0	4.5	7.2	2.5	nd
Total (De)nitrification	7.6	5.1	7.8	3.1	—
N leaching	2.8	5.7	2.0	—	—
N surface runoff	0.35	0.4	0.75	—	-
Total leaching/runoff	3.1	6.1	2.7	5.9	6.6
Total surplus	10.4	10.8	13.2	11.8	—
Total	25.7	23.3	24.5	25.3	—

[a] Details of the comparison between MITERRA and IMAGE are described in De Vries *et al.* (2009).
[b] Source: EEA (2008).
[c] Uptake includes the removal from grassland, rough grazing areas and the net crop removal from arable land.
[d] N_2O emission refers to direct N_2O-N emission only that is calculated by all models.

basis although the IDEAg N manure inputs are consistently lower (see also Table 15.4). There are relatively large differences for the other N inputs (deposition and fixation) at country level, but this hardly affects the total N inputs by the four models, which are comparable for all countries. Total N uptake is quite different between the various approaches. As with the results at European scale (see Table 15.4), INTEGRATOR results are consistently higher than the other models. The uptake mostly decreases according to INTEGRATOR > IMAGE > IDEAg > MITERRA. Furthermore, there is quite some scatter at country level. This is reflected in an even larger scatter for the N surplus per country, indicating an uncertainty near 50% for country estimates of the N surplus.

Nitrogen emissions to air and water at country level

Instead of quantifying just the gross N surplus, the N excess input can be further defined in terms of N (NH_3, N_2O, NO_x and N_2) emissions to the atmosphere, N leaching and N runoff. The N budget models described before can derive such detailed agricultural N budgets not only at European level (see Section 15.2.1), but also at country level. An example of such an output calculation using INTEGRATOR is given in Table 15.5.

To gain insight in the comparability of the results obtained, a comparison is given of agricultural emissions of NH_3-N, N_2O-N and NO_x-N and N leaching for 27 EU countries for the year 2000 as derived with INTEGRATOR with those obtained by the complete N budget models (IDEAg, MITERRA and IMAGE). Furthermore, results for the N emissions were compared with standard activity data-emission factors approaches (UNFCC/IPCC, 2010; GAINS, 2010; OECD, 2010; EDGAR, 2010; and EMEP 2010b). Data used for the results of the various models for NH_3-N, N_2O-N and NO_x-N are found in the Supplementary data for Chapter 15.

A comparison of country emissions for NH_3-N, N_2O-N and NO_x-N and of N leaching plus runoff (kton N yr^{-1}) within EU 27 as derived with INTEGRATOR with the various other approaches is given in Figure 15.3. Results show comparable estimates for NH_3 emissions, which is due to the use of comparable databases for the estimation. Both INTEGRATOR and MITERRA use the N excretion and NH_3 emission constants derived by GAINS and consequently, the differences should be

Table 15.4 Annual N emissions from agriculture in Europe for the year 2000

		N emissions in 2000 (kton N yr^{-1})			
N source	**Emission source**	**INTEGRATOR EU 27–2000**	**IDEAg EU 25–2002**	**MITERRA[a] EU 27–2000**	**IMAGE[a] EU 27–2000**
NH_3	Housing and storage	1189	1428	1279	1048
	Fertiliser application	1413[b]	678	540	798
	Manure application		759	823	683
	Grazing	271	201	231	319
	Total agriculture	2873	3066	2873	2848
N_2O	Housing and storage	55	48	54	52
	N application[c]	242	316	208	289
	Grazing	124	67	66	92
	Indirect emissions	43	80	51	76
	Total agriculture	401 (444)[d]	431 (531)	328 (379)[d]	434 (510)[d]
NO and NO_2	Housing and storage	20	32	36	0
	N application[c]	123	16	25	23
	Grazing	63	59	32	196
	Total agriculture	**207**	**108**	**93**	**219**

[a] Details of the comparison between MITERRA and IMAGE are described in De Vries *et al.* (2009).
[b] Includes emissions through soil inputs by fertiliser and manure application.
[c] Includes emissions through soil inputs by fertiliser and manure application, deposition, mineralisation, fixation and crop residues.
[d] The value in brackets are the total N_2O emissions calculated by INTEGRATOR, IDEAg, MITERRA and IMAGE including also indirect N_2O emissions due to N leaching and NH_3 and NO_x emissions.

small and are mainly due to the use of different statistics for animal numbers. Furthermore, all models use comparable statistics for N fertiliser use and NH_3 emissions from manure.

The differences in different N_2O emissions, however, are much larger, reflecting the larger variation in model approaches, specifically the use of N_2O emission factors. For example, a comparison of INTEGRATOR results with the N_2O emissions reported by the EU countries to the UNFCCC-IPCC shows quite a disagreement. For MITERRA, there is a good agreement with estimated N_2O emission from manure management, and direct soil N_2O emission (Velthof *et al.*, 2009), since both methods are based on the same N_2O emission fractions as a function of N inputs. Deviations between UNFCCC figures and MITERRA are thus only due to differences in activity data and the use of specific emission factors by some countries. By contrast, INTEGRATOR uses emission factors that depend on N source and environmental conditions. In both INTEGRATOR and MITERRA, the estimated indirect N_2O emission (not shown here) are much smaller than those reported to the UNFCCC, owing to both a lower N_2O emission factor and a lower N leaching fraction. Firstly, the revised IPCC emission factor for N leaching (IPCC, 2006) was used in both INTEGRATOR and MITERRA-EUROPE (i.e. 0.0075 kg N_2O-N for each kg N that leaches), whereas the values of the UNFCCC for most countries were obtained using the former emissions factor of 0.025 kg N_2O-N per kg N leached (IPCC, 1997). Secondly, IPCC uses a simple method to calculate leaching, i.e. 30% of the total N input via fertiliser, manure, grazing and other sources leaches to ground water and surface water (Mosier *et al.*, 1998). INTEGRATOR and MITERRA use a different approach to calculate N leaching which resulted in leaching losses of 11% of the total N input in EU-27.

The NO_x emissions appear to be very uncertain (see Figure 15.3). This is in line with results obtained by Butterbach-Bahl *et al.* (2009), who applied the approach used in IMAGE and three other empirical emission models, using the same input data for all models. More information on that approach and related results is given in the Supporting material in Chapters 15 and 16. The sum of N leaching plus runoff also varies largely within EU 27 and is systematically higher for IDEAg and IMAGE as compared to INTEGRATOR and MITERRA, in line with the results at European level. This implies that the used N leaching factors are highly uncertain and need further refinement.

15.3.3 Mapping the European agricultural nitrogen fluxes

The national N inputs and N outputs presented in Section 15.3.2 do not show the regional differences in N fluxes. In this section we provide maps showing such differences, focusing on presentations with IDEAg and INTEGRATOR for agricultural ecosystems in EU-27 for the year 2000. These two models were used to illustrate the geographic variation in model results, because of their highly disaggregated model input data. With respect to the emission of greenhouse gases, such as N_2O, it is

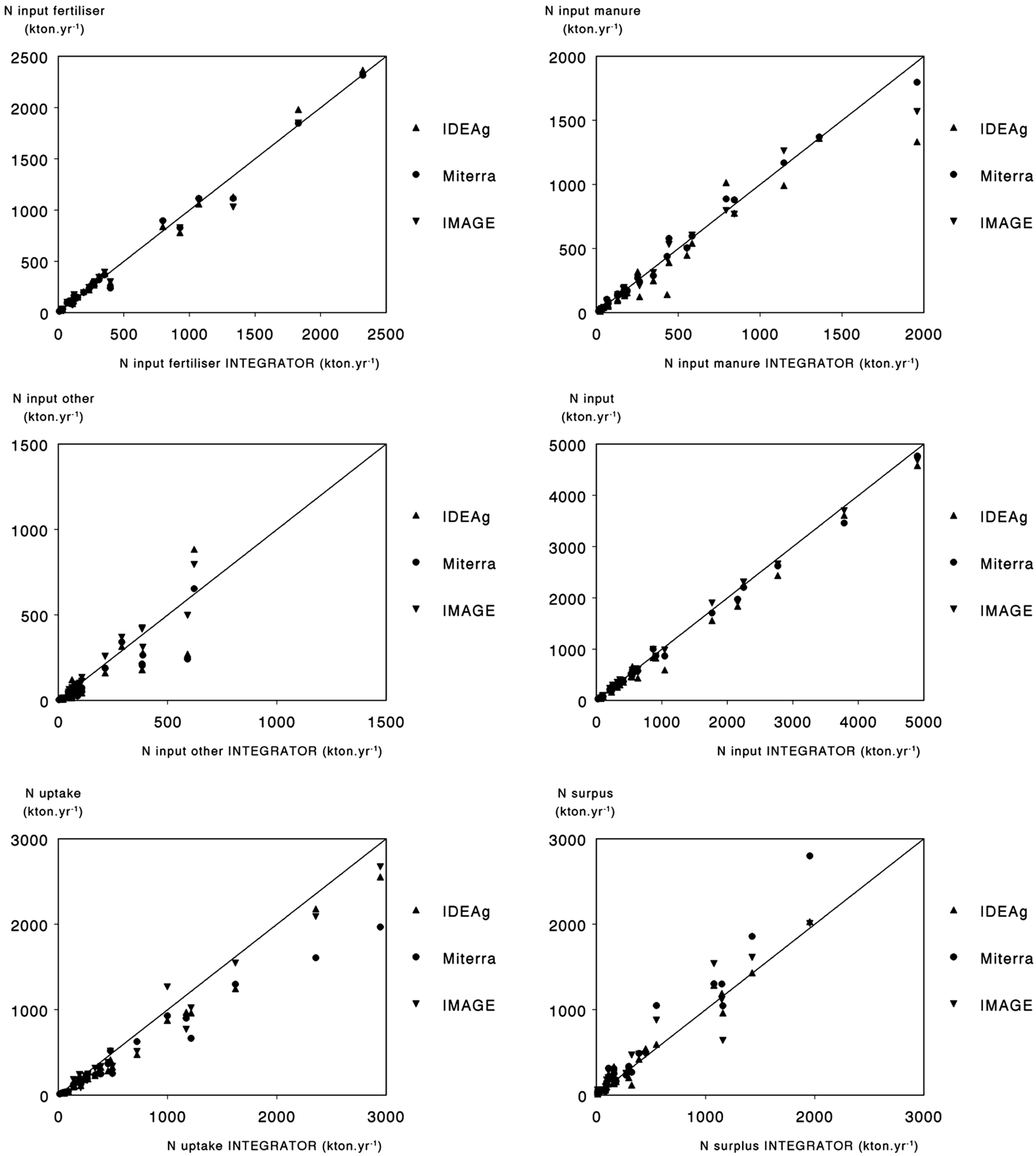

Figure 15.2 A comparison of country N inputs by fertiliser, manure, other inputs (deposition and fixation), total N inputs, total net N uptake and N surplus within EU27 as derived with INTEGRATOR, IDEAg, MITERRA and IMAGE for the year 2000 (IDEAg is 2002).

