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Farm nitrogen balances in six European agricultural landscapes – a method for farming system assessment, emission hotspot identification, and mitigation measure evaluation

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Abstract

Six agricultural landscapes in Poland (PL), the Netherlands (NL), France (FR), Italy (IT), Scotland (UK) and Denmark (DK) were studied, and a common method was developed for undertaking farm inventories and the derivation of farm nitrogen (N) balances and N surplus from the in total 222 farms and 11 440 ha of farmland.

In all landscapes, a large variation in the farm N surplus was found, and thereby a large potential for reductions. The highest average N surpluses were found in the most livestock-intensive landscapes of IT, FR, and NL; on average 202 ± 28, 179 ± 63 and $178 \pm 20\,\mathrm{kg\,N\,ha^{-1}\,yr^{-1}}$, respectively. However, all landscapes showed hotspots, especially from livestock farms, including a special UK case with landless large-scale poultry farming. So, whereas the average N surplus from the land-based UK farms dominated by extensive sheep grazing was only $31 \pm 10\,\mathrm{kg\,N\,ha^{-1}\,yr^{-1}}$, the landscape average was similar to those of PL and DK (122 ± 20 and $146 \pm 55\,\mathrm{kg\,N\,ha^{-1}\,yr^{-1}}$, respectively) when landless poultry were included. However, the challenge remains how to account for indirect N surpluses and emissions from such farms with a large export of manure out of the landscape.

We conclude that farm N balances are a useful indicator for N losses and the potential for improving N management. Significant correlations to N surplus were found, both with ammonia air concentrations and nitrate levels in soils and groundwater, measured during the landscape data collection campaign from 2007–2009. This indicates that farm N surpluses may be used as an independent dataset for validation of measured and modelled N emissions in agricultural landscapes. However, no significant correlation was found to N measured in surface waters, probably because of the short time horizon of the study.

A case study of the development in N surplus from the landscape in DK from 1998–2008 showed a 22 % reduction, related to statistically significant effects ($p < 0.01$) of measures targeted at reducing N emissions from livestock farms. Based on the large differences between the average and the most modern and N-efficient farms, it was

concluded that N-surplus reductions of 25–50 % as compared to the present level were realistic in all landscapes. The implemented N-surplus method was thus effective at comparing and synthesizing results on farm N emissions and the potentials of mitigation options, and is recommended for use in combination with other methods for the assessment of landscape N emissions and farm N efficiency, including more detailed N sink and N source hotspot mapping, measurements and modelling.

1 Introduction

Nitrogen (N) is essential for agricultural production, but is also a key driver of environmental pollution, and can result in N concentrations in air and water exceeding critical limits for eutrophication (de Vries et al., 2011), significant greenhouse gas emissions (Alcamo and Olesen, 2012), biodiversity deterioration (Dise et al., 2011), and severe human health impacts (Brink and van Grinsven, 2011).

With agriculture responsible for most of the human-induced changes to the global N-cycle (Galloway et al., 2003), a global population increase of about 88 million people per year (United Nations Populations Fund, 2011), and a rapid growth in the global middle class with higher food consumption rates, an efficient, low N-surplus agricultural sector becomes increasingly important. Consequently, the balance between nitrogen input and output has been recognised as one of the key indicators for the development of sustainable agricultural systems (European Environmental Agency, 2005; OECD, 2008).

In the last few decades, the European Union has launched initiatives to mitigate the effects of N from agriculture, with a special focus on the most intensively farmed agricultural regions in Central and Western Europe (Oenema et al., 2011). The effectiveness of these N-mitigation measures, especially related to the National Emissions Ceilings Directive (2001/81/EC), the Nitrates Directive (1991/676/EC) and the Water Framework Directive (2000/60/EC), are undisputable (Kronvang et al., 2008; Hansen et al., 2011). However, there are considerable differences in N surpluses and N losses between countries and regions (van Grinsven et al., 2012; Hansen et al., 2012), and there is a lack of knowledge concerning the effects of spatial variation in N surplus at the watershed (Bartoli et al., 2012) and landscape levels (Drouet et al., 2012). Previous studies have focused on larger watersheds (Bartolli et al., 2012; Billen et al., 2012; Lassaletta et al., 2012) or regions (Leip et al., 2008; Neumann et al., 2011), and are typically based on statistics and publicly-available geo-databases rather than empirically collected data. These studies provide valuable insight into the consequences of N hotspots at these larger scales but there is a lack of knowledge concerning the interactions between the local farm management and the natural processes in specific landscapes with agricultural N-pollution hotspots (Cellier et al., 2011; Dalgaard et al., 2011).

In 2006, the pan-European research project NitroEurope was launched (Sutton et al., 2007; NitroEurope, 2012). This included a landscape-scale component that aimed to provide new knowledge on N losses from agricultural landscapes closing parts of the information gap between plot/field-scale experiments, and regional/national scale N statistics (Dalgaard et al., 2009; Bende-Michl et al., 2011). The authors of the present paper, and the related research institutions, all contributed to this landscape component of NitroEurope, including the inventory of six study landscapes with significant farm-related N-emission hotspots, and experiences from previous national research projects (Bouraoui et al., 1999; Dalgaard et al., 2002a,b; Dragosits et al., 2002, 2006; Hansen, 2004; Molenat et al., 2008).

The aim of the present paper is to:

– Compare farm-scale crop and livestock production data and the descriptions of the biophysical environment in the six case study landscapes in Poland, the Netherlands, France, Italy, Scotland and Denmark.

– Analyse the farm N-balance results, the differences between the input and output components of the N-balances, and the derived N surpluses across landscapes

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(The N surplus is defined as the different between the sum of inputs and the sum of outputs in the N-balance).

To this end we document the method developed to inventory farm data and calculate N balances in European landscapes. Moreover, we discuss the effects of N-surplus hotspots and the farming system heterogeneity within the landscapes as well as between landscapes. Finally, we assess the use of farm N-balance calculations and modelling for the independent verification of measured N concentrations in the environment, and the evaluation of possible measures to increase agricultural N efficiency and reduce N-emissions from agricultural landscapes.

2 Materials and methods

2.1 Study landscapes

As illustrated in Fig. 1, the study included farm data from six landscapes in Poland (PL), the Netherlands (NL), France (FR), Italy (IT), Scotland (UK), and Denmark (DK), all with 75 % or more of the total area taken up by agricultural land use (Fig. 2).

Based on local knowledge of relevant sites for the study of agriculture-related N hotspots, these landscapes were selected at the beginning of the NitroEurope project, and information on general land use and farming systems characteristics was collected. Compared with the average percentage of Utilised Agricultural Area for all 27 EU countries, which was 40.1 % in 2007 (Eurostat, 2011), all six landscapes have a very high proportion of their land under agriculture, dominated by grasslands in the Scottish and Dutch areas, and arable crops in the other landscapes (Fig. 2). The highest proportion of agricultural land use was found in Turew and Naizin (around 90 %), whereas it was around 80 % in NFW (the North Friesian Woodlands), Piana del Sele and Bjerringbro, and about 75 % in the Scottish landscape where the majority of grassland and moorland areas was extensively grazed. Other types of land use were mainly small woodlands, hedgerows and urban land including roads, farmhouses, gardens, etc., and this land cover ranged from 11 % in Turew to 36 % in Scotland (Fig. 2). All landscapes included water bodies. Surplus water from the Italian and the Dutch fields was pumped into channels bordering the area, whereas the boundaries in the other landscapes were defined by small watersheds into which surplus water from the areas drained.

In the following paragraphs, the biophysical environment and the farming systems of the six landscapes (with longitude, latitude coordinates), are briefly described. However, the exact borders of the landscapes and the farms studied are not given to respect the privacy of the farmers, from which interview data were collected (see Sects. 2.3 and 2.4). Specific land use, livestock, fertiliser and N input/output data from these interviews and landscape surveys are presented in the results section.

