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Estimating nitrogen flows of agricultural soils at a landscape level – A modelling study of the Upper Enns Valley, a long-term socio-ecological research region in Austria

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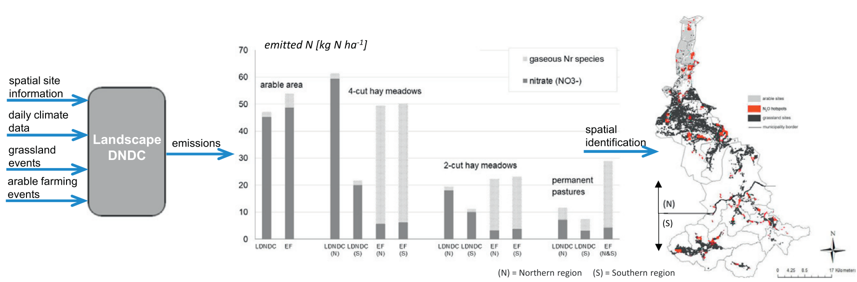
HIGHLIGHTS

- Biophysical soil modelling focusses on a complete balance of N flows to air and water.
- Comprehensive simulation of grassland and arable soils on a landscape scale
- Emission factor method overestimates environmental impacts of N_2O and NO.
- Focusing abatement measures towards identified hot-spots will maximize their effects.
- Approach offers easy linkages to dynamic decision models.

GRAPHICAL ABSTRACT



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ABSTRACT

This paper explores the fate of reactive nitrogen (Nr) on the landscape scale of present agricultural production practice on arable and grassland soils. We use the soil modelling tool LandscapeDNDC (landscape scale DeNitrification-DeComposition model) to quantify resulting flows of Nr distributed to the atmosphere, hydrosphere and the crops. Test area is a watershed in the Austrian Alps characterized by arable production in the low-lying areas and grassland in the mountains. The approach considers an overall budget of nitrogen, and determines the nitrogen use efficiency for individual crops and crop rotations, with average levels found at 85% for the arable area and 68–98% for the grassland areas. Modelled Nr flows are compared to the values resulting from the national emission factor (EF) method used for the Austrian emission inventory. For the arable part of the study region, the annual sum of released Nr emissions derived from LandscapeDNDC modelling is lower than the result of the EF method by about 13% (or 7 kg N ha^{-1}). Model results are lower also for other Nr species, yet nitrate leaching rates as well as ammonia emissions contribute a major share. For grassland areas, nitrate leaching values estimated by LandscapeDNDC greatly depend on local specifics and substantially exceed EF estimates. All other modelled Nr species are lower than the EF results. The model set-up allows to characterize spatially explicit effects of mitigation measures. As an example, we identify nitrous oxide (N_2O) hot spots in the study region, and we quantify the N_2O emission saving potential if focusing reduction efforts to such hot spots. Reducing fertilization of hot spots by half could remove 14% of N_2O emission for 5% less crop yield and a loss of grassland yield by <1% when extrapolated to the whole study area.

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1. Introduction

Developing an industrial method of fixing nitrogen (N) from its elements (Haber-Bosch process) converted a previously scarce plant nutrient into an abundant commodity. Since technological innovation allows the large-scale use of mineral fertilizer, intensification of agricultural production could be implemented globally (Erisman et al., 2008). This development may be regarded decisive to feeding the growing world population, but N now available in excess contributes to a range of adverse environmental effects such as nitrate leaching to water bodies, eutrophication, acidification of soils, emissions of air pollutants as well as greenhouse gases. Globally, about 50% of N in fertilizers and manure spread to agricultural areas is lost to the surrounding environment (Erisman et al., 2013; Sutton et al., 2011). The anthropogenic input of N to the biosphere substantially alters the earth system and leads to exceedance of the planetary boundaries (de Vries et al., 2013; Steffen et al., 2015).