crucial to know whether total emissions for the area considered are correct, whereas accurate information on the spatial distribution of the emissions is less relevant. The latter aspect is, however, crucial when assessing the risk of elevated NH_3 emissions, and related N deposition, and of N leaching and N runoff in view of eutrophication impacts on terrestrial and aquatic ecosystems. Here, aggregation of input data for large areas may cause accurate average N deposition and N leaching levels, but a strong deviation in the area exceeding critical N deposition loads or critical N concentrations in ground water

Table 15.5 N emissions to air and water calculated at country level with INTEGRATOR for the year 2000

		N output fluxes (kg N ha^{-1} yr^{-1})				
Country[a]	**Area (Mha)**	**Emission NH_3**	**Emission N_2O**	**Emission NO_x**	**Emission N_2**	**Leaching + runoff**
Austria	3.336	13.2	2.1	0.9	23.1	11.1
Belgium	1.779	41.6	5.6	2.2	70.3	32.6
Bulgaria	6.816	4.7	0.9	0.4	16.3	4.4
Czech. Rep	4.776	11.1	2.5	1.0	38.7	18.4
Denmark	3.273	19.9	1.8	0.9	27.5	25.4
Estonia	1.846	3.8	1.1	0.5	25.5	5.4
Finland	6.914	2.0	0.4	0.1	17.6	4.6
France	35.346	14.8	2.6	1.2	30.8	13.7
Germany	21.566	20.2	2.4	1.1	42.4	19.5
Greece	8.404	6.2	1.2	0.7	20.9	10.4
Hungary	6.739	8.6	1.5	0.7	38.1	10.1
Ireland	5.043	15.5	5.4	2.8	37.3	11.5
Italy	18.434	17.6	1.9	0.9	33.3	19.1
Latvia	3.343	2.7	0.6	0.3	15.9	6.0
Lithuania	4.246	5.4	1.2	0.5	32.5	17.9
Luxembourg	0.144	20.8	6.9	0.0	34.6	13.9
Netherlands	2.491	52.6	4.8	2.4	94.3	45.0
Poland	20.265	10.6	1.3	0.4	32.0	15.6
Portugal	5.411	8.1	1.1	0.6	30.1	14.4
Romania	14.517	7.9	1.1	0.5	23.8	8.1
Slovakia	2.664	9.4	1.5	0.8	21.4	13.5
Slovenia	0.779	20.5	2.6	1.3	28.2	12.8
Spain	35.027	7.2	0.8	0.4	16.2	7.6
Sweden	7.914	4.2	0.6	0.3	12.9	5.8
UK	16.237	15.2	4.1	1.8	39.1	15.3
EU-27	237.310	12.1	1.9	0.9	29.3	13.2
EU-27[b]	237.310	2873	444	207	6965	3136

[a] Data for Cyprus and Malta are not included.
[b] Data given in kton N yr^{-1}.

and surface water (De Vries *et al.*, 2009). For this reason, it is relevant to make use of models with the highest level of spatial detail with respect to inputs and outputs, such as IDEAg and INTEGRATOR. The datasets mentioned in the Supplementary materials in Chapters 15 and 16 in combination with various downscaling techniques have been used to 'regionalise' the agricultural N inputs from statistical data at national or sub-national level to the NCU or HSMU level.

Nitrogen inputs

Inputs by manure and fertiliser Input of mineral N fertiliser and manure N as derived by IDEAg and INTEGRATOR are shown in Figure 15.4. The legend of 170 kg N is chosen as this is the maximum allowed manure N input in the EC, with the exception of a derogation (accepted after 2000) of 250 kg N for the Netherlands and 230 kg N for Denmark, Germany and Austria. High manure N application rates occur in areas of high livestock density in Europe and include parts of Denmark, the Netherlands, Belgium, Wales, Ireland, Catalonia and Galicia in Spain, and the north of Italy. Regions of high N fertiliser input can be identified in most intensive agricultural areas in Europe, again including Denmark, Belgium, the Netherlands, UK and Ireland, Brittany (France) and the Po Valley (Italy).

Results show that an exceedance of the N manure input of 170 kg N occurs mainly in various dense livestock population areas, such as the Netherlands, where even the derogation of 250 kg N is often exceeded in the year 2000. There is a

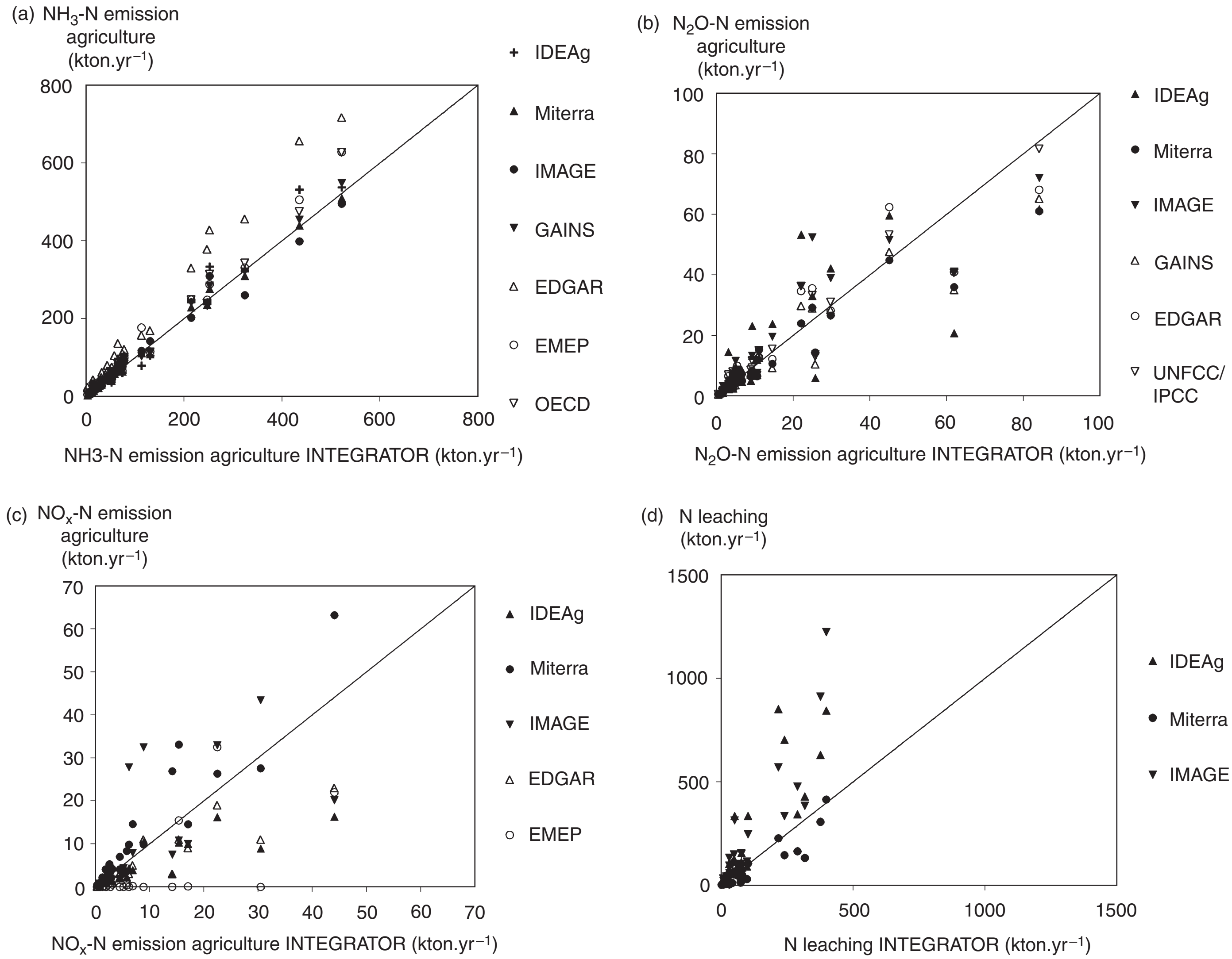

Figure 15.3 A comparison of country emissions for NH_3-N, N_2O-N and NO-N and of the sum of N leaching and runoff for the year 2000 within EU 27 as derived with INTEGRATOR and with various other model approaches (IDEAg, MITERRA, IMAGE, GAINS, EDGAR, EMEP and UNFCC/IPCC).

clear difference between IDEAg and INTEGRATOR in western France, where the latter model calculates much higher N manure inputs. The reason for this difference is seemingly a different disaggregation of animal numbers. In general N application by mineral fertiliser is higher in IDEAg, specifically in Western Europe, but also in the Nordic countries where it is possibly an artefact due to division of N inputs by very small areas of agricultural land (Figure 15.4a, b). Inversely, N application by animal manure, including grazing, is generally higher in INTEGRATOR, except for parts of the Netherlands and Denmark. INTEGRATOR shows hot-spots, e.g. in parts of France and Eastern Europe that are not resulting from IDEAg (Figure 15.4c, d). A comparable picture for the estimated N inputs by mineral fertilisers and animal manure for the year 2000 in EU25 is given by Grizzetti *et al.* (2007), using a 10 km × 10 km resolution. Details on the approach, combining agricultural statistics on administrative basis and geographic land cover information, are given in Grizzetti *et al.* (2007).

NH_3 and N_2O emissions

Calculations by both INTEGRATOR and IDEAg show that the regional variation in total NH_3 and N_2O emissions is large (Figure 15.5). Hot spots are located in areas with intensive animal husbandry in the Eastern and central part of Ireland, in England and Wales, in the Netherlands, Belgium, Denmark, in north-western and southern Germany, in the north of Italy and in the Catalonia region in Spain. In general, NH_3 emissions calculated by IDEAg are higher than by INTEGRATOR in Western and Central Europe, but the reverse is true for the Nordic countries (Figure 15.5a, b). Inversely, N_2O emissions calculated by IDEAg are higher everywhere, specifically in the Nordic countries, where the high emissions might be an artefact of the extremely high N fertiliser input but lower in the UK and Ireland (Figure 15.5c, d). The variation in NH_3 and N_2O emissions is in general comparable with the geographic variation in N surpluses, which in turn are strongly related to the variation in manure N inputs. The high correlation between