2.1.1 Turew, PL (52.0° N, 16.8° E)

The Turew landscape (5.1 km^2) covers the Wyskoć channel catchment, located in the West Polish Lowland. The terrain consists of a rolling plain, made up of a slightly undulating moraine, with many drainage valleys. The elevation ranges from 75 m in drainage valleys to 90 m at the highest points. In general, light-textured soils with favourable conditions for infiltration are found in the higher areas. Annual rainfall in this area is 594 mm. Most of the rain is concentrated in the spring and summer months (365 mm). The mean annual air temperature is 8 °C, with high seasonal differences.

The farming systems are dominated by 98 traditional family farms (average area: 12.8 ha, typically with mixed farming including both beef cattle, pigs, poultry and dairy production, and high-value horticultural crops), with manure commonly managed as farmyard manure (Fig. 1). The area also includes three large commercial farms with more modern livestock housing and manure handling techniques (average area: 875 ha, two with dairy cows and one with horse breeding, accounting for about 54 % of the total livestock in the area). The arable land use is dominated by rye and triticale cereals (with a relatively low N application of about 60–160 $kg\,N\,ha^{-1}$), and more heavily fertilised maize, forage, oilseed rape and horticultural crops. In this catchment there are multiple sources of N emissions with scattered manure storage and livestock

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buildings, as well as fields and gardens surrounded by extensive forests and hedgerow patches.

2.1.2 North Friesian Woodlands, NL (53.1° N, 9.1° E)

The Dutch landscape (5 × 5 km) is flat with relatively homogeneous soils, dominated by Gleyic Podzols and lies just above sea level. Via pumping, the groundwater level in the central parts of the landscape was controlled between 25 and 40 cm below surface, whilst the average groundwater level in other parts of the area was more than 120 cm below surface (Sonneveld et al., 2006). The average temperature ranges from 2–4 °C in winter to 16–20 °C during summer, with a mean annual precipitation of 763 mm over the last 30-yr period.

Northern Friesland has for generations been the heartland of dairy farming in the Netherlands (Tress et al., 2006), and the central study site in the NFW landscape is totally dominated by dairy farms. Therefore, only dairy farms were included in this study, with an average farm size of 50 ha, and with more than 1.5 high-yielding dairy cows per ha (annual yield of 7200 kg milk) and 2.6 other cattle per ha. Seven of these farms were placed inside the landscape, and the rest outside. There were no pigs or poultry in the central parts of the landscape, but there were seven small farms with sheep and six with horses inside the area, and outside the central landscape – more than 500 m from the fields of the dairy farms in the study – there were five commercial chicken farms. Within the landscape, high-yield grassland is the most widespread type of agricultural land use, followed by silage maize. Grazing is common, although most of the manure in the landscape is collected in the form of slurry from loose housing systems, and spread to fields during the growing season. In contrast to other agricultural landscapes of the Netherlands, the area is characterised by many hedgerows along ditches and water channels (hence the name, Fig. 1). However, the 19 % of non-agricultural land in the area is dominated by urban land use (11 %) and roads (5 %), with less than 4 % taken up by woodlands, orchards, water bodies and other semi-natural areas.

2.1.3 Naizin, FR (48.0° N, 2.8° W)

The Kervidy-Naizin catchment in Brittany covers an area of 4.9 km^2. It is characterised by gentle slopes of less than 5 %, with the northern part being particularly flat. The soils are loamy (dominated by luvisols), with well-drained upper slopes and poorly-drained lower slope areas (INRA, 2008). The mean annual precipitation over the last 30 years and mean annual potential evapotranspiration from 1994 to 2004 are 909 and 710 mm, respectively. The maximum and minimum average monthly precipitation occurs in January (116 mm) and July (45 mm), respectively (Molenat et al., 2008).

The land use is mainly agriculture, dominated by intensive livestock farming with cattle, pigs and poultry. About 32 % of the agricultural surface area of the catchment is covered by meadows, most of which are grazed intensively by dairy cows or other cattle (Molenat et al., 2004, 2008, Fig. 1). The arable land is dominated by winter wheat and maize crops, with the remainder taken up by leguminous plants, setaside land and oilseed rape. The soil surface N surplus in the Naizin catchment was estimated at around 220 $kg\,N\,ha^{-1}$ during the 1990s (Bouraoui et al., 1999), while Durand (2004) evaluated the leachable nitrogen at 150 $kg\,ha^{-1}$ for the same period. The non-cultivated area is occupied by roads and housing, with only a few forested patches.

2.1.4 Piana del Sele, IT (40.5° N, 14.9° E)

Located on an alluvial river plain situated at the coast of Southern Italy (Campania Region), the 3 × 4 km study landscape is characterised by a typical Mediterranean climate with hot and dry summers and cool rainy winters. The mean annual rainfall and temperature are 900 mm and 15.5 °C, respectively. The soils are generally coarse-loamy, but with large variations including fine-loamy, fine-silty and coarse-silty soils. On the lower part of the landscape, drainage water is pumped and channelled to the sea. Many areas are occasionally flooded during winter, especially in the large areas covered by plastic tunnels for vegetable production, where the soil absorption of rainfall is impeded.

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The landscape is characterised by highly productive farming systems. Vegetables with multiple annual croppings cover more than 80 % of the agricultural area harvested, with a fifth of the area under plastic cover, and a few cereal fields (<2 % of the agricultural area). The remaining area belongs to two very intensive water buffalo dairy farms (for mozzarella cheese production), with livestock houses, muddy paddocks (Fig. 1) and fodder crop areas (primarily alfalfa and silage maize). The area features high N emissions from mineral and organic fertilisers, silage fodders, and other livestock-related activities, and it is one of the strongest N and greenhouse gas emitting agricultural areas in Southern Italy, representing irrigated, high-input and high-income agriculture under Mediterranean conditions. The coastal-forested area accounts for about 15 % of the total landscape area, and is the other main land use type apart from agriculture.

2.1.5 Southern Scotland, UK (56° N, 3° W; approximate location to protect farm anonymity)

The northernmost study landscape (6 × 6 km) includes two similar-sized catchments, one dominated by moorlands and peaty soils, the other containing a variety of agricultural and other land uses on mixed soils including brown forest soils, peaty alluvial soils, peaty podzols and non-calcareous gley soils. The annual average temperature is 8 °C, and with a mean annual precipitation of 1040 mm the water surplus is considerable.

The agricultural activities are mostly related to extensive beef and sheep farming and a number of poultry sheds housing laying hens (incl. free-range systems). The north-western part is dominated by semi-natural moorland, whereas the southeastern part is mainly agricultural land. Within the wider landscape, the contrasting catchments are characterised by (a) peat bog with very low density sheep grazing (Fig. 1), and (b) agricultural land consisting of mainly grazed grassland at different stocking densities, with small areas of fodder crops and two major poultry farms, the largest of these without land and with manures exported from the landscape. In the Scottish landscape, grasslands included both improved pastures (48 %) and rough grassland with some grazing

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(14 %), and the 36 % of other land uses included moorland that also had occasional and very low intensity grazing (13 % of the area).

2.1.6 Bjerringbro, DK (56.3° N, 9.7° E)

The Danish study landscape is centred around the 843 ha upper catchment of the small stream Tyrebækken, which runs into the river Gudenå approximately 3 km downstream of the study area (Wohlfart et al., 2012), south of the town Bjerringbro. The soils are sandy-loamy on the relatively flat and fertile moraine plateau covering most of the area, but with more sandy soils on the lower-lying river terraces, and with narrow areas of organic soils along the stream (Dalgaard et al., 2002b). The elevation ranges from 25 to 58 m above sea level, with a mean annual temperature of 7.7 °C and an annual rainfall of 712 mm. The mean temperatures of the coldest and warmest months of the year (February and July) are 0.1 °C and 15.4 °C, respectively (PlanteInfo, 2011).