When N enters soils in a reactive form (for instance as fertilizers, or via microbial fixation from the atmosphere), it serves as a substrate for multiple microbial processes and becomes available for plant uptake. Successful allocation of N to plants depends on environmental factors such as temperature, moisture, aeration or the availability of other substrates, such as carbon. Consequently, environmental conditions determine the fate of N and do not lead to desired plant growth only (Butterbach-Bahl et al., 2013; Isobe and Ohte, 2014; Signor et al., 2013). In soils, denitrification and nitrification are dominant processes to convert N species into environmentally harmful substances (Benckiser et al., 2015; Bouwman et al., 2013). Excessive N not used by plants leads to elevated ammonia volatilization (NH_3), nitrous oxide (N_2O) emissions to air and N run-off to groundwater and surface water as nitrate (NO_3^-) and ammonium (NH_4^+). Specifically looking at N_2O, agriculture currently accounts for 56–81% of gross anthropogenic emissions and several emission scenarios project a doubling of those emissions up to 2050 (Davidson and Kanter, 2014).

National emission inventories, established for the annual reporting to the *United Nations Framework Convention on Climate Change (*UNFCCC), most commonly use emission factor methods, developed by the Intergovernmental Panel on Climate Change (IPCC, 2006). For air pollutants, the respective methods are developed under the Convention on Long-Range Transboundary Air Pollution of the *United Nations Economic Commission for Europe* (UNECE) and published by the European Environment Agency (EMEP/EEA, 2013). These guidelines use widely available statistical information as input data allowing for comprehensive assessments. When using universal emission factors (tier 1) of those guidelines, variations in local conditions and management are not considered respectively. This approach consequently leads to high uncertainties connected to empirical inventories of agricultural soil N emissions (Milne et al., 2014; Winiwarter and Rypdal, 2001; Wójcik-Gront and Gront, 2014) and may lead to undesirable wrong assessments.

Today, biophysical soil models are primarily applied for 'local' investigations on a site basis, allowing to be validated by emission field-measurements. For instance, Rafique et al. (2011) evaluated management effects on nitrous oxide emissions from Irish grassland sites and Cui et al. (2014) assessed the biogeochemical effects of different management practices for a cropland site in northern China. Moreover, applications of soil models at 'regional' scale, typically applied for areal units indicating similar environmental conditions identified by GIS-data, are increasing (Delgrosso et al., 2005; Follador et al., 2011; Henseler and Dechow, 2014). A 'region' may be defined, for example, by climate and vegetation zones or by a national border. Benefits of the latter one result from data availability as statistic typically refer to. 'Landscape' commonly refers to a spatially heterogeneous area at scales of hectares to many square kilometres (Turner and Gardner, 1991) and is a fundamental trait of a specific geographic area including its biological composition, physical environment and anthropogenic or social patterns (Young, 2000). 'Landscape modelling' often associates with a nutrient transfer within geographic areas (i.e., N transfer by biogeochemical, atmospheric, or hydrological pathways, as has been simulated by Duretz et al., 2011; Klatt et al., 2017 and by Romero et al., 2016).

In this study, we investigate the exchange of N species between environmental compartments for a 'landscape' in Austria. This approach considers the C and N cycling within the domain, but not the physical transport between geographic points. Establishing comprehensive inventories on a landscape scale holds several advantages. Firstly, the modelling deals with agriculturally manageable units. Combining information on local site (as soil and climate variability) with the acquisition of spatially attributable management data leverages synergies, coming towards the need for research on a regional or landscape scale, especially in context to the necessity of adaption of agricultural practices to climate change in agriculture (BMLFUW, 2012; Olesen et al., 2011). Secondly, providing data with preferably high spatial and temporal resolution of agricultural process information and local conditions is a contribution to decrease uncertainties coming along with inventories on national or larger scales, since effects and responses of N management practices have been shown to be regionally and locally specific depending on interactions with soils, current climate and cropping systems (Dalgaard et al., 2012; Olesen et al., 2011). Thirdly, considering different kinds of land-use and management activities at hand is a key to address the environmental relevance of human activities. Gaube and Haberl (2013) as well as Booth et al. (2016) exemplarily established methods to translate socio-economic scenario assumptions into quantitative drivers providing input parameters for biophysical models. As the modelling study at hand processes management activities to specific plots, a combination with farm-based decision models (agent-based models) is feasible. This enables an integration of the social dimension and its decision-relevant, trans-regional framework.