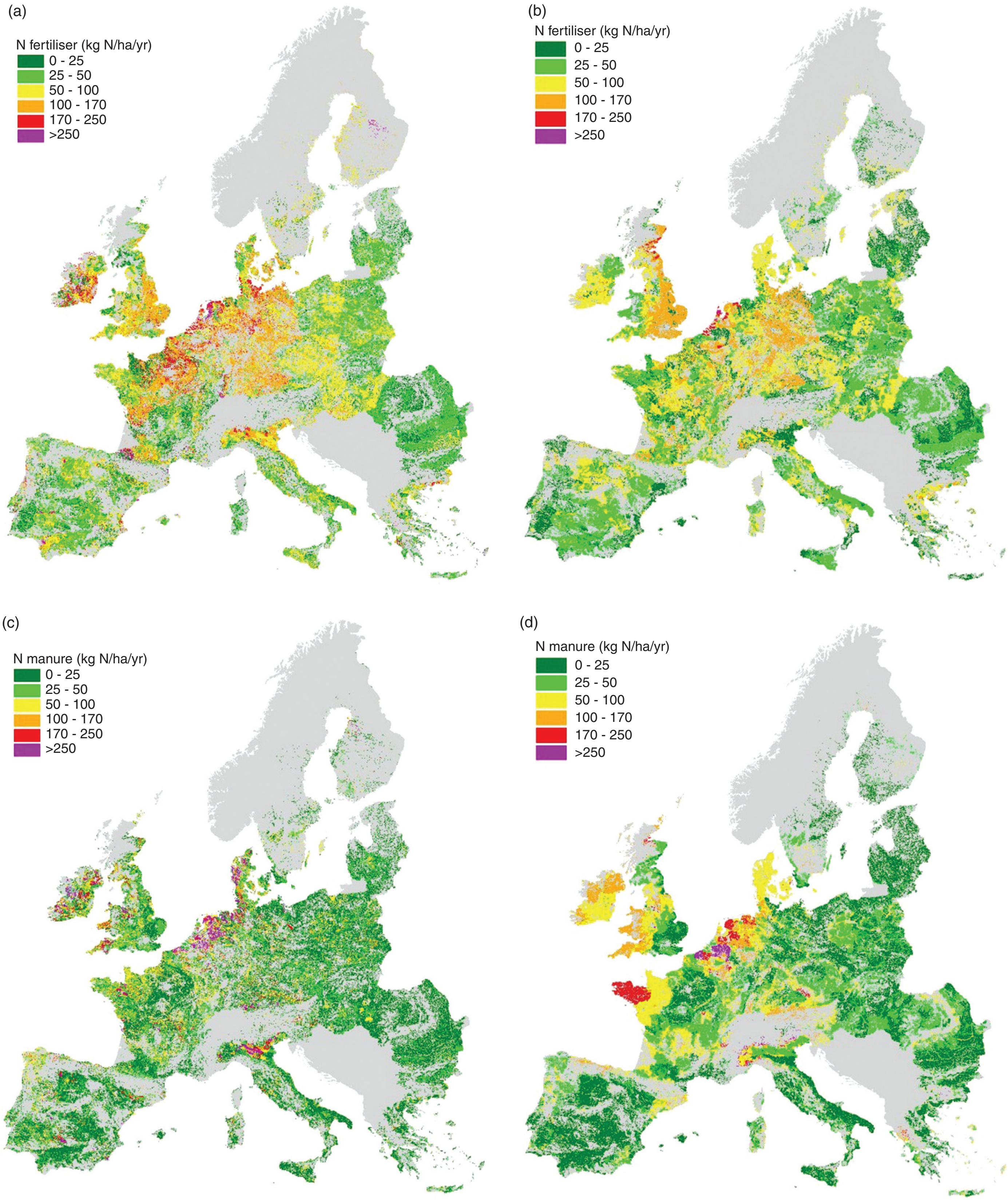

Figure 15.4 Nitrogen application from mineral fertiliser (a, b) and manure, including grazing (c, d) in the year 2000 in EU-27. Calculation with IDEAg on the geographic resolution of HSMUs (left) and with INTEGRATOR on the geographic resolution of NCUs (right). Grey shading in the EU-27 denotes non-agricultural areas. Countries outside EU-27 are also included by a grey shade.

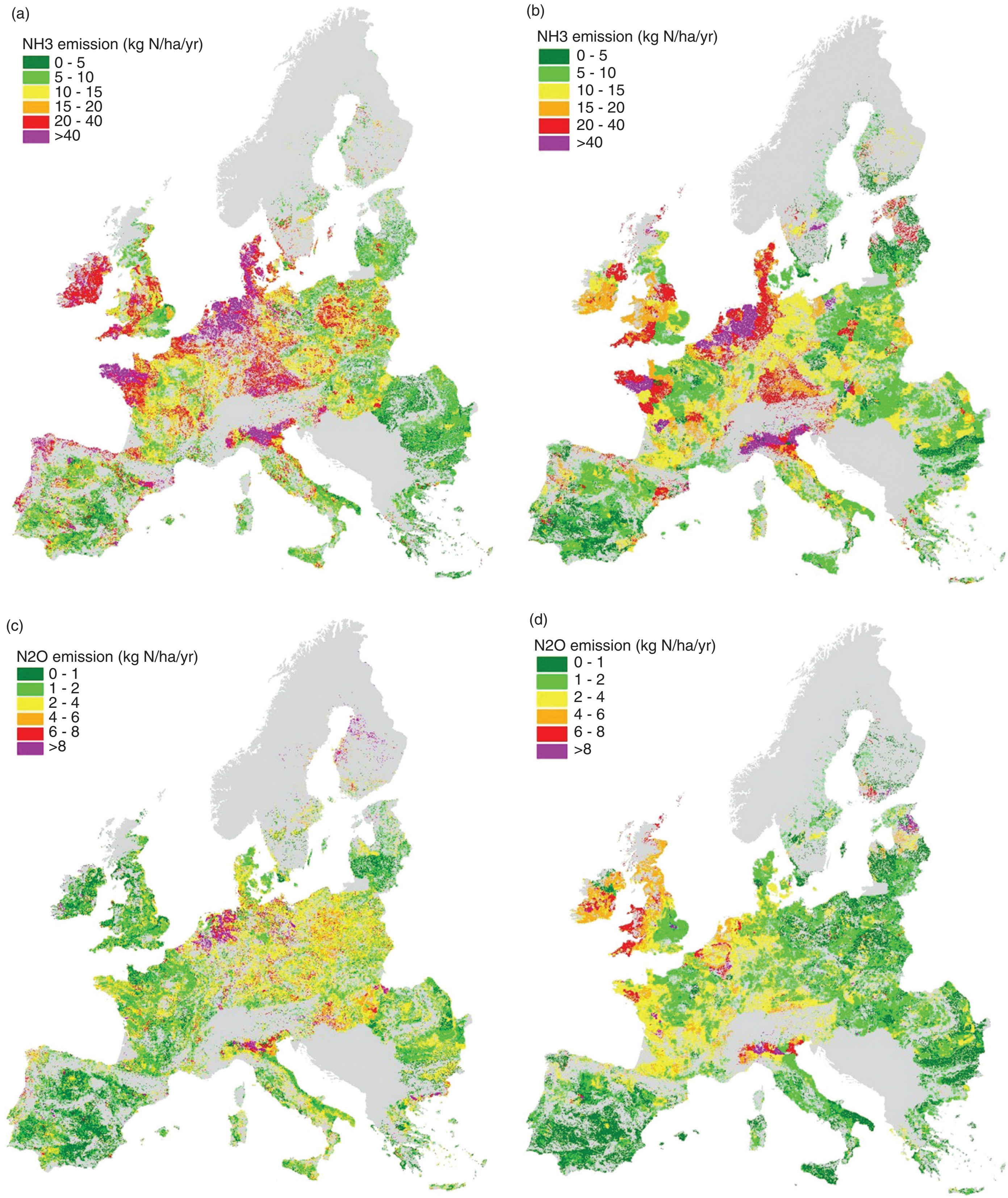

Figure 15.5 Total NH_3 emissions (a, b) and N_2O emissions (c, d) from agriculture in the year 2000 in EU-27. Calculation with IDEAg on the geographic resolution of HSMUs (left) and with INTEGRATOR on the geographic resolution of NCUs (right). Grey shading in the EU-27 denotes non-agricultural areas. Countries outside EU-27 are also included by a grey shade.

N surplus, being a main driver for N emissions and manure application is illustrated in detail by Leip *et al.* (2011b).

Nitrogen losses to ground water and surface water

Nitrogen losses to either ground water or surface water can be achieved using models, which include the major N inputs and the main processes of N transport and transformation, including surface runoff (overland flow) and runoff (interflow) to surface water and leaching to ground water. Various models have been developed and applied to address the issue of N fate in the river basin, and they vary for process description, scale of study and data requirement (http://euroharp.org). On a European wide scale, both detailed (EPIC) and simple process based models (INTEGRATOR, IDEAg) and statistical models (GREEN) are available (see Table 15.2). Here, we show results derived with both INTEGRATOR and IDEAg and with GREEN. The estimated regional variation N losses from soil to both ground water and surface water in 2000 as derived with IDEAg and INTEGRATOR is given in Figure 15.6. It should be emphasised that INTEGRATOR estimates are only slightly influenced by meteorological data, since the model uses N leaching fractions that depend on soil type, land use, soil organic content, precipitation surplus, temperature and rooting depth (Velthof *et al.*, 2009). In IDEAg, however, N leaching from soils is based on the DNDC-CAPRI meta-model (Britz and Leip, 2009a), which in turn is derived from CAPRI-DNDC model simulations using meteorological data to asses water fluxes and related N leaching fluxes. In this context, use is made of the JRC-MARS database, being a spatial interpolation of more than 1500 weather stations across Europe onto a 50 km × 50 km grid (Orlandi and Van der Goot, 2003).

In line with Table 15.4, results obtained by IDEAg show a much higher N leaching rate all over Europe, as compared to INTEGRATOR. Most likely, the N leaching by IDEAg is an overestimation, since there is a reasonable comparison between measured NO_3 concentrations in ground water and those estimated by the MITERRA model, being the agricultural module in INTEGRATOR in an adapted form (see Section 15.5.1 on model evaluation).