Specialised farms with pig-, dairy- and cereal cash crop production dominate the farming systems of the landscape, supplemented by smaller hobby and part-time farms, typically with a more extensive crop and beef cattle production. N-efficient, slurry-based manure handling systems are implemented on most farms (Fig. 1), with an obligatory 24-month storage capacity, and the potential to spread all manure during the growth season where high N-efficiency can be obtained (Kronvang et al., 2008). Cereals and oilseed rape are typically grown on the moraine plateau, with permanent grasslands along the stream and on steeper slopes, but high-yield rotational grass/clover and maize silage fodder crops also grown on the best moraine soils, with significant N input from both synthetic fertilisers, manure and clover N-fixation (Dalgaard et al., 2002a; Hutchings et al., 2004). It is a landscape with mixed land use, including significant patches of woodlands, bogs, permanent set-aside, hedgerows, gardens and other urbanized land use.

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2.2 Farm N balance and surplus

For the synthesis of results on agricultural N balances in the landscapes studied, the farm N balance was defined as from the farm gate (Dalgaard et al., 1998), including N inputs (i) to the farm, and N outputs (o) from the farm (Fig. 3).

The N surplus was, for all individual farms, calculated from Eq. (1) as the difference between net N output from the farm in the form of milk (o1) and meat, and the net N input to the farm in the form of net fodder import, net fertiliser import and N from the atmosphere. The net meat export was calculated as N in the meat exported (o2) minus N in imported livestock (i5), where "meat" exported in this context also included N in the form of eggs and wool sold (in the present study, N in eggs and wool was only relevant for a few farms in some of the landscapes). The net fodder import was calculated as the sum of N in imported fodder (i1) and seed (i2), minus N in cash crops sold (o4). Net imported straw was also included here. If a particular farm sold more N in cash crops than it imported in the form of fodder, straw and seeds, the net fodder import was negative. Similarly, the net fertiliser import was calculated as the sum of N in imported synthetic fertiliser (i3) and manure (i4) minus N in exported manure (o3), where the term "dressing" in Fig. 3 covers the sum of synthetic fertilisers and animal manures. Finally, N from the atmosphere is defined as the sum of the atmospheric N deposition (i6) and N fixed by legumes (*Leguminosae* sp.) (i7). Here, the N deposition for each farm was obtained from EMEP (2008, 2010) as the average modelled total annual dry and wet N deposition in the EMEP grid square containing the respective landscape centre coordinate for 2006 and 2008 (annually 11.2 kg N ha^{-1} for PL, 16.9 kg N ha^{-1} for NL, 17.4 kg N ha^{-1} for FR, 8.4 kg N ha^{-1} for IT, 8.1 kg N ha^{-1} for UK and 11.8 kg N ha^{-1} for DK). These actual values are probably higher in the landscapes with intensive livestock farming (Durand et al., 2010), an issue that is included in the sensitivity analysis section of the discussion of farm inventory method (Sect. 4). Based on Høgh-Jensen and Schjørring (1994) and Heij and Erisman (1997), i7 was simply estimated at 100 kg N ha^{-1} yr^{-1} for field peas, lupines and faba beans, 150 kg N ha^{-1} yr^{-1}

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for alfalfa and grass/clover ley fields with more than 25 % clover, and 20 $kg\,N\,ha^{-1}\,yr^{-1}$ for other grass/clover fields with a lower clover content. This is discussed in Sect. 4.

$$\text{N-surplus} = i1 + i2 + i3 + i4 + i5 + i6 + i7 - o1 - o2 - o3 - o4 \quad (1)$$

N-surplus ($kg\,N\,ha^{-1}\,yr^{-1}$) thus summarises N lost from the farm (in the form of emissions to the atmosphere or leaching to the soil-water system) or accumulated in the farming system (in stores, soils, perennial crops, etc.) during a particular year.

The farm N-surplus, and the split between the N-input and N-output categories of Fig. 3, are summarised for all farms in each of the landscapes studied. This allows a comparison of the overall N balance in the landscapes and provides the background for a discussion of differences in the characteristics of farming landscapes and the potential for N mitigation.

2.3 Farm data collection

At the start of the project, a common template for the collection of farm data from the six study landscapes was prepared, together with questionnaires to be used when interviewing farmers in the landscapes (Drouet et al., 2011). Data were organised in a relational database and included general farm data and related information about management of individual fields (Hutchings et al., 2012), manure stores and livestock houses (Dragosits and Dalgaard, 2008; Happe et al., 2011). The aim was to interview all farmers with fields in the defined landscapes. This was generally successful, except in the Dutch landscape, where less than 30 % of the farm area was covered by interviews, and half of the farms included for NL were actually placed outside the landscape. However, animal counts were available for all NL farms, and since the farms of that area were all relatively similar dairy farms, this was not considered a serious problem. Moreover, a preliminary comparison of N surpluses from the group of farms inside and outside the NL study landscape, respectively, did not show significant differences. In the other landscapes, the inventories covered over 90 % of the farmland,

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with very few farms not included, either because the farmer did not want to participate, the quality of the data collected was considered poor, or most of the farm was located outside the landscape.

The results of the present study mostly rely on the general farm data collected, including:

- Types and numbers of animals on the farm at the start of the calendar year and arrived or left during the year.
- Types and quantities of manures on the farm at the start of the calendar year and produced, imported or exported during the year.
- All other main N-containing materials and produce generated on the farm, imported or exported during the year.
- Stores of all other main N-containing materials on the farm at the beginning and by the end of the year.

In addition, for most of the farms, data on field areas, crop types, the proportion of the time the fields were grazed, and the consumption of synthetic and organic fertilisers were collected for each field (except for the Netherlands, where only general farm data with aggregated figures for each farm were included in the database). All data were collected for one calendar year, except for field operations associated with winter cropping, which were collected for the cropping year. For example, fertiliser applied in the previous autumn to the crop harvested in the calendar year was included, whereas the field operations after harvest were excluded (e.g., the fertiliser distributed after the last harvest date was not included, because it was considered as preparation for the following year's crop). In Poland, France, Scotland and Denmark the farm interviews were carried out for the year 2008, whereas data from the Netherlands relied on data for 2007, and for logistical reasons data for Italy were collected for 2009.

A total of 222 farms were included in the study, with a total farm area of 11 440 ha. This sample covered almost all farms included by local partners in the common

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database, except for the few entries with data quality problems and where farmers declined to participate in the study (in total less than 10 farms). The sample is therefore considered representative for the study landscapes, except perhaps for Italy, where one of the two large water buffalo farms was not included. Moreover, for Scotland the large landless poultry farming was deliberately treated separate in the N-balance calculations; partly because the collected data on manure export from this farming system was uncertain, and partly to avoid division by zero when the individual farm N surplus values were summarised per farm area (see also Sect. 4). Finally, for FR and UK three farms declined to participate. However, as elaborated in the discussions section, this was considered not to have significant consequences for the overall results.

2.4 Templates and default values for N-containing materials and products

For the farm data collection, template lists with all main N-containing materials and products were made, including livestock types, crop types, manure types, and other imported and exported farm inputs and outputs. In an iterative process, an initial draft list was sent to the local partners and revised to include all major types present in the landscapes. Additionally, a default N content for each type was proposed based on figures from Dalgaard et al. (1998, 2002a) and Strudsholm et al. (1997), with the possibility to be locally adapted for each of the landscapes. Tables 1 and 2 show the default N contents for imported and exported materials and types of manure, respectively. In general, the local revisions to these standard values were few and minor and are not shown here. However, in addition to the general values, more specific default values for subtypes were included in the database, and used by the partners. This, for example, included default N contents for specific types of crops; e.g. a specific default value of 14.96 $\mathrm{kg\,N\,t^{-1}}$ for winter rye cereals (*Secale cereale*), 18.79 $\mathrm{kg\,N\,t^{-1}}$ for winter wheat (*Triticum aestivum*), 31.45 $\mathrm{kg\,N\,t^{-1}}$ for oilseed rape (*Brassica napus)*, 48.71 for fava beans (*Vicia faba*), and specific N contents for the different types of synthetic N fertilisers used in the landscapes.