Here we use the ability of process-based biophysical soil modelling to reflect and assess environmental effects of actual socioeconomic decisions on the N cycle. To evaluate the local management practice, we analyze nitrogen-use efficiency (NUE) for the produced agricultural goods. The full N balance of the respective products is obtained by applying the soil model. Nr flows, customized for the study region, are compared to results built upon the methods used for the Austrian national emission inventory. Finally, we discuss N_2O mitigation strategies on the landscape-scale assessed by the achieved modelling results.

2. Method

2.1. Study area

This study focuses on a study area, the so-called Upper Enns Valley, being part of the Long-Term-Socio-Economic-Research Platform (LTSER) "Eisenwurzen" (Mirtl et al., 2015). This area is a dedicated study region for basic ecosystem research (Environment Agency Austria, 2012), for biodiversity conservation research (Haberl et al., 2009) as well as for socio-ecological research. An integrated model, developed for this LTSER region, shows which impacts both internal and external factors have upon material flows, community structure and agriculture (Gaube et al., 2009).

The region extends over an area of >1400 km^2, covers 17 municipalities (status as of 2018) and is situated at the border of the Austrian provinces of Styria and Upper Austria. About 1070 km^2 are used for various types of agricultural production: 6% is used for arable area and permanent cultures, 18% is used as grassland and pasture, and 76% is managed as forest (data derived from national INVEKOS GIS data of 2014, and Corine Land Cover 2006 raster data (EEA, 2010)).

To date, the region experiences typical problems of marginalized rural areas such as declining agriculture, a lack of jobs, low incomes and creeping deterioration of infrastructure (Gaube et al., 2009; Stieber, 1998). Evolved socio-economic conditions cause reinforcing feedback leading to further migration or at least to rural depopulation.

Despite considerable political efforts to counter this trend, Austrian grassland and cropland areas are declining since the middle of the 20th century. Moreover, segregation of cropland agriculture, meat production, and milk-producing grasslands in different regions occurs, a phenomenon dramatically changing cultural landscapes, breaking up local nutrient cycles and increasing the volumes of freight transport (Krausmann et al., 2003). Scientific studies give attention to a large and growing separation between producing and consuming areas for biomass-based products (see, e.g., Erb et al., 2009; Gavrilova et al., 2010).

On the Austrian national level, from 1999 to 2010 > 1700 km^2 of the agricultural area have been abandoned (−0.5% per year). In the same period, managed forests increased by 1400 km^2 (+0.4% per year), croplands lost about 200 km^2 (−0.16% per year), and agriculturally used grassland showed a decrease of 4760 km^2 (−2.3% per year). >75% of these lost grassland areas are in alpine regions and highlands (BMLFUW, 2013a), to which the study region belongs.

2.2. Evaluation of nitrogen use efficiency

Among many options to evaluate nitrogen use efficiency (NUE) (see, e.g., Baligar et al., 2001), we present NUE in terms of gross NUE as a ratio of N removed from the fields by harvest divided by the sum of N input by fertilizer, manure, biological nitrogen fixation (BNF) and by atmospheric deposition. This calculation uses the same output/input flows as the OECD (2008) for their gross nitrogen budgets, only that we do not consider N losses during manure storage and manure application, and we also neglect seed and planting materials. Data sources correspond to soil model input data (see Section 2.4.2). The statistical yield of each crop is distributed among the single crop rotations (CR) via the achieved soil model results.