Figure 15.7 (left) shows an estimate of N diffuse losses to surface water for the year 2000 for Europe (Grizzetti *et al.*, 2008; Bouraoui *et al.*, 2009), based on the GREEN model taking into account N sources, river network and climate conditions. According to these estimates, the regions affected by higher N losses to surface waters include Belgium, the Netherlands, the Po Valley (Italy), the Brittany region (France), which are already totally or partially designated as Nitrates Vulnerable Zones (Nitrate Directive). Figure 15.7 (right) shows the estimated N source apportionment per sub-catchment for Europe for the year 2000. This map provides a picture of the relative contribution of diffuse sources (mainly agriculture) and point sources (mainly urban settlements) to the water N pollution. According to these estimates, agriculture is the main

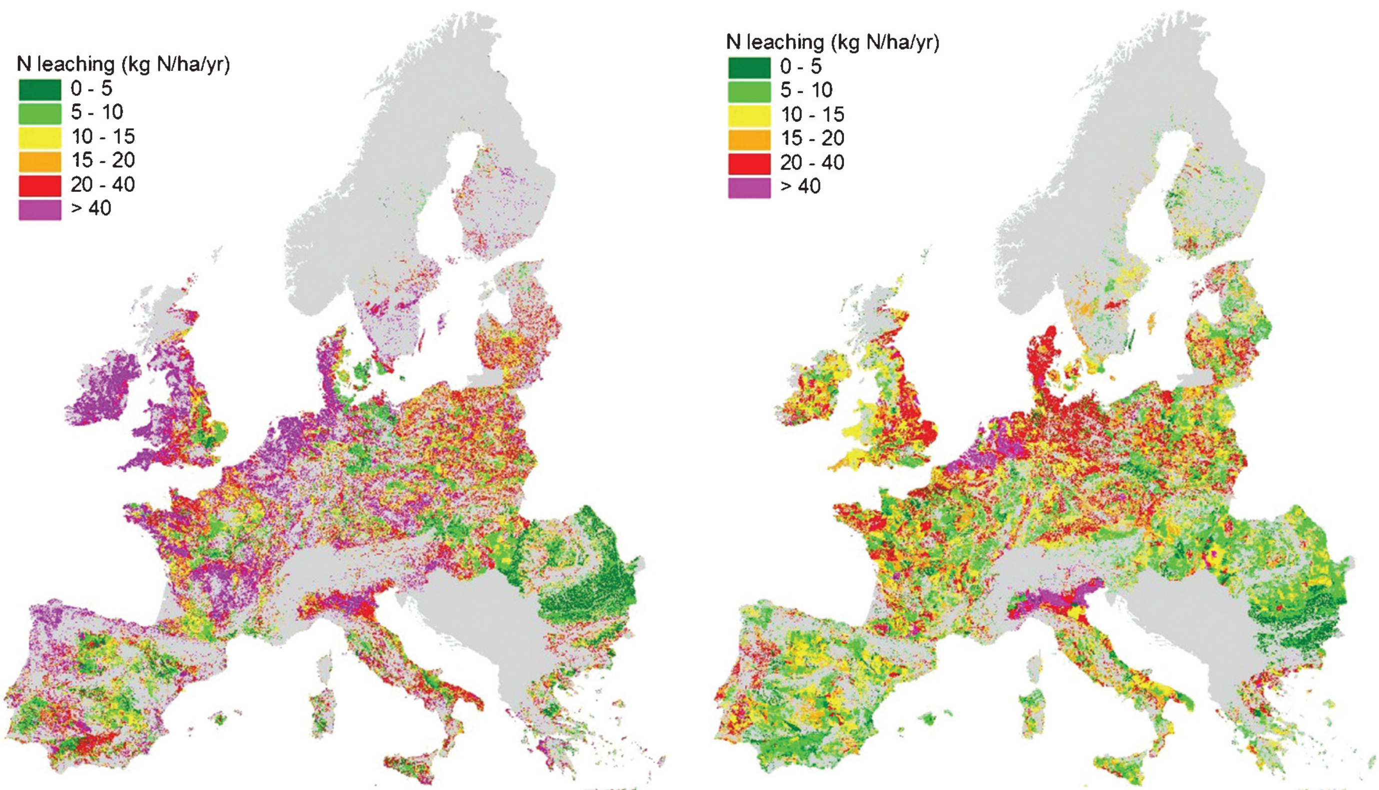

Figure 15.6 Regional pattern of N leaching plus runoff in the year 2000 in EU-27 based on calculations with IDEAg on the geographic resolution of HSMUs (left) and with INTEGRATOR on the geographic resolution of NCUs (right). Grey shading in the EU-27 denotes non-agricultural areas. Countries outside EU-27 are also included by a grey shade.

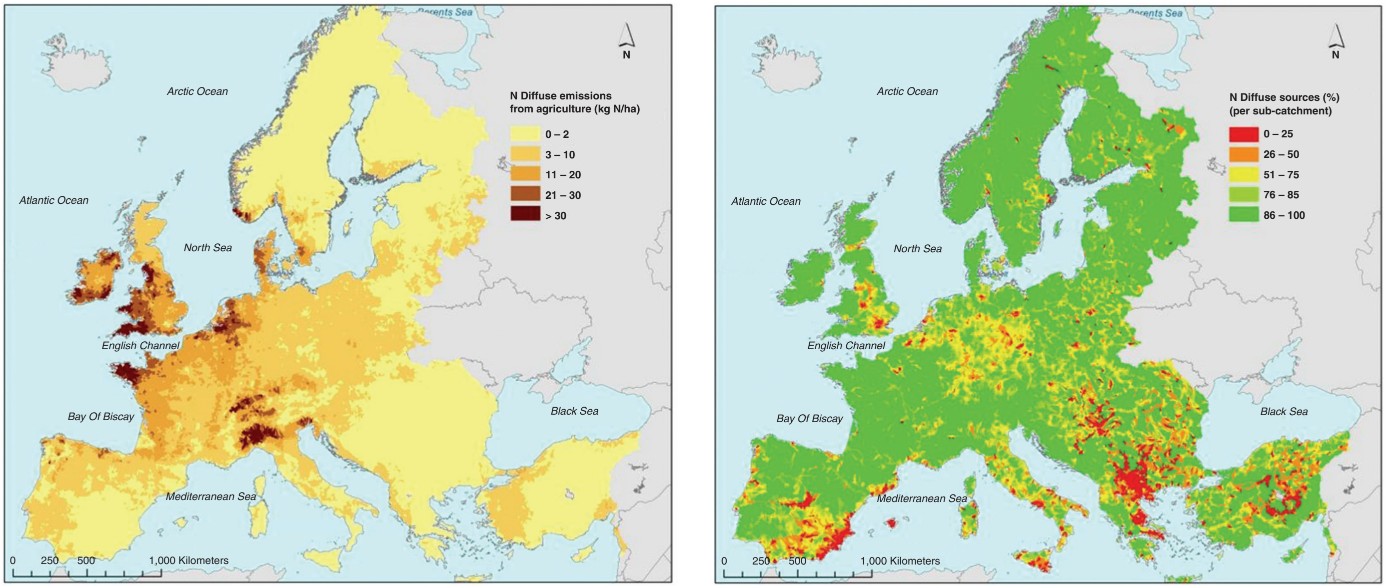

Figure 15.7 Regional pattern of N loads to surface water as diffuse emissions (left) and N source apportionment (right) for Europe in year 2000, based on calculations with the GREEN model on a geographic resolution of sub-catchments (average size 180 km^2).

contributor of N for surface waters in most of the river basins, while in Mediterranean catchments point sources have a relative higher contribution, which is probably due to a less effective implementation of waste water treatments and the lower precipitation and thus N losses to surface waters.

15.3.4 Trends in nitrogen fluxes since 1970

Trends in N fluxes since 1970 up to the year 2000 are derived on the basis of INTEGRATOR using the following.

- Data on N fertiliser use, animal numbers and crop yields from the FAO database.
- Scaled N excretion rates to those used for 2000 on the basis of RAINS/GAINS data. The scaling is based on a simple N excretion model described by Witzke and Oenema (2007), using the milk production as a scaling factor for dairy cattle and the meat production as a scaling factor for other cattle, pigs and poultry. Data on the milk and meat production per country in the period 1970–2000 were taken from the FAO database.
- N deposition history based on historical NO_x emissions by EMEP and NH_3 emissions by INTEGRATOR, while adding non-agricultural sources from IMAGE and using an emission-deposition matrix based on the EMEP model (EMEP, 2009).
- Constant N fixation rates for the grassland and arable land, but using FAO data on trends in the area of dry pulses and soy beans, mainly affecting N fixation. Information on trends in data for alfalfa and clovers, affecting the estimate for biological fixation by grasslands are missing and consequently we assumed no trends in N fixation rates by grassland.
- Scaled N contents in crops, based on a change in N availability (this is automatically calculated in INTEGRATOR).
- Trends in NH_3 emission factors in view of changes in housing systems and manure application techniques. For the year 2000, GAINS data are used for the fraction of housing systems and manure application techniques with high, medium and low emissions per country. For the period 1970–1980, we assumed that all emission fractions were high and in the period 1980–2000, we assumed a linear interpolation from high emissions to the present emission percentage.

Note that the available data on both crop yields and N fertiliser use in the FAO databases include trends in N use efficiency, which is mostly defined as the crop yield divided by the N input by fertiliser (Bouwman *et al.*, 2005).

Results derived by INTEGRATOR for the trends in all N inputs, N surplus and N outputs, in terms of N emissions to the atmosphere and N leaching to ground water and surface water, for the period 1970–2000 are given in Figure 15.8. The results show a steady increase of N inputs by fertilisers in the period 1970–1985, followed by a decrease since then, mainly in response to the increased or decreased crop production in those periods (or vice versa). Despite a slight decrease in cattle, the N input by manure excretion has increased up to 1985 due to an increase in N excretion rates, related to an increase in milk production, followed by a slight decrease in response to the decrease in livestock and the relatively constant excretion rates. The trend is also influenced by the increase in pigs anol poultry between 1970–2000 (see Oenema *et al.*, 2007), but the dominant effect is that of changes in N excretion rates by dairy cattle. There is a more clear increase in the average N input in agricultural systems than in the total N input, due to a decrease in agricultural area. This holds also for the trends in the total N uptake and the related N surplus for the period 1970–2000. Results show a slightly declining trend in NH_3 emission in response to a decline in livestock since 1990, but the trends in N_2O and NO_x emissions and N leaching are almost constant.