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2.5 Verification measurements

In addition to the farm data collection, measurements were made of various N compounds in the air, soil and water within the landscapes during the period 2008–2009 (Theobald et al., 2011), in order to assess the fate of the N surplus produced by the farms. Measurements of mean monthly ammonia (NH_3) concentrations were made at up to 31 locations within each of the six study landscapes. Soil nitrate concentrations in the top 20 cm soil layer were measured periodically (up to 18 times per year) at up to nine locations within four of the study landscapes (DK, FR, IT and PL). Nitrate concentrations were also measured periodically (up to 12 times per year) in the groundwater at up to 15 locations within three of the landscapes (DK, FR, and PL) and in stream water at up to nine locations within five of the landscapes (all except IT).

3 Results

This section summarises results from the farm data collection (Sect. 3.1), the derived N balances for the six study landscapes (Sect. 3.2), comparison with independent measurement data of N in air and soils (Sect. 3.3), and analysis of the N-surplus variation and hotspots (Sect. 3.4). A special case study on the effect on N-mitigation measures carried out in the Danish study landscape, is also shown, with the N-surplus results for 2008 compared to a previous study from the period 1994–1998 (Sect. 3.5).

3.1 Farm data

The number of farms studied, and the farm areas covered, varied between landscapes, with the largest number of farms in Poland and the smallest sample from the Netherlands (Table 3). Fortunately, the most homogeneous farm size and farm type distribution was also found in the landscapes with the smallest number of samples, where the differences between the average and the median farm size in NL, DK and FR were 6 %, 31 % and 51 %, respectively, compared to much larger differences in the landscapes of

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IT, PL and UK. This is because the latter three landscapes are characterised by many small farms and a few very large farms, with median farm sizes of 5 ha, 11 ha and 42 ha compared to an average farm size of 19 ha, 47 ha and 193 ha, respectively.

The results of the farm interview confirmed the general patterns observed during the initial characterisation of the landscapes, with the most mixed livestock production in PL and FR, specialised dairy production in NL and IT, and meat production from pigs, poultry, beef or sheep dominating in DK and UK (Table 3, Sect. 2.1). In general, the crops grown in the landscapes correspond to the needs of the livestock being raised in the individual landscapes, with grasslands and forage crops for ruminants (cattle and sheep) and cereals for non-ruminants (pigs and poultry). However, an exception to this pattern is the landless poultry production in UK and the intensive water buffalo dairy production in IT, which were both based on imports of feedstuff, uncoupled from the local crop land use. In addition, these two systems export almost all of their manure out of the landscape. Nevertheless, even if these systems were included in the calculations, the livestock densities in these two landscapes are relatively low compared to especially the landscapes in NL and FR, and have a more heterogeneous distribution between farms than in the other landscapes. Finally, the use of synthetic fertiliser was much higher in Piana del Sele (IT) than in the other landscapes, mirroring the large production of outdoor and plastic-covered vegetables, with up to four crops per year, and a subsequent high fertilisation rate.

3.2 Landscape nitrogen balances

The farm N surpluses and the Fig. 3 components of the N balance were calculated and compared for the six landscapes (Fig. 4). The highest N surpluses were in descending order found in the landscapes of Piana del Sele (IT), Naizin (FR), NFW (NL), and Bjerringbro (DK), but with no statistically significant differences between the N surpluses in these four landscapes. However, the N surplus in Turew (PL) was significantly lower than in both NL and FR, and the N surplus from the land-based farming in Southern Scotland (UK) was significantly lower than from any of the other landscapes. However,

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if the landless poultry farming was included, the Scottish landscape showed an N surplus similar to those of PL and DK.

As expected, the highest N export of products in the form of meat and milk (NL and FR) or feed and vegetables for human consumption (IT) was found in areas with the highest net N inputs of fodder (FR and NL), atmospheric N input (DK, FR and NL), and imports of fertiliser and manure dressings (IT and NL), whereas the lower N-input systems of PL and UK also showed significantly lower net N outputs, and as mentioned a subsequently lower N surplus.

3.3 Comparison with independent N measurements

In order to investigate links between farm N surplus and N losses to the environment, correlations between the concentrations of N compounds measured within the six landscapes and the average landscape N surpluses were calculated (Fig. 5). A significant, linear correlation ($R^2 = 0.59$) was found between the lowest site-mean atmospheric NH_3 concentration for each landscape and the respective N surplus (Fig. 5, left). The lowest site-mean is indicative of the emission density of the landscape and surrounding areas. This correlation was very much determined by the low value for Scotland, which also could be determined by the low concentration in air flowing in from the Atlantic. By contrast, no significant correlation was found between maximum NH_3 concentrations and N surpluses because the maximum values measured within a particular landscape depended on the proximity of the measurement equipment to individual emission sources in the landscape (data not shown). Other significant correlations between the measurements and farm N surpluses were found for maximum soil and groundwater nitrate concentrations ($R^2 = 0.83$ and 0.97, respectively) (Fig. 5 centre and right). By contrast, no significant correlations were found between stream-water nitrate concentrations and farm N surplus reflecting the poor direct connectivity between the fields and streams within the landscapes. These general results show relatively clear correlations between N surpluses and N concentrations in the surrounding environment, and thereby for example potential losses to the atmosphere (e.g. through NH_3 emissions)

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and to the soil and water (indicated by soil and groundwater nitrate measurements). Thereby, these results will serve as background material for the further discussion and analyses of the regional variations and potentials for N pollution mitigation options.

3.4 N-surplus variation and hotspot farms

To explore the N-surplus variation indicated by the confidence intervals of Fig. 4, and to identify and discuss specific N-surplus hotspots and potential N-mitigation measures in the six landscapes, the farm N-surplus values were plotted with the average farmland N input in the form of synthetic fertilisers, manures, N fixation and deposition from the atmosphere (Fig. 6). A statistically significant ($R^2 = 0.31$) but not very clear, positive linear correlation between land-based N input and the derived per area N-surpluses was found, and with interesting differences between the hotspot farms in each of the six landscapes.

Both the highest average N-surplus and some of the largest hotspots were found in Piana del Sele (IT), which contains the largest proportion of farms above the linear regression line of Fig. 6. The largest single hotspot was the main water buffalo farm (furthest to the right i Fig. 6), but the intensive vegetable production sites of IT also showed significantly higher N surpluses than the average. This was in sharp contrast to the two roughage fodder arable farms of Piana del Sele (IT) which had the lowest N-surplus values (about 22 kg N ha^{-1} yr^{-1}) despite relatively high N inputs of 91 and 248 kg N ha^{-1} yr^{-1}, respectively. This was due to a large export of high N-content alfalfa and maize silage to the main farm section where animals were bred, and therefore in reality these farms were closely coupled to the water buffalo milk production and were therefore not really examples of an independent farming system with a low N surplus.

The other major hotspot farms were in Naizin (FR), Turew (PL) and Bjerringbro (DK). The farms with the largest N-surplus in DK were all hobby-based beef cattle farms with a large proportion of N-fixing grass/clover crops, significant feed imports and no export of plant products, whereas the major hotspots in FR and PL were pig farms. This was in contrast to some of the other pig farms in these landscapes, and especially to

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the two industrial pig farms in DK which were both, despite relatively large land-based inputs, examples of farms with very low N surpluses compared to the average (the two points with an N input of 199 and 302 $\mathrm{kg\,ha^{-1}\,yr^{-1}}$, respectively in Fig. 6). However, the best examples of farms with a high N efficiency were probably the twelve dairy farms in NL, which all showed a lower N-surplus compared to the average line, even though they were a significant source of N losses (Fig. 6), and an average N-surplus not differing significantly from the average in IT, FR and DK (Fig. 4). However, as discussed later, there are important lessons to learn from these systems in relation to options for N mitigation. Finally, it should be mentioned, that landless poultry farming, with the largest N surplus in the UK landscape, was not included in Fig. 6, which only contains land-based systems, and the relatively low UK farm N surpluses illustrated.