2.3. Calculation of Nr flows by the emission factor method

Emission factors (EFs) used here basically refer to those used for Austria's National Inventory Report (NIR) established by the Environment Agency Austria (2016). Following this method, the basic information to estimate arising N emissions is the amount of N added to soils (activity data), not accounting for application losses. We consider national values for gaseous losses during manure and mineral fertilizer application (17% and 4% of N applied to the field, respectively) accordingly. As in the NIR, we use universal EFs combined with activity data to estimate N_2O emissions following IPCC (2006). Indirect N_2O soil emissions (volatilized and re-deposited N and N losses through leaching) are not considered as they are also not covered by the emission results calculated by LandscapeDNDC. As a factor for NO_3^- leaching (FracLEACH), we follow the study by Eder et al. (2015), referenced in the NIR. This study determined national FracLEACH values for arable and grassland management separately. Respective factors are 0.254 kg NO_3^--N per kg N of added fertilizers and left crop residues for arable land, and 0.021 kg NO_3^--N per kg N of added manures for grassland. NH_3 emissions are estimated according to the CORINAIR methodology (EMEP/EEA, 2013) with national values for the content of total ammoniacal nitrogen (TAN) in manures (50% TAN in cattle liquid manure and 15% TAN in cattle farmyard manure; provided in the NIR as well). For NO conservative factors of 0.01 kg NO-N per kg animal manure N applied (Freibauer and Kaltschmitt, 2001) and 0.003 kg NO-N per kg mineral fertilizer N applied (EMEP/EEA, 2013) are used. N deposited by grazing animals is calculated according to the average N excretion rates on pastures, range and paddocks of Austrian cattle (data derived from Environment Agency Austria, 2016).

2.4. Calculation of N flows by LandscapeDNDC modelling

2.4.1. Software

A recent version of the LandscapeDNDC model (LandscapeDNDC version 1.3.4, download available under KIT, 2016), which is based on the DNDC (DeNitrification-DeComposition model of Li et al., 1992), is used to estimate land-use related N emissions for multi-ecosystems. The LandscapeDNDC model has been designed for the regional simulation of ecosystem C and N cycling and associated biosphere-atmosphere-hydrosphere exchange processes and is based on a generalized soil biogeochemical process description unifying the arable and the Forest-DNDC. For details see Haas et al. (2013).

2.4.2. Development of homogeneous spatial mapping units (HSMUs)

Applying the LandscapeDNDC model requires the description of the initial conditions of the soil and vegetation for each geographic unit as well as its boundary conditions during the simulation describing the weather conditions and the agricultural management. Fig. 1 illustrates respective data sources and the information.

For the comprehensive regional emission inventory, we delineate areas with common properties. Following the concept described by Leip et al. (2011), we call these areas “homogeneous spatial mapping units (HSMUs)”. The geospatial intersection is performed with the GIS-software ArcGIS, version 10.3.

2.4.2.1. Homogeneous spatial soil mapping units (HSSMUs). In a first step, we aggregate soil information to homogeneous spatial soil mapping units (HSSMUs). Parameters and scales used for the soil type aggregation were taken from a digital soil map of Austria (eBOD, see Fig. 1), developed by the Federal Research Centre for Forests (BFW). We identify HSSMUs as such when they had a specific value-range of three selected parameters in common. Selected parameters, essentially influencing nitrification/denitrification rates, were i) texture, ii) soil pH, and iii) humus content (Butterbach-Bahl et al., 2013; Stange et al., 2000). Further details on soil mapping units can be retrieved from Section 1 in the Supplementary material.

2.4.2.2. Spatial management information. Spatial records of management categories are intersected with the previously obtained HSSMUs (see Fig. 1). Area-specific accounts of crop production and animal husbandry are obtained from a database generated by the European Integrated Administration and Control System (IACS). The system has been established to provide background information on agricultural subsidies. In Austria, this system is known under the acronym INVEKOS (see Fig. 1). Again, further details on and geographic records of management categories can be retrieved from the Supplementary material, Section 2.