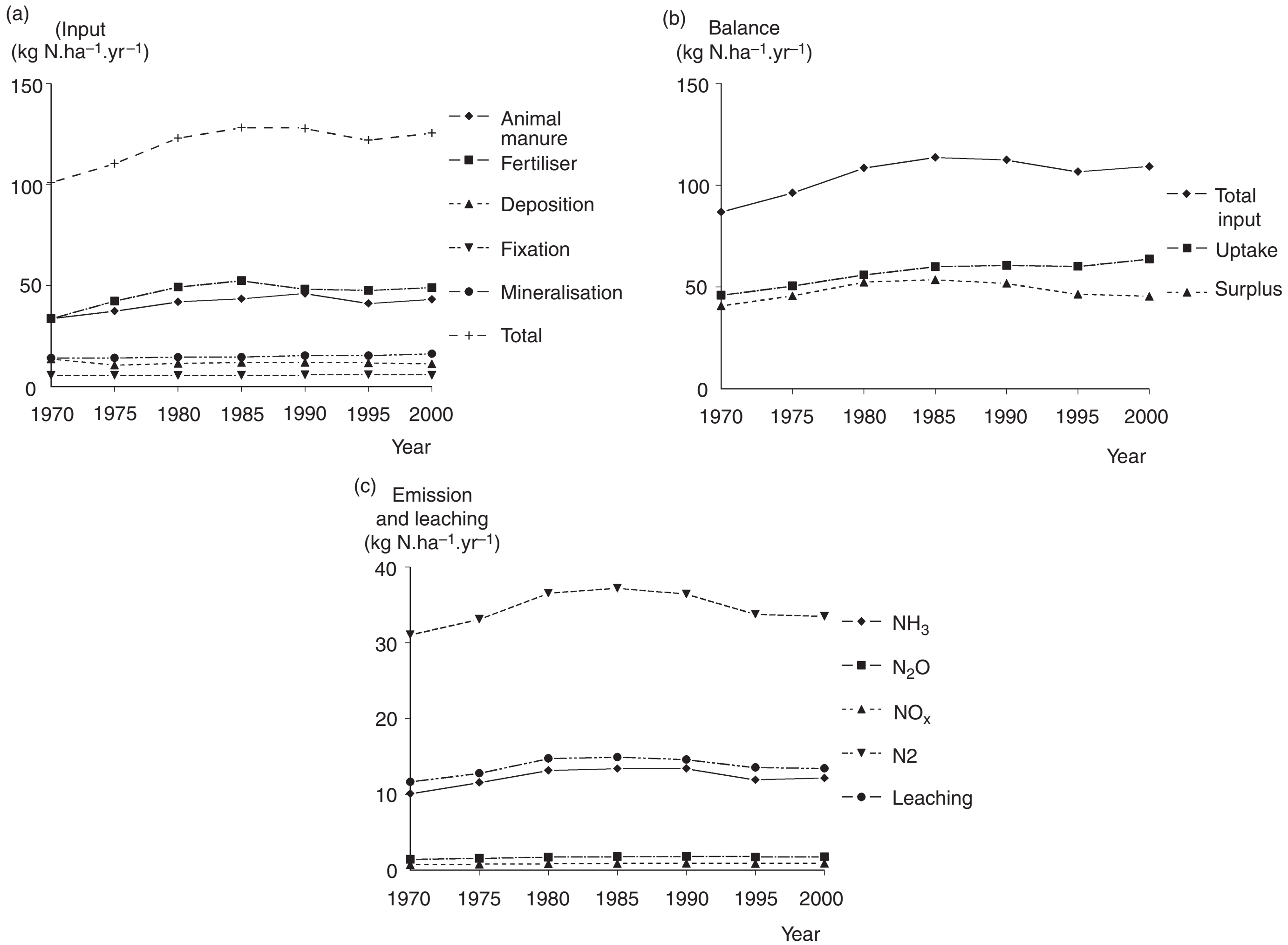

Figure 15.8 Trends in average N inputs (a), and average N surplus (b) and average N outputs (c) at EU-27 level for the period 1970–2000 as estimated by INTEGRATOR. NB: leaching stands for leaching plus runoff.

Trends in N_2 emissions, being most uncertain, are clearly increasing up to 1985 and declining afterwards (Figure 15.8).

15.4 Land nitrogen budgets for non-agricultural systems

15.4.1 Detailed land nitrogen budgets at European level

An overview of the land N budget for all terrestrial non-agricultural systems (forests and semi-natural vegetations) at the European scale (EU-27) as calculated with INTEGRATOR is given in Table 15.6. For non-agricultural systems, there is no differentiation between land and soil N budgets as all fluxes are related to the soil system. N deposition is derived with an emission deposition matrix, using NO_x and non-agricultural NH_3 emissions from EMEP and NH_3 emission estimates from agriculture by INTEGRATOR as inputs. The N manure input to semi-natural vegetations is mainly due to rough grazing, but it also includes some manure application being calculated in the MITERRA sub-model of INTEGRATOR. For forests, rough grazing is assumed to be negligible. Net N immobilisation (accumulation) in both forests and semi-natural vegetations is calculated as a fraction of the net N input, which is dependent on the C/N ratio of the soil, using an approach described in De Vries *et al.* (2006). NH_3 emissions in forests are background emissions due to wild animals derived from Simpson *et al.* (1999), whereas the NH_3 emission from short vegetations is calculated as a fraction of the N manure input by grazing animals. In forests, the estimated N_2O, NO and N_2 emissions by INTEGRATOR are derived with a statistical relationship with environmental factors based on results of a European wide application of the process oriented biogeochemical model Forest-DNDC (Li *et al.*, 2000) by Kesik *et al.* (2005). Apart from this meta-model of Forest-DNDC, INTEGRATOR includes an empirical relationship with various environmental factors, based on hundreds of measurements assessed in the literature (Bloemerts and de Vries, 2009). In short vegetations, the N_2O and NO emissions are calculated as a fraction of the N input, using emission factors that are a function of N source, soil type, pH, precipitation and temperature (see Supplementary materials Chapter 15 and 16). Finally, N leaching is assessed by multiplying the net N input by an N leaching factor and N_2 emissions

Table 15.6 The annual N budget of forest soils and semi-natural vegetation (EU 27) in Europe in 2000, as derived with INTEGRATOR.

	N budget (kton N yr^{-1})		
Source	**Forests**	**Semi-natural vegetations**	**Total nature**
Inputs			
Manure input (grazing)	—	1003	1003
Deposition	1367	345	1712
Biological N fixation	271	214	485
Total	1638	1562	3200
Outputs			0
Net uptake	302	779	1081
N accumulation	729	−26	703
Emissions of			0
NH_3	21	221	242
N_2O	45	37	82
NO_x	13	18	31
N_2	256	431	687
N leaching + runoff	272	113	385
Total	1638	1572	3210

are then calculated as N input minus all N output terms. In forests, N_2 emission is already calculated and N leaching is calculated as all N input minus all N output terms.

The results show that while the total N input is comparable in forests and semi-natural vegetations, N deposition dominates the N input in forests, whereas manure input by grazing animals dominates the N input in semi-natural vegetations. This high manure input also causes a much larger NH_3 emission in semi-natural vegetations as compared to forests. Compared to semi-natural vegetations, net N uptake and N_2 emissions are lower in forests, whereas N accumulation (net N immobilisation) and N leaching are higher. In semi-natural vegetation, net N growth uptake is set equal to N excretion by grazing animals, since these animals continually remove the vegetation, but also excrete nearly the same amount on the field. In percentage of the N surplus (N input minus N uptake), the N leaching and runoff is approximately 20% from forests and 8% from semi-natural vegetations, being (much) lower than the default IPCC factor of 30%.

15.4.2 Nitrogen budgets at country level and regional level

N budgets calculated at country level

An overview of the N budget for forests for the EU-27 countries, based on INTEGRATOR results, is given in Table 15.7. In this table, removal refers to the net N removal due to wood harvesting and accumulation stands for the N pool change in the soil. Results show large variations in all N fluxes, related partly to the size of the country. There is also a large uncertainty in the N flax, specifically in the N_2O and NO_x emissions, as discussed below by comparing results of various model approaches.

N_2O emissions and NO emissions at country level and regional level

A comparison of the results per country by the original Forest-DNDC model with those obtained by the meta-model in INTEGRATOR is presented in Figure 15.9. For regionalisation purposes, Forest-DNDC was coupled to GIS with a resolution of 50 km by 50 km holding all relevant information for initialising (soil and forest stand properties) and driving the model (atmospheric input, daily meteorological data). Before application of Forest-DNDC on a regional scale, the model was evaluated for its suitability by applying it to different field sites of the NOFRETE project, which were well distributed across Europe. For further details, we refer to Kesik *et al.* (2005). Results of INTEGRATOR are based on the application of meta-models for N_2O and NO from DNDC at NCU level, while making checks on the N balance. We checked whether the N input by deposition and fixation, minus the net N uptake by trees, minus the calculated total N emission and N immobilisation is above a minimum N leaching rate (near zero kg N). If this is not the case, both N emission and N immobilisation are reduced, assuming that these terms are more uncertain than the estimated N deposition and N uptake. Only in cases where zero N emission and N immobilisation still leads to a leaching rate below the minimum value, the N fixation is increased. The rationale behind this check is that in low N input systems, where trees take all the N to maintain growth, there is not enough N available for N emissions, unless there is net N mineralisation (e.g. drained forest on peat soils).

The results with the meta-model for N_2O are quite comparable with the original DNDC model (Figure 15.9), except for two countries (Sweden and Finland), where the original DNDC model predicted an N_2O emission of 11.9 and 10.5 kton N yr^{-1}, whereas the meta-model predicted an N_2O emission of 0.7 and 2.3 kton N yr^{-1}, respectively. This large difference is due to the check on the N balance. In these Nordic countries with low N inputs, N is simply not available for large N_2O emissions. The results with the meta-model for NO_x are generally lower than the original model and this holds again specifically for Sweden and Finland but also for other countries such as Germany and France. Apart from the N balance checks, the differences are also due to the large dependence of the NO_x emissions on soil properties, such as pH, being differently used in the INTEGRATOR meta-model application that in the original DNDC model.

Regional patterns of the N_2O and NO emissions for forests calculated with INTEGRATOR are presented in Figure 15.10. Regional patterns obtained with Forest-DNDC are presented in the supporting material in Chapters 15 and 16, showing higher N_2O and NO emissions calculated by Forest-DNDC, as compared to INTEGRATOR, in the Nordic countries for reasons given above.

Table 15.7 N budgets calculated at country level for forests with INTEGRATOR for the year 2000

Country	Area (Mha)	N input fluxes (kg N ha^{-1} yr^{-1})			N output fluxes (kg N ha^{-1} yr^{-1})						
		Deposition	Fixation	Total	Removal	Accumulation	Emission NH_3	Emission N_2O	Emission NO_x	Emission N_2	Leaching + Runoff
Austria	3.698	16.5	2.0	18.5	4.3	8.1	0.12	0.03	0.01	2.5	3.5
Belgium	0.611	21.5	2.0	23.5	4.1	11.5	0.37	0.31	0.23	1.9	5.1
Bulgaria	3.418	9.4	2.0	11.4	0.4	5.4	0.26	0.18	0.00	2.8	2.3
Czech. Rep.	2.549	18.0	2.0	20.0	3.2	10.9	0.20	0.02	0.01	0.9	4.7
Denmark	10.261	0.5	0.1	0.5	0.1	0.3	0.03	0.01	0.00	0.0	0.1
Estonia	0.358	41.8	11.4	53.1	13.4	9.0	0.21	1.48	0.21	8.4	20.4
Finland	2.036	37.3	19.3	56.5	13.5	22.4	0.54	1.91	0.19	8.3	9.6
France	11.243	15.2	2.6	17.8	3.4	7.5	0.19	0.54	0.05	3.0	3.2
Germany	19.606	11.7	1.0	12.7	1.6	6.7	0.24	0.13	0.14	0.8	3.1
Greece	14.636	1.7	0.5	2.2	0.1	1.2	0.02	0.06	0.00	0.6	0.2
Hungary	3.369	6.0	1.0	7.0	1.2	2.8	0.11	0.18	0.01	1.3	1.4
Ireland	1.717	1.4	0.3	1.8	0.4	0.2	0.01	0.03	0.00	0.2	0.9
Italy	0.300	433.6	54.7	488.3	121.7	251.2	7.05	6.79	0.47	68.8	32.3
Latvia	8.195	2.7	0.6	3.4	0.8	1.2	0.00	0.14	0.06	0.4	0.8
Lithuania	1.804	9.9	2.0	11.9	1.6	5.1	0.00	0.54	0.27	1.4	3.0
Luxembourg	0.091	20.5	2.0	22.5	5.1	9.9	0.13	0.39	0.09	2.6	4.2
Netherlands	2.658	3.5	0.2	3.7	0.4	2.1	0.05	0.08	0.14	0.0	1.0
Poland	0.306	424.3	59.6	483.8	40.8	248.5	5.97	12.61	10.57	36.2	129.2
Portugal	9.117	1.6	0.6	2.2	0.0	1.3	0.07	0.23	0.21	0.2	0.2
Romania	2.583	28.0	5.2	33.2	9.7	10.4	0.36	1.13	0.02	7.1	4.5
Slovakia	6.757	4.1	0.6	4.7	0.8	2.3	0.02	0.03	0.00	0.5	1.0
Slovenia	25.247	0.8	0.1	0.8	0.1	0.4	0.00	0.00	0.00	0.1	0.2
Spain	1.115	66.8	20.2	87.0	26.9	23.6	1.12	4.40	0.25	27.4	3.2
Sweden	1.895	69.8	26.7	96.4	20.9	39.9	0.93	5.38	1.09	26.2	2.0
United Kingdom	1.773	10.5	2.0	12.5	2.2	3.2	0.35	0.13	0.01	3.0	3.6
EU-27	135.342	10.1	2.0	12.1	2.2	5.4	0.15	0.33	0.10	1.9	2.0
EU-27[a]	135.342	1367	271	1638	302	729	21	45	13	256	272

[a] Data given in kton N yr^{-1}.