3.5 Example on the effect of N-mitigation measures in the Danish landscape

Based on results from Dalgaard et al. (2002a) the average N-surplus from farms in the Danish landscape for the period 1994–1998 was $186 \pm 46\ \mathrm{kg\,N\,ha^{-1}\,yr^{-1}}$, excluding about 10 % of the agricultural area that was used for setaside during that period in order to receive EU subsidies (Levin and Jepsen, 2010).

These results were used to evaluate the effects of measures implemented in general legislation between 1998 and 2008 to increase N efficiency and which were expected to have an impact on farm N surpluses in the particular landscape (i.e. regulation of the maximum farm livestock densities, statutory norms for crop N fertilisation set to 10 % below the economic optimum, and obligatory farm N accounts with specified demands for manure N utilisation and thereby restrictions on fertiliser imports etc., see details in Kronvang et al., 2008). The average N surplus in the landscape was reduced over the period by about 22 % overall when compared to the results from Fig. 4, which include insignificant areas of setaside (<1 %) and represent a livestock and crop production in 2008 similar to the 1998 situation. However, because of the large uncertainty and variation between farms in such a small study area, the general reduction was not statistically significant ($p < 0.07$), but suggests that there was a larger reduction on

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farms with a high livestock density compared to farms with a low livestock density and less manure applied per field area. The Danish N legislation has specifically focused on measures to reduce N emissions from livestock farms and manure systems (Kronvang et al., 2008), and the present dataset and N-surplus accounting methods presented an opportunity to make an independent test of the effect of such measures. Consequently, the N-surplus reductions were tested separately for farms with respectively less and more than 1 LSU ha^{-1} yr^{-1} (where 1 livestock unit (LSU) equals 100 kg N in manure produced ex store, or distributed during grazing). No significant difference ($p = 0.80$) was identified for farms with <1 LSU ha^{-1} yr^{-1}, but for farms with >1 LSU ha^{-1} yr^{-1} there was a significant difference ($p < 0.01$), both when setaside areas were included and when they were not (Pedersen, 2011).

4 Discussion

The results show that the farm N-balance method presented is useful for comparing farming systems in Europe, identifying hotspots for N emissions, and evaluating effects of N-mitigation measures. In particular, it is interesting that this method enabled comparisons across a large range of biophysical conditions, from Scotland in the north to Italy in the south. This indicates that the N-surpluses may be used as an independent dataset for validation of measured and future modelled N emissions in agricultural landscapes. Nonetheless, there are important uncertainties, shortcomings and potentials for further development in relation to the N-balance method and its application, which we will discuss in more detail in the following.

4.1 The farm N inventory method

Like many previous farm N-balance studies (for example Beukes et al., 2012; Cameron et al. 2012; Dalgaard et al., 2011, 2002a, 1998; Halberg, 1999; Shingo and Kiyotada, 2012; Spiess 2011), the present calculations were based on a set of standard values for

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N contents in the classes of farm inputs and products defined (Tables 1, 2). However, although these standards were reviewed and agreed among all landscape partners, they involve significant uncertainties, and differences between N contents of materials in the different landscapes and farming systems must be expected. This uncertainty was reduced via the option in each landscape to use specific values for product subclasses (for example a specific N content value for wheat cereals instead of the generally lower standard class value for cereals of 16.3 $kg\,N\,t^{-1}$). However, this option was only used by local partners in a few cases. From this we concluded that there is no reason to expect a systematic over- or underestimation from using the current method, where the standard value approach overestimates N contents in some cases and underestimates in others.

Another critical point may be the reliance on values from a single year's N balances, thus ignoring potential annual variations. Significant differences between years have previously been revealed (Hansen and Kyllingsbæk, 2007), especially in very dry years with higher N surpluses related to crop yield depressions. Therefore it is important to state that all the results included here were from years without extreme yields or weather conditions, and as a consequence, we consider that the results may be interpreted as typical for the farms and landscapes studied. This also includes the values for N-fixation which are similar to those reported by Smil (1999) and Spiess (2011). Nevertheless, it must be emphasised that, at least at farm level, the N-fixation values are approximate and uncertain estimates, rather than measured values, and that the sensitivity to changes in these estimates is quite important for the interpretation of the final N-balance results. This is particularly the case for the results in Bjerringbro (DK), Naizin (FR) and Turew (PL), where some cattle farms have extensive areas with N-fixing grass/clover, compared to the grassland in the NFW (NL), which Heij and Erismann (1997) considered to have a much lower average N fixation rate of 20 $kg\,N\,ha^{-1}\,yr^{-1}$. This uncertainty and potentially skewed distribution should be kept in mind when interpreting the results of Figs. 4 to 6. Moreover, as mentioned in Sect. 2.2, the N depositions taken from the EMEP (2008, 2010) represent average values for the relevant 50 × 50 km grid and may be underestimated, especially for local areas and farms with a high livestock density. This may be the case for NL, which according to Fig. 5 had the highest measured ammonia concentration, even though the EMEP estimated the highest N deposition to occur in the French landscape. According to Durand et al. (2010) the EMEP deposition value for FR was also set too low. In reality, farm level N depositions may therefore be underestimated for livestock farms and for farms near large livestock facilities such as the large poultry farms in the UK (Skiba et al., 2006; Dragosits et al., 2002). Conversely, it may be overestimated for low livestock density farms such as the large semi-natural areas with extensive sheep grazing in Scotland (UK), or the coastal vegetable farms in Piana del Sele (IT) where fertilisation, although at a very high level, was based on synthetic fertilisers with a relatively low N emission compared to manure-based and livestock-related systems (Oenema et al., 2011). Moreover, both these farm types were located in the western parts of the landscape. For the Italian landscape, westerly winds of the Mediterranean Sea dominate, so the actual N-deposition values may be lower than expected for the particular farming systems. Consequently, as discussed below, the inclusion of such landscape heterogeneity and boundary condition effects should be a topic for further research.

4.2 Landscape differences in farm N surplus and efficiency

The general N balance results of Fig. 4 show an interesting pattern, with the highest feed imports and animal production in the grassland and forage crop-dominated landscapes of the central Atlantic biogeographical zone in Fig. 1 (NL and FR, and the landless poultry farming systems in UK). By contrast, the landscapes of PL and DK (respectively in the middle of and on the border to the continental zone) represent more cereal-based production systems, with a less intensive livestock production and consequently a lower net feed import. The N surplus was also generally lower in these landscapes. Finally, the landscapes of IT and UK (respectively in the Mediterranean and the Northern Atlantic zones of Fig. 1) represent more heterogeneous farm N balances that vary across a wide range of systems: from large water buffalo and poultry

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farms with very high feed imports and where manure is exported, over vegetable farms with a high crop export but a large import of fertiliser, to extensive sheep farms with non-fertilised semi-natural grasslands and only marginal N exports per land area.

If N efficiency is defined as net N output in products sold divided by net N inputs purchased by the farmer ($[o1+o2-i5] \times ([i1+i2-o4]+[i3+i4-o3])^{-1}$ in Fig. 3), the highest N efficiency was found in FR and NL (32 % and 31 %, respectively) , whereas the average efficiency was 24 % for IT, 21 % and 19 % for PL and DK, respectively, and only 5 % on average for the extensive, land-based farming systems in UK. However, if the approximate figures from the poultry farming were included, the average N efficiency would be about 60–80 % for the UK landscape. However, this values is very difficult to compare with those from the other landscapes because of the large manure export, which is here considered a product, but would lead to N surpluses and N pollution in the neighbouring landscapes to which the manure was exported. Thus the systems with the highest N inputs are also those which are most N-efficient, even though they also have large N surpluses and losses to the environment, and this contrasts much of the thinking about regulations that are often input-related. However, overall N efficiencies in the landscapes studied are relatively low, with less than one third of the N inputs utilized in the products sold; and this even without the inclusion of N inputs from the atmosphere in the equation. This would certainly indicate room for improvement, and based on the large differences between the N-surplus of the average and the most modern and N-efficient farms, it was concluded that N-surplus reductions of 25–50 % as compared to the present level were realistic in all landscapes.

It is important to note the large uncertainties in the N-surplus and N-efficiency figures. To be able to draw general conclusions, in line with those discussed above, a large sample size is needed. The farm sample size was small, in particular for Bjerringbro (DK) with 13 farms, and Naizin (FR) with 17 farms. For DK there seemed to be a trend towards relatively lower N surpluses from large and full-time farms compared with the relatively large values of small hobby farms. The farm sample was, however, too small to document this statistically, even though this effect has been shown in previous

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studies (Dalgaard et al., 2002a, 2011) and follows the general trend of higher N efficiencies for intensive farming systems. Moreover, in DK two of the largest and most N-efficient farms included in the study had most of their fields outside the actual watershed study landscape, and one of the farms located in the middle of the watershed closed its dairy production immediately before the study year, but kept some heifers, and some remaining manure was spread within the study year. This affected the estimates of average N balances and illustrates the importance of local dynamics, and the uncertainties and peculiarities of studies in specific landscapes. Future studies of landscape N balances should include a larger number of farms, to counteract such effects. In contrast, we consider the farming systems of NL more uniform, whereby even the small sample of only 12 farms is likely to have given representative results for the central dairy farming area of the NL landscape. The two water buffalo farms included in IT and the large poultry farms in the UK must be considered special cases, and more data from similar farms are needed to draw general conclusions for the N balance of such systems.

4.3 N-surplus hotspots and effects of landscape heterogeneity

This study includes farm N-surpluses calculated at the farm gate i.e. from inputs and outputs recorded in the annual farm accounts and from estimated atmospheric inputs (Fig. 3). The advantage of this approach is that the farm N balances are largely based on measured flows and are thus considered robust. However, a disadvantage of this whole-farm method is that the N-surpluses can only be used as a general indicator of N lost or accumulated within the whole farming system and do not indicate whether N is lost to the aquatic environment or to the atmosphere and from which component of the farm the loss occurs. The data collected from the farms would have permitted the use of an alternative approach in which the farm N surplus was calculated from the N-inputs and outputs from individual fields, livestock houses and manure storages. While this would have permitted the N surplus to be partitioned between the farm components (principally between the animal housing/manure storage and the fields), the N input

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and output data at the component scale were considered too uncertain to justify the use of this approach. The uncertainty in the data arises both from uncertainties in the measured flow of material (e.g. manure, crop yield) and the difficulties measuring changes in the short-term storage of N in the components (mainly manure in animal housing and storage). The latter uncertainties largely disappear if data can be collected over several years.

To partition the N-surplus into types of losses (nitrates, ammonia, nitrous oxide etc.) and soil N accumulation/erosion, a much more detailed approach would be needed, including modelling and partitioning of N-inputs and N-outputs to fields, livestock houses and manure systems (Dalgaard et al., 2011; Happe et al, 2011), as well as the inclusion of N exchange dynamics with non-agricultural areas (Drouet et al., 2012). This would make it possible to geographically map hotspots for N-sources (and N-sinks) to these particular compartments within the landscape. Such an approach would be worthwhile, since the identification of hotspots would enable mitigation measures to be targeted to these areas, which is likely to result in a more cost-effective reduction in N pollution. However, the experience from the current study is that it is difficult to collect the more detailed data in the quantity and with the quality that is necessary. The uncertainties could be reduced by increasing the number of farms included in the survey but this would also add significantly to the cost; the more detailed data are either not collected by the farmer, so the cost of collecting the individual data is high, or are not collected by the farmers in a standard format.

A second difficulty encountered in this study was how to treat farms that exported significant quantities of manure to areas outside the study landscapes. One option would be to increase the area of the landscape to include the recipient areas. This might already be necessary to combat measurement uncertainties (see above) but for areas with high livestock densities, the pressure from national and EU legislation is forcing farmers to export manure significant distances (e.g. the Netherlands), so this is likely to be too expensive. An alternative would be to identify the recipient areas and

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include them in the study, either by surveying or by using modelling or using appropriate emission/loss factors to account for the associated losses to environment.

4.4 Landscapescale measurements and sustainable farm N management designs

The correlation between the average farm N-surplus data collected for the relatively small landscapes of around 5 × 5 km or smaller, and the independent measurement results of atmospheric ammonia and soil and groundwater nitrate concentrations in the landscapes provide an indication of the usefulness of N-surplus for informing on N-pollution problems (Fig. 5). The correlation between N-surplus and nitrate concentrations was the strongest, which would also be expected because the nitrate measurements relate directly to soil and groundwater within the landscape, whereas the ammonia concentrations relate to emissions from farms both within and outside the actual landscapes. General correlations between trends in farm N-surplus and nitrate concentrations in Danish groundwater for the period 1950–2007 have been published by Hansen et al. (2011, 2012), but the potential remains for further investigation of more detailed landscape-level effects and correlations (Bende-Michl et al., 2011). In this context, the present results are promising for further investigation of correlations between the site-based N measurements carried out in the landscapes during the NitroEurope project (e.g. Schelde et al., 2012; Wohlfart et al., 2012) and the geographical location of farms and derived N surplus and N sources from specific fields, livestock houses, manure stores, etc., as well as the transfer to N-sinks in the landscapes (Drouet et al., 2012).

There is a further potential to protect sensitive semi-natural areas vulnerable to N pollution not only through increasing N-efficiency, but also via landscape-level spatial planning by for example planting of hedgerows and trees nearby livestock facilities (Dragosits et al., 2006) or along water courses (Christen et al., 2012). For future studies, it would therefore be interesting to investigate this potential within the landscape sites presented here. Such analyses could also include an investigation of how

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N losses from local hotspots in the landscape cascade from, for example, N in manure to ammonia in the air, of which some will deposit either to nature areas or agricultural land, wherefrom it may again either be recycled in the system via harvest or eventually be lost in the form of nitrate. All these mechanisms are complex and include important feedback mechanisms, the proper use of which in agricultural management may help to mitigate the N pollution problems and improve production N efficiency.

An assessment of the effects of N-mitigation measures in the Danish study landscape 1998–2008 exemplifies the results of such measures on N-surplus. Significant reductions in the N surplus from livestock farms were documented via a better utilisation of N in livestock manures over the period. Consequently, in the Danish landscape 2008 there was a tendency to less use of synthetic fertilisers with a higher farm-level use of manures. However, such a tendency was not found in the farm level datasets of any of the other landscapes, indicating potentials for a better utilisation of livestock manures similar to that achieved in the Danish area via the use of technologies and management for improving manure N use to replace synthetic fertilisers. Moreover, there was a slight tendency for a non-linear, exponential relationship between land-based N input and N surpluses in Fig. 6, indicating a potentially better N-utilisation via a more uniform distribution of manures and other types of N inputs from hotspot farms to other farms with less intensive N-input regimes (Dalgaard et al., 2011). As the results show a positive relationship between farm N surpluses and landscape N concentrations in the air, soil and water, it can be surmised that a reduction in surpluses will lead to a reduction in N losses to the environment. A take-home message must be that methods are available to identify and evaluate levels of N surplus in specific landscapes, and to estimate the overall effects of measures tailored to reduce farm N losses.

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5 Conclusions

The method presented here was used to calculate farm N surplus as an indicator of N losses and of the potential for improved N management in six agricultural landscapes of Europe.

As an average, the highest N surpluses for the study period were found in the most livestock-intensive landscapes of Naizin (FR), North Friesland (NL), and Piana del Sele (IT) where intensively-fertilized, multiple annual croppings in vegetable production also contributed significantly to the N surplus. However, all landscapes showed hotspots from livestock farms, including a special case of large "landless" poultry farming in Southern Scotland (UK). For future studies the question will be how to include indirect N surplus and N emissions from such farms with a large export of manure out of the landscape.