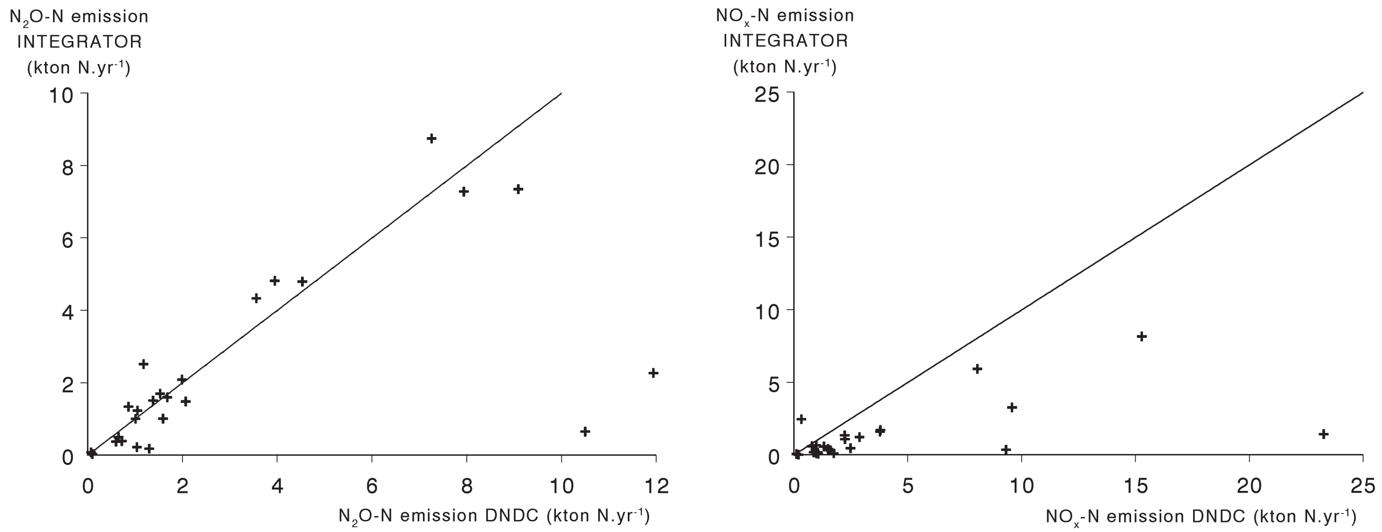

Figure 15.9 A comparison of country emissions for N_2O emissions (left) and NO_x emissions (right) from forests in EU 27 for the year 2000, calculated by DNDC, as estimated by Kesik *et al.* (2005), and calculated with INTEGRATOR.

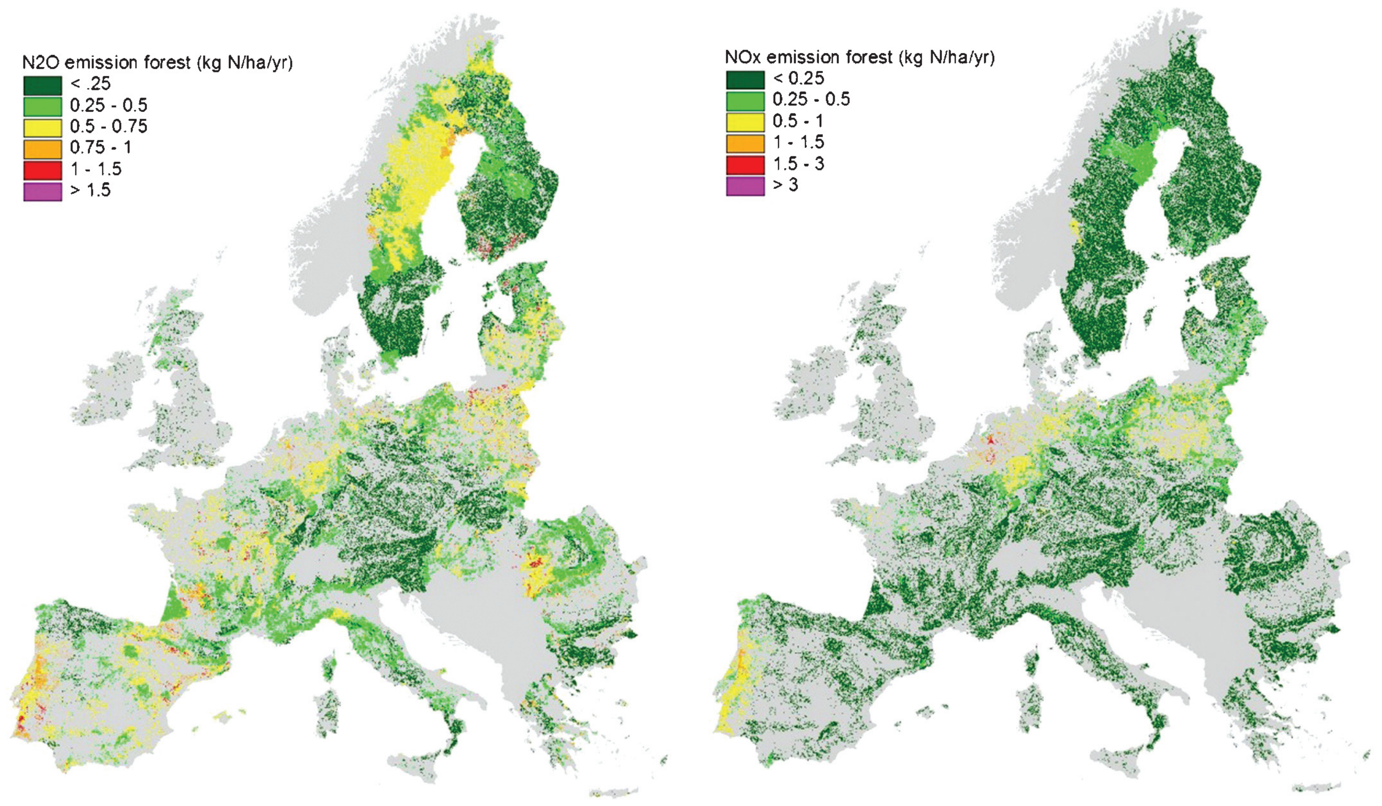

Figure 15.10 Regional pattern for the emissions of N_2O (left) and NO (right) from forest soils in EU 27 in the year 2000 as derived with INTEGRATOR. Grey shading in the EU27 denotes non-agricultural areas. Countries outside EU27 are also included by a grey shade.

Nitrogen losses to ground water and surface water

The geographic variation in estimated NO_3-N leaching and runoff from forest soils and short vegetations (with rough grazing) in 2000, as derived with INTEGRATOR, is shown in Figure 15.11. In line with the high N deposition inputs, N leaching below forests is high in the Netherlands and Germany and low in the Nordic countries and in Spain. In the Nordic countries, N leaching does not reflect the N deposition pattern, mainly due to impacts of temperature. In the north, growth is very limited owing to low temperatures, this leading to extremely low N uptake rates. N leaching from semi-natural vegetations reflects the high N manure input regions due to rough grazing, mainly occurring in western UK and central Europe.

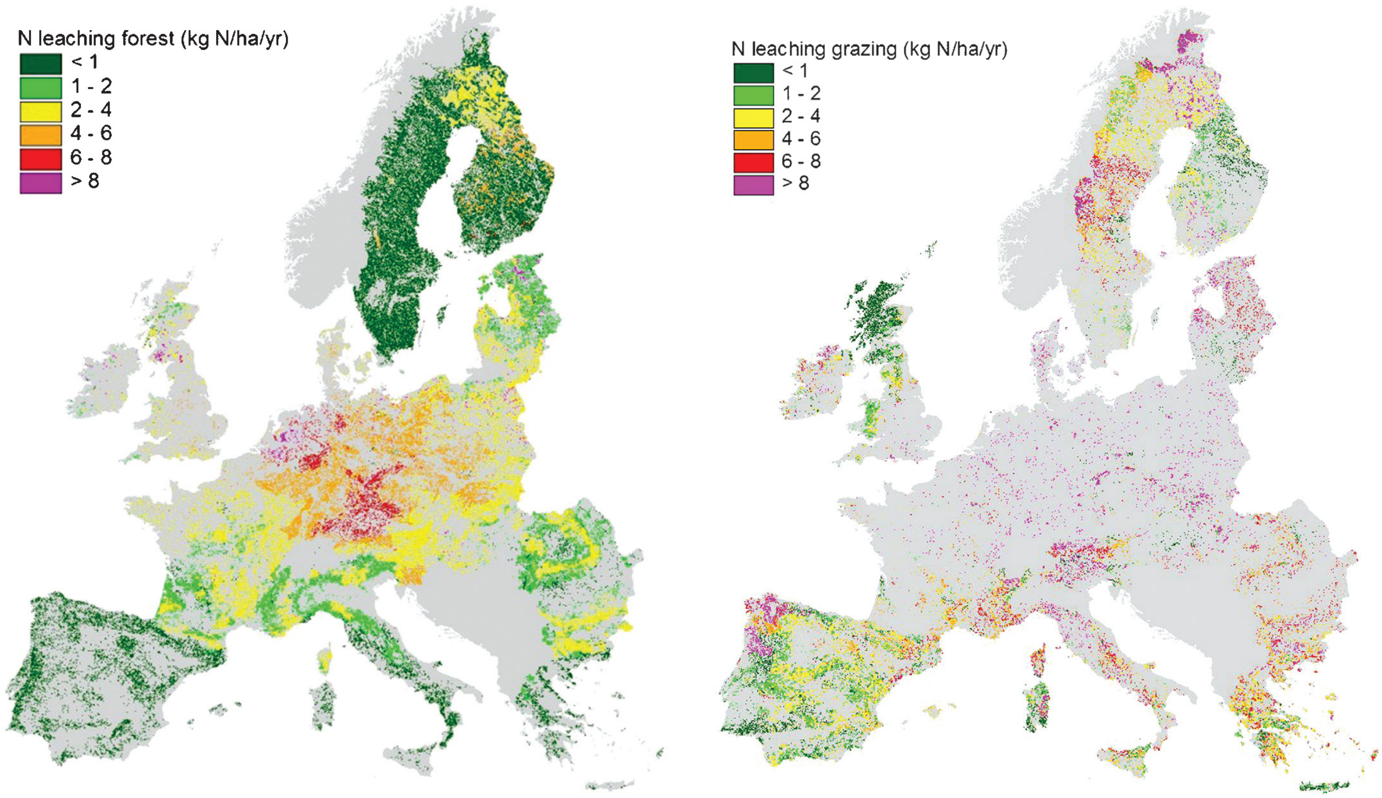

Figure 15.11 Regional pattern of N leaching and surface runoff to ground water and surface water (left) and semi-natural vegetation (right) from forest soils in EU-27 in the year 2000 as estimated by INTEGRATOR. Grey shading in the EU-27 denotes non-agricultural areas. Countries outside EU-27 are also included by a grey shade.