Positive correlations between average landscape farm N surpluses and measured concentrations of ammonia in the air, nitrates in soils, and nitrates in groundwater were found, indicating that N surpluses may be used as an independent dataset for validation of measured and modelled N emissions in agricultural landscapes. Such significant couplings of reductions in N surplus with groundwater nitrates have previously been published for Denmark (Hansen et al., 2011, 2012), consistent with the present results from the moraine-soil-dominated landscapes in Turew (PL), Naizin (FR) and Bjerringbro (DK). In this context, an average 22 % N surplus reduction was achieved in the Danish study landscape from 1998–2008, due to measures to reduce N surplus from livestock farms.

In summary, this study indicates that farm N surpluses can be related to empirical measurements of N concentrations in the environment and therefore support their use as indicators of the environmental impact of N lost from farming systems. We conclude that partitioning the N surplus between farm components and spatially over the landscape would allow mitigation measures to be targeted and therefore to be more cost

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effective. However, we also identify methodological and financial barriers to achieving this partitioning.

In all six study landscapes, a large variation in the farm N surplus was found, and thus a large potential for N-surplus reductions of maybe 25–50 %, compared with the present level. The N-surplus method was shown to be effective for comparing and synthesizing farm N emissions and the potential of N mitigation options. The method is recommended for use in combination with other methods for the assessment of landscape N emissions and farm N efficiency, including more detailed N sink and N source hotspot mapping, measurements and modelling.

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Table 1. Default N contents of imported and exported materials.

Material	Default N content (kg N^{-1} fresh weight)
Beet pulp (dried)	14.4
Cereals	16.3
Eggs	18.1
Feed milk	56.3
Fertiliser nitrogen	1000.0
Fresh milk	5.0
Fresh green forage (alfalfa)	6.0
Fresh green forage (grass)	6.3
Fresh green forage (grass/clover)	5.7
Full-ration concentrate mix	25.6
Hay	16.0
High energy concentrate	52.2
Low energy concentrate	25.8
Medium energy concentrate	43.9
Meat (live animals)	46.0
Rape cake	49.3
Silage (alfalfa)	18.0
Silage (beet pulp)	3.8
Silage (clover grass)	9.1
Silage (grass)	8.5
Silage (maize)	3.9
Silage (whole crop)	6.0
Soy beans	56.4
Soybean oil cake	70.2
Straw	5.4
Sugar beets	2.1
Wet distillery grain	3.4
Whey	35.0
Whole crop fresh	5.8
Wool	3.0

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Table 2. Default N contents in imported and exported manure.

Manure type	Default N content ($kg\,N^{-1}$)
Pig slurry (fattening pigs)	5.4
Cattle farm yard manure (FYM)	8.4
Composted manure/compost from other materials	2.0
Degassed cattle slurry	3.9
Degassed pig/mixed slurry	4.0
Horse FYM	7.5
Liquid fraction of cattle manure	5.4
Liquid fraction of mixed manure	5.0
Liquid fraction of pig manure	4.0
Mixed FYM	8.6
Mixed slurry	5.4
Other organic fertiliser (e.g., bone meal)	2.0
Pig FYM	8.8
Pig slurry (sows and piglets)	4.6
Sewage sludge	6.0
Sheep/goat FYM	8.4
Solid fraction of cattle manure	5.6
Solid fraction of pig (fatteners) or mixed manure	5.9
Solid fraction of pig manure (sows + piglets)	8.1
Solid poultry manure	21.0

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Table 3. Agricultural land use and livestock production characteristics of the study landscapes.

Landscape	Turew, PL	NFW, N	Naizin, FR	Piana del Sele, IT	Southern Scotland, UK	Bjerring-bro, DK	Total
Farms studied (number)	100	12	17	53	27	13	222
Farm area (ha)	4556	658	1246	931	3092	957	11 440
Crops (% of farm area)							
Alfalfa	3.5			8.1			2.1
Covered orchards				0.7			0.1
Covered vegetables			1.0	15.7			1.4
Setaside grassland						0.6	0.1
Fava bean			0.7				0.1
First yr grass ley	0.5				0.8	14.8	1.7
First yr grass/clover ley	0.0			2.1	0.3	1.7	0.4
Fodder beet	0.2				0.7	0.4	0.3
Other grass/clover	0.0	88.9	10.6		1.6	5.9	7.0
Maize (silage)	10.5	11.1	23.5	2.4		2.0	7.5
Oats	2.3		1.2			3.2	1.3
Orchards	0.4			1.5			0.3
Outdoor vegetables	0.5			67.8			5.8
Peas			2.6				0.2
Permanent grass	15.0		8.9		24.9	1.4	13.9
Permanent grass ley			13.1		18.5	1.7	6.5
Potatoes	0.8		6.4		0.0	0.3	0.9
Rough/extensive grassland	1.0		0.3		49.7		14.2
Rye	11.7						4.8
Second year grass ley			0.1		0.9	1.5	0.4
Spring barley	5.4				2.1	3.6	3.1
Spring rape	0.1				0.2		0.1
Spring wheat	0.9						0.4
Sugar beet	3.5						1.4
Triticale	26.4			1.3		6.0	11.4
Winter barley	0.8		3.7			15.2	2.0
Winter rape	7.8		1.5			10.0	4.2
Winter wheat	8.6		26.5	0.4	0.2	31.6	8.7
Livestock (number)							
Dairy cows/buffalos	1233	1010	477	667		60	3447
Other cattle/buffalos	1426	1735	1466	209	1049	21	6100
Sows	729		1105			21	2051
Piglets	2968		10 250			2079	1529
Pig Finishers	2092		7714			4125	1393
Poultry	536		40 500		1 419 692		1 460 728
Sheep and lambs					9338		9338
Horses and ponies	301				50		351
Livestock density[a] (LSU ha^{-1})	0.7	2.9	2.5	1.0	0.3 + 1.8[b]	0.9	0.9
Free-range grazing (% manure)	2	8	19	9	25	16	12
Synthetic fertilisers (kg N ha^{-1})	112	112	79	251	18	80	91

[a] 1 livestock unit (LSU) equals 100 kg N in manure produced ex store or distributed during grazing.

[b] For UK about 1.8 LSU ha^{-1} poultry manure was exported out of the landscape.

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Fig. 1. Location of the six study landscapes: (1) Turew, PL; (2) The North Friesian Woodlands, NL; (3) Naizin, FR; (4) Piana del Sele, IT; (5) Southern Scotland, UK; and (6) Bjerringbro, DK; superimposed onto the European Environmental Agency's (2002) biogeographical regions of Europe. The photographs show important farming systems in these landscapes (clockwise from bottom left corner, and with number corresponding to the actual landscape): intensive cattle grazing on wet, permanent grasslands in France and the Netherlands, sheep grazing on rough grassland in Scotland, pig slurry application with trailing hose to winter cereals in Denmark, farmyard manure heap outside a traditional farmhouse in Poland, and a water buffalo paddock in Italy.

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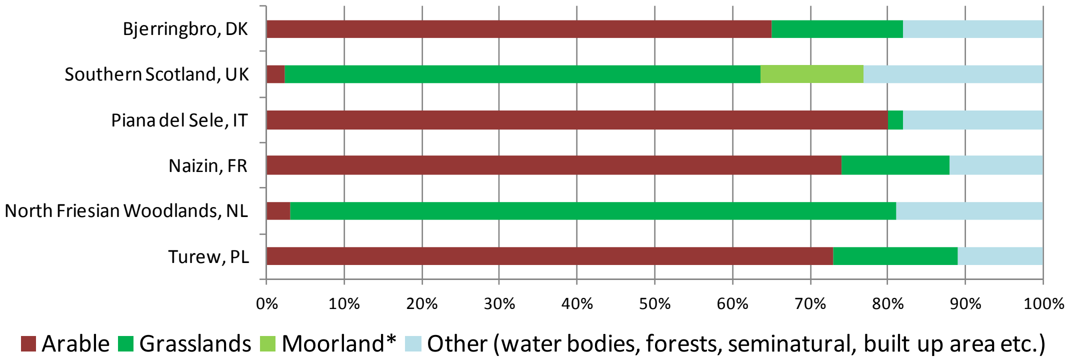

Fig. 2. Overall land use distribution in the six study landscapes 2007–2009. * Moorland and rough grassland account for about one fifth of the defined grasslands in the Southern Scotland study area and are only extensively grazed.

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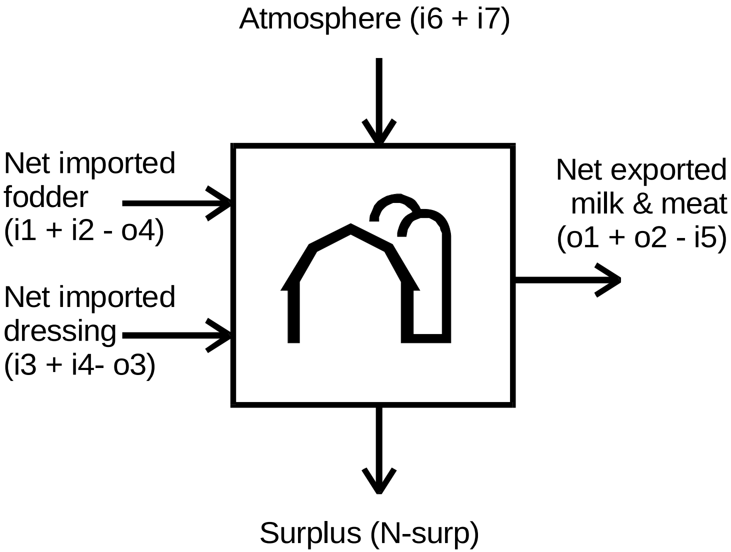

N-inputs:
i1= imported fodder
i2= imported seeds
i3= imported fertilisers
i4= imported manure
i5= imported animals
i6= deposited atmospheric N
i7= fixed atmospheric N

N-outputs:
o1= exported milk
o2= exported meat
o3= exported manure
o4= exported sales crops

Fig. 3. Farm N inputs (i1–i7), N outputs (o1–o4) and N surplus, N-surp (from Dalgaard et al., 1998). The same balance can be calculated for a number of farms within a landscape. In this context "exported meat" (o2) also includes N in the form of eggs and wool sold, and the term "dressing" is used as the sum of synthetic fertilisers and animal manures.

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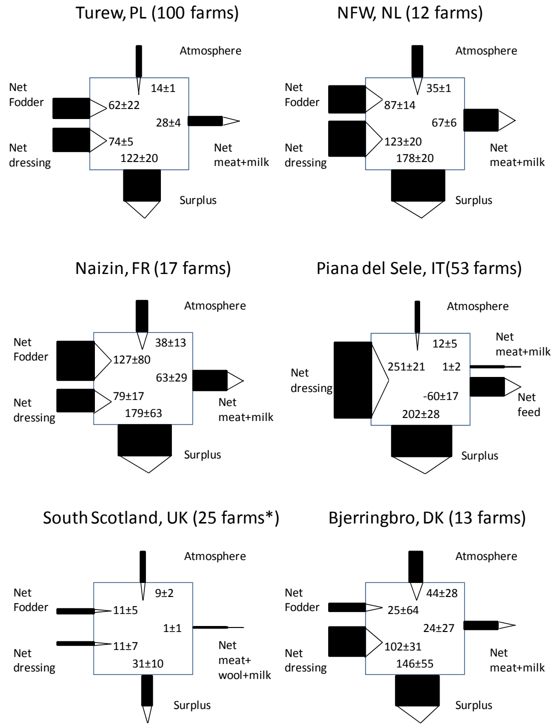

Fig. 4. N surpluses (kg N ha^{-1} yr^{-1}) for each of the six landscapes, partitioned into the Fig. 3 components of the N balance, including N fixed or deposited from the atmosphere, net N fodder import (or feed export if negative), net import of dressing in the form of fertilisers or manure, and the net export of milk and meat, also including N in eggs and wool (all with 95 % confidence intervals under the assumption of a normal distribution). * Excluding landless poultry farming in Scotland. If the landless poultry farming was included, the UK N-surplus would be the same size as in PL and DK, but with a much larger fodder import of around 300–400 kg N ha^{-1} yr^{-1}, a net manure export of around 150–200 kg N ha^{-1} yr^{-1} and a net export of eggs, meat, wool and milk of around 120 kg N ha^{-1} yr^{-1}.

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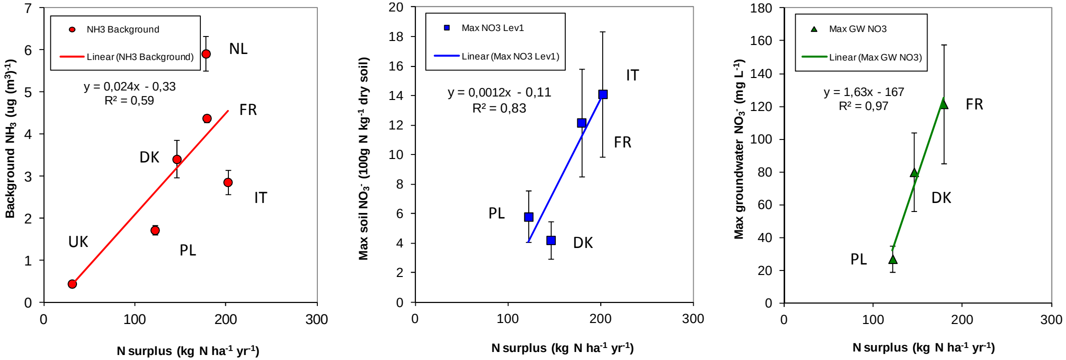

Fig. 5. Comparison of average landscape N surpluses with measured ammonia concentrations (lowest site-mean, left) and soil nitrate levels (maximum nitrate concentration measured in the A-horizon of any site, centre) and groundwater nitrate levels (maximum nitrate concentration measured at any site, right). Measurement uncertainty for ammonia was calculated from the standard errors of the individual monthly concentrations and for nitrate from the mean uncertainty for the concentration values where uncertainty was reported (±30 % for both soil and groundwater).

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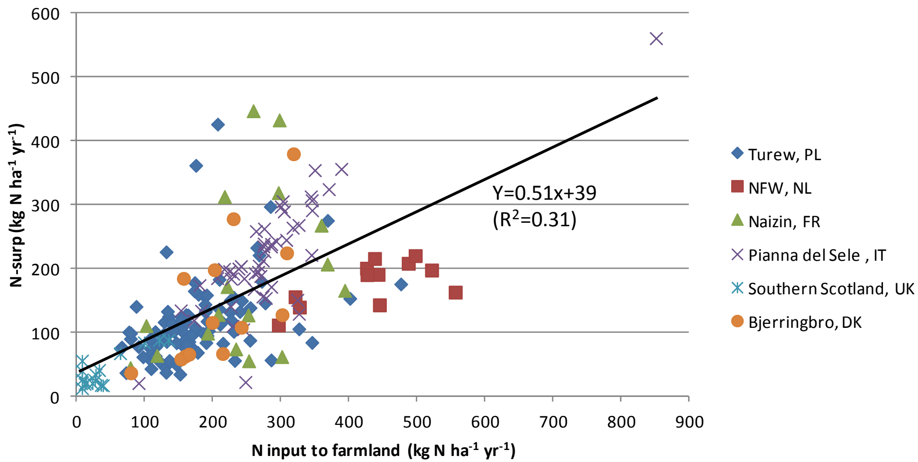

Fig. 6. Calculated farm N surplus (N-surp) as a linear function of N input to farmland, estimated as the average input per field area (total N input to each farm from fertilisers, manures, N-fixation and deposition from the atmosphere).

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