15.5 Discussion and conclusions

15.5.1 Model evaluation

Comparability of model results

In general, results of various N budget models (INTEGRATOR, IDEAg, MITERRA and IMAGE) in terms of annual N inputs and N fluxes on a European (EU27) wide scale are reasonably comparable for the year 2000. This holds specifically for N fertiliser inputs that are all based on the same source and to a lesser extent for N manure input where livestock sources are mostly comparable, but where N excretion rates differ. Despite the overall comparability, the estimated geographic variation in N inputs differs considerably between models.

A comparison of agricultural emissions of NH_3-N, N_2O-N and NO_x-N for all the 27 EU countries as derived with the four complete N budget models and standard activity data-emission factors approaches (UNFCC/IPCC, GAINS, OECD, EDGAR and EMEP) also shows comparable estimates for NH_3. The differences in N_2O emissions, however, are much larger, while NO_x emissions are most uncertain. This holds both on a European wide scale and with respect to the geographic variation in the emissions.

Very uncertain are also the N leaching and runoff estimates, which show a very large deviation between models. This holds both for the European wide estimates and for the geographic variation. Most uncertain are also N_2 emissions that are often calculated as a rest term from all other N inputs and outputs in a budget approach. It is important to mention that this seemingly simple compound is almost not measurable and model results are quite speculative as they cannot be validated. The N_2 release can be derived from radioactive labelling and there are only a handful of studies focusing on N_2 measurements. In view of a complete N budget, it would be worthwhile to put more emphasis on the measurement of N_2.

Comparison of results with inverse modelling results for nitrous oxide emissions

Inverse modelling is an important tool for regional emission estimates and independent verification of international agreements on emission reductions, such as the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol (IPCC, 2001; Bergamaschi, 2007). Atmospheric measurements combined with inverse atmospheric models can provide independent 'top-down' emission estimates of atmospheric trace gases. Inverse modelling has been widely applied for CO_2 and CH_4 (IPCC, 2007), while only relatively few studies are available for N_2O. The first inverse analysis of the global N_2O cycle was presented by Prinn *et al.* (1990) based on a 9-box model and atmospheric observations from the ALE-GAGE network for 1978–1988. They concluded that beside the use of fertiliser and fossil fuel combustion in mid latitudes, tropical sources (probably from tropical land use change) are likely to play an important role for the global budget and the observed N_2O increase (32%–39% for 1978–1988). The more recent studies of Hirsch *et al.* (2006)

Table 15.8 N_2O emissions for the year 2000 for Ireland and UK and for Western Europe as derived with INTEGRATOR and based on results by the ^{222}Rn tracer method and the inverse model NAME (after Messager *et al.*, 2008)

	N_2O emissions (kg N_2O-N ha^{-1} yr^{-1})			
			INTEGRATOR	
Area	^{222}Rn Tracer method	Inverse model NAME	Agriculture	Total[b]
Ireland + UK	8.3–9.8	9.0–11.1	6.8	11.2
Western Europe[a]	6.6–8.9	7.5–10.2	4.7	7.7

[a] The sum of emissions from France, Germany, the Netherlands, Belgium, Luxembourg and United Kingdom.
[b] Total emissions by INTEGRATOR were derived by multiplying the agricultural N_2O emissions with the ratio of total/agricultural N_2O emissions based on GAINS.

and Huang *et al.* (2008), based on 3D global inverse models suggest an even larger contribution of the tropical sources between 0 and 30ºN.

First inverse modelling estimates of European N_2O emissions were provided by Ryall *et al.* (2001) and Manning *et al.* (2003), using N_2O observations from Mace Head and the NAME Lagrangian particle model. Their estimates for North West European countries showed an agreement within ~30% or better with emissions reported to UNFCCC. Another example is downscaled emissions for parts of Europe based on the NAME model and a model-independent approach using the ^{222}Rn tracer method, presented by Messager *et al.* (2008). A comparison of N_2O emissions derived by INTEGRATOR with those estimates is given in Table 15.8.

Results show that the comparison is reasonable. It needs to be emphasised, however, that top-down approaches generally estimate total emissions, while emission reported to UNFCCC cover only anthropogenic emissions. Hence, for quantitative comparisons good bottom-up estimates of the natural sources are needed.

While the above European top-down emission estimates are based on one single station only (Mace Head), improved emission estimates require the use of further atmospheric measurements, to provide a better coverage of the European domain. Additional continuous N_2O measurements are now available from the European RTD project CHIOTTO ('Continuous HIgh-precisiOn Tall Tower Observations of greenhouse gases') for 2006, which has set up a European network of tall towers for GHG measurements. The measurements from the CHIOTTO towers and further monitoring stations are currently used in the NitroEurope project to provide European N_2O emission estimates using five independent inverse models. A particular challenge constitutes the fact that measurements from different stations / networks may have small calibration offsets, hence requiring sophisticated bias correction procedures in the inverse modelling systems. Results from the NitroEurope inverse modelling will be available early 2011.

There are also great opportunities for constraining NH_3 or NO_x emissions by independent datasets based on wet concentration measurements and satellite measurement (Gilliland *et al.*, 2003; Konovalov *et al.*, 2010). Whenever such datasets come available, they will be used for independent model validation.

Comparison of results with measurements for nitrate concentrations in ground water and N concentrations in surface water

Use was made of data on NO_3 concentration measurements in groundwater in the period 2000–2003 (EC, 2007) to validate the results of the MITERRA-Europe model. The measurements of NO_3 concentration showed that 17% of EU-27 monitoring stations had NO_3 concentrations above 50 mg NO_3 l^{-1}, 22% were in the range of 25 to 50 mg NO_3 l^{-1} and 61% of the groundwater stations had a concentration below 25 mg NO_3 l^{-1} (EC, 2007). A preliminary validation of the MITERRA model on these NO_3 concentration measurements (Velthof *et al.*, 2009) showed that the distribution of calculated mean NO_3 concentrations in NUTS2 regions of EU-27 according to MITERRA-EUROPE agrees very well with the distribution of the means of measured NO_3 concentrations in the EU-27. For the year 2000, MITERRA estimates that 16% of the NUTS2 regions had NO_3 concentrations above 50 mg NO_3 l^{-1}, 20% were in the range of 25 to 50 mg NO_3 l^{-1}, and 65% had a concentration below 25 mg NO_3 l^{-1}. The calculated NO_3 concentrations were also in the same range of the means of measured NO_3 concentrations in groundwater bodies. For Belgium, Czech Republic, Denmark, the Netherlands and Poland, the calculated NO_3 concentrations appear somewhat higher than the measured NO_3 concentrations. Possible reasons for these apparent differences are that monitoring stations measure NO_3 concentrations at various depths, while MITERRA-EUROPE estimates NO_3 concentration in the soil water at uniform depth (below rooting zone). Moreover, monitoring stations may include forests and natural land, whereas MITERRA-EUROPE only calculates NO_3 concentration for agricultural land. Finally, it has to be realised that the model results refer to the NO_3 concentration in leachate to ground water and not to the concentrations in ground water as measured in the ground water stations.

15.5.2 Nitrogen budgets and effects on ecosystems

There is an increasing demand by policy makers for easy to interpret and understand indicators that assess the environmental performance and 'sustainability' of agriculture. Results presented before thus need to be interpreted in view of possible

Table 15.9 Variation in number of countries with estimated NH_3-N emissions in 2000 exceeding the National emission ceilings for 2010, depending on the model approach

Model	Number of countries exceeded	Percentage countries exceeded[a]	Total exceedance kton NH_3-N yr^{-1}
INTEGRATOR	7	28	103
IDEAg	7	28	264
MITERRA	9	36	75
IMAGE	7	29	167
GAINS	10	40	109
EDGAR	18	72	1269
EMEP	14	56	261
OECD	12	63	245

[a] The countries included in the calculation were 25 (EU27 minus Cyprus and Malta) for INTEGRATOR, MITERRA, GAINS, EDGAR and EMEP, 24 for IDEAg and IMAGE and 19 for OECD. The percentage equals the number exceeded divided by these totals.

effects to be of use in policy making. Below, we discuss various options for performance indicators, based on either gross or detailed N budget approaches, including the exceedance of the following.

- Maximum N manure inputs and NH_3 emission ceilings. Note that these are policy criteria based on impacts but not critical levels related to actual impacts.
- Critical NH_3 concentrations and critical N loads in view of biodiversity impacts and in view of elevated N saturation of forest soils, associated with damage by plagues and diseases.
- Critical NO_3 concentrations in ground water in view of health effects and critical N concentrations in surface waters in view of eutrophication of terrestrial ecosystems.

The assessment is focused on the year 2000. Trends in the changes of risks can be derived from the trends in N fluxes since 1970, as presented earlier.

Nitrogen surpluses and manure nitrogen inputs as performance indicators

In the Pan European initiative, SEBI2010, which stands for Streamlining European 2010 Biodiversity Indicators, the agricultural N balance (implying the N surplus) is one of the 26 indicators that are developed to monitor progress towards the European target to halt the loss of biodiversity by 2010 (see: http://biodiversity-chm.eea.europa.eu/information/indicator/F1090245995). The N surplus is, however, a typical pressure indicator and not an effect indicator, since agro-ecosystems and environment both have a strong impact on the actual N emissions to the atmosphere and the N (NH_4 and NO_3) concentrations in leaching and runoff water, being relevant for the effects that may occur. For example, ammonia losses from agriculture are associated predominantly with animal production systems. Nitrate concentrations in the leachate to groundwater depend not only on N balance (N surplus) but also on climate (excess rainfall which dilutes the concentration), and soil type, affecting denitrification. As a result, the relationship between N surplus and N fluxes to the air and to water is diffuse.

Because of this complexity and variability, there are very few common and accepted reference levels against which to evaluate nutrient surpluses. In the Netherlands, the regulatory policy instrument MINAS has been used in the past in which reference values for N surpluses have been set tentatively at 60 and 100 kg per ha for arable land on sandy soils and clayey soils, respectively, and at 140 and 180 kg per ha for grassland on sandy soils and clayey soils, respectively.

At present, N surplus is not used as a performance indicator in policy making. Instead, use is made of a maximum N application rate by animal manure of 170 kg N with exceptions (so-called derogations) of 250 kg N for the Netherlands and 230 kg N for Denmark, Germany and Austria (after the year 2000). Maps of the N input by animal manure for the year 2000 (Figure 15.4) indicate that there still exist a number of areas in Europe where this limit is exceeded.

Ammonia emission and related ammonia concentrations and nitrogen deposition as performance indicators

The variation in NH_3 emissions will affect the N deposition on terrestrial ecosystems. Plant species diversity of terrestrial ecosystems is affected largely by N deposition and in this context empirical and model based critical N loads have been derived. Specifically in intensive livestock areas with high NH_3 emissions, the resulting N deposition may lead to an exceedance of critical N loads. In this context, national emission ceilings (NEC) have been set. A comparison of NECs for 2010 (EEA, 2010) and results of total NH_3 emissions by the various models described in this chapter is given in Table 15.9. For INTEGRATOR, IDEAg, MITERRA, and IMAGE, the estimated agricultural NH_3 emissions per country were multiplied by a factor 1.07, since approximately 7% of the NH_3 emissions come from non-agricultural sources. The number of countries with estimated NH_3-N emissions in 2000 exceeding the National emission ceilings for 2010 depends on the model approach and varied between 7 and 18, while the total exceedance varied between 75 and 1269 kton NH_3-N yr^{-1}. The large exceedances derived by EDGAR are clearly deviating from all other model approaches. The lowest emission exceedances

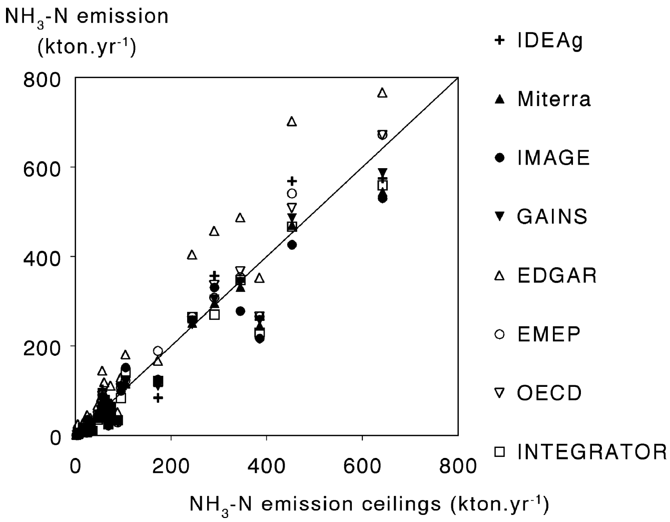

Figure 15.12 A comparison of the estimated national emissions and national emission ceilings for NH_3, derived with INTEGRATOR and various other model approaches (IDEAg, MITERRA, IMAGE, GAINS, EDGAR, EMEP and OECD/IPCC) for the year 2000.

are estimated by INTEGRATOR, MITERRA and GAINS, all being based on the same animal numbers and NH_3 emission factors.

The variation in NH_3-N emission exceedances, limited to those countries where all models calculate an exceedance is illustrated in Figure 15.12. For most countries, the exceedance is comparable, but for some countries the variation is considerable up to a fourfold variation.

Insight in the actual risk of elevated NH_3 emissions on terrestrial ecosystems can amongst others be derived by comparing either the actual NH_3 concentration with a critical NH_3 concentration in view of plant species diversity impacts. Recently updated critical levels are 1 $\mu g.m^{-3}$ for lichens and bryophytes and 3 $\mu g.m^{-3}$ for herbaceous plants (Cape *et al.*, 2009). A comparison of EMEP model predicted NH_3 concentrations with these critical levels during the last 15 years show that NH_3 concentrations violate the limit for lichens and bryophytes except for Fennoscandia and Scotland, as presented in Moldanová *et al.*, 2011 (Chapter 18, this volume). The limit for herbaceous plants is also exceeded in parts of Western Europe and Northern Italy.

Indirectly, insight in the actual risk of elevated NH_3 emissions on terrestrial ecosystems can also be derived by comparing present N depositions, which are largely determined by NH_3 emissions together with NO_x emissions, with the critical N deposition at the European scale. The critical N deposition is related to impacts on plant species diversity and is either derived from empirical field data or by model assessments, as discussed in Dise *et al.*, 2011 (Chapter 20 this volume). The exceedance of critical N loads in view of impacts on plant species diversity is one of the 26 performance indicators in SEBI 2010. A comparison of exceedances of critical N loads in 1980 and in 2010 is given in Dise *et al.*, 2011 (Chapter 20 this volume), showing that the N emission reductions in the past three decades has led to a significant reduction in the risk of N affecting plant species diversity, despite the limited emission reductions in NH_3 (see also Figure 15.8 lower graph for the period 1980–2000). This effect is specifically due to NO_x emission reductions in that period.

Nitrogen leaching and nitrogen runoff as performance indicators

Critical NO_3 concentrations in ground water in view of health effects and critical N concentrations in surface waters in view of eutrophication of aquatic ecosystems are also important targets to evaluate the N leaching and N runoff fluxes on a European wide scale. A critical NO_3 concentration in view of health impacts is set at 50 mg $NO_3\,l^{-1}$. Eutrophication is the result of nutrient (both N and P) enrichment in the aquatic system, but the severity of the phenomenon largely depends on the specific regional characteristics, climate, morphology, water residence time, nutrients ratio, tropic web status, and generally on the ecosystem resilience. Therefore, similar nutrient loads may produce different effects in reason of the regional sensitivities. Similarly, the impacts are related not only to N loads, but rather to its specific synergies with the availability of other elements, such as carbon, phosphorus and silica (see also Billen *et al.*, 2011; Grizzetti *et al.*, 2011, Chapters 13 and 17 this volume). Nevertheless, N concentrations in surface waters, being a major driving force of the problems, are used as a proxy to evaluate the risk for water eutrophication. A critical limit of 0.5–1.0 mg N l^{-1} has been proposed by Camargo and Alonso (2006) based on an extensive study on the ecological and toxicological effects of inorganic N pollution in aquatic ecosystems. At present, N concentrations are generally exceeding those limits (see also Grizzetti *et al.*, 2011, Chapter 17 this volume).

15.5.3 Conclusions and recommendations

Key findings regarding the temporal and geographic variation in N budgets in agricultural and other terrestrial ecosystems over Europe are as follows.

- Trends in N fluxes in agro-ecosystems since 1970 show an increase in N inputs by fertilisers and manure up to 1985, followed by a decrease since 1985 in response to a change in crop production and in animal numbers. Actually, livestock decreased since 1970, but in the period 1970–1985 the N input by manure excretion still increased due to an increase in milk production and related N excretion rates.
- For EU-27, the models estimates a total N input in European agriculture for the year 2000 of 23.3–25.7 Mton N yr^{-1} which is mainly due to fertiliser and animal manure inputs and to a lesser extent by atmospheric deposition and N fixation. Total N inputs at EU-27 level are comparable for all models, since they all use comparable basic data on fertiliser use and animal numbers. There exist a number of areas in Europe where a maximum N application rate by animal manure of 170 kg N is exceeded. The N uptake varies from 11.3–15.4 Mton N yr^{-1} leading to total N surpluses varying from 10.4–13.2 Mton N yr^{-1} at EU-27 level.

- The four complete N budget models for agro-ecosystems give in general very similar results for the emissions of NH_3 (2.8–3.1 Mton N yr^{-1}) and N_2O (0.33–0.43 Mton N yr^{-1}) but vary largely for NO_x (0.02–0.23 Mton N yr^{-1}). Similar results and differences are found when including standard activity data-emission factors approaches (UNFCC/IPCC, GAINS, OECD, EDGAR and EMEP).
- Even though NO_x emissions are more uncertain, the uncertainty in the NH_3 emissions is more important for the overall uncertainty in the reactive N budget, since NO_x contribute little to the overall N budget. The contribution of agriculture to total NO_x emissions is less than 5%, while the contribution of agricultural NH_3 emissions is more than 90%, making the variation in NH_3 emissions more important. The uncertainty is illustrated by the number of countries with estimated NH_3-N emissions in 2000 exceeding the National emission ceilings for 2010. Depending on the model approach, this number varies between 7 and 18, while the total exceedance varied between 75 and 1269 kton NH_3-N yr^{-1}.
- The estimated sum of N leaching and runoff at EU 27 is roughly equal to the sum of NH_3, N_2O and NO_x emissions to the atmosphere, but estimates vary by a factor two, from 2.7–6.3 Mton N yr^{-1}. This strongly affects the area with N concentrations exceeding critical N concentrations in surface water.
- In non-agricultural system (forests and semi-natural vegetation), the estimated total input is near 3.2 Mton N yr^{-1}, while the net N uptake is near 1.1 Mton N yr^{-1}, leading to a surplus near 2.1 Mton N yr^{-1}. Compared to agricultural systems, the estimated N fluxes in non-agricultural systems are about 5 times lower for N_2O emissions and 10 times lower for NO_x and NH_3 emissions and for the sum of N leaching and runoff.
- The regional variation in N fluxes is mainly determined by N inputs, being highest in areas with high livestock density and intensive agricultural crop production areas, while land/soil characteristics and climate are secondary factors influencing the magnitude of N fluxes.

Recommendations that can be made based on this assessment are as follows.

- Future research priorities should focus on major uncertainties, in particular N_2O emissions and N leaching and runoff from agricultural ecosystems. Furthermore, studies on denitrification are needed to reduce the large uncertainty in this process at the European scale.
- A database should be set up of N contents in various plants and in various regions to improve estimates of N uptake and N surplus at the European scale.
- Information on NH_3 concentrations in air should be used in inverse modelling approaches to derive independent datasets to validate the various NH_3 emission calculations.
- A European-wide monitoring network of ground- and surface water, using standardised methods and covering a range of habitats, should be initiated to provide consistent and reliable information on the long-term effects of air pollution on water quality, to be used for validation of N budget models.
- It is relevant that data use is harmonised for models predicting air emissions and N loss to waters for consistent environmental decision-making relevant to air quality, ecosystem deposition and water quality.

Acknowledgements

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Supplementary materials

Supplementary materials (as referenced in the chapter) are available online through both Cambridge University Press: www.cambridge.org/ena and the Nitrogen in Europe website: www.nine-esf.org/ena.

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