2.4.2.3. Spatial climate information. Spatial climate information consists of attributable topography (elevation), atmospheric deposition and concentration of N species, and local weather conditions. Elevation is derived from a freely available digital elevation model of Austria (oe3d, see Fig. 1) and ranges from 201 m to up to 2585 m (including alpine grassland regions). For each management category, a “typical” elevation is selected as the median for all HSMUs connected to the respective management. Atmospheric deposition and atmospheric concentration are derived from model results provided by the European Monitoring and Evaluation Program (EMEP, see Fig. 1). This information is provided in a 50 × 50 km grid over Europe. Averaged annual deposition densities (2011–2015) of the three grid cells, respectively covering the arable area, the southern grassland, and the northern grassland, are used. For the weather conditions, we use the information of three local weather stations, respectively assignable to the arable area, the northern grassland and the southern grassland.

2.4.2.4. Developed HSMUs and limitations. The intersection results in 12 HSMUs for the arable land and 144 HSMUs for grassland areas. Out of these 144 HSMUs, 36 HSMUs (or 7673 ha) covering alpine pastures are excluded from modelling since the model version used is not able to process soils with high organic matter soil content (starting from 6 to 8%) in an appropriate manner. Some areas are not attributed due to lacking soil data (see Supplementary material, Section 1). As a result, 90% (7121 ha of 7891 ha) of the reported arable land and 59% (13,696 ha of 23,043 ha) of the reported grassland are represented by the model results. We assume these areas to represent the total and accordingly extrapolated when displaying area (or sub-regional) totals. The "base map" as well as properties of major soil types and corresponding input data for the soil model are listed in the Supplementary material (Fig. S2, and Tables S3 and S4).

2.4.3. Daily model input data

For the simulation runs, HSMUs are connected to corresponding daily input data. LandscapeDNDC processes weather events and land farming events (i.e., seeding, tilling, harvesting, fertilization, or grazing) in daily time-steps. Simulation time is from 1st January 2011 to 31st December 2015.

2.4.3.1. Daily weather events. Information on temperature, precipitation, global radiation, relative air humidity, and wind speed are provided on a daily base by the Austrian Central Institute for Meteorology and Geodynamics (ZAMG, see Fig. 1). We use the information of three local weather stations, respectively assigned to the arable area, the northern grassland and the southern grassland. Climate indices of data recorded by the three weather stations are summarized in Section 3 in the Supplementary material.

2.4.3.2. Daily arable farming events. For the arable region, we refer to typical crop rotations (further expressed as CR), land management practices and fertilization practices obtained from A. Eder, personal information, following Eder et al. (2015). For the respective CRs, we process received information on i) the dates of seeding, harvest, tillage and fertilization (the specific months and if these events are carried out at the beginning, in the middle or at the end of the month), ii) the N fertilization amounts, iii) the kind of fertilizer (mineral fertilizer or farm manure), and iv) if harvest residues, such as straw, are removed by harvest. Respective practices base on guidance from the Austrian Chamber of Agriculture. Having regard to weather events, we synthesize the explicit dates for the simulated farming events. Three CRs typical for the region are compared, extending over three and four years, respectively. The CRs consist of grain maize – winter barley – catch crop – sugar beet or soya bean – winter wheat – catch crop – and starting again with grain maize, where CR 1 and CR 2 differ in the variation of sugar beet/soya bean only. CR 3 follows a sequence of grain maize – winter wheat – rapeseed – catch crop – and starting again with grain maize.

2.4.3.3. Daily grassland management events. Animal counts by municipality are derived from INVEKOS (see Fig. 1) and fertilization practice for grassland from the directive for proper fertilization (BMLFUW, 2006). Grazing periods and animal stocking rates assume Austrian principles in grassland practice (Steinwidder and Starz, 2015). Grazing duration is adapted to the average grazing time of Austrian livestock (data derived from Environment Agency Austria, 2016). Further details can be taken from the Supplementary material, Section 5.

2.4.4. Model calibration

Model plant growth is calibrated against district-based yield data. We approximate the area-weighted average outcome of different HSMUs to 100% of the recorded annual mean yield in the years 2011–2015. Exemplarily, for arable yields, Fig. 2 illustrates simulated and reported year-to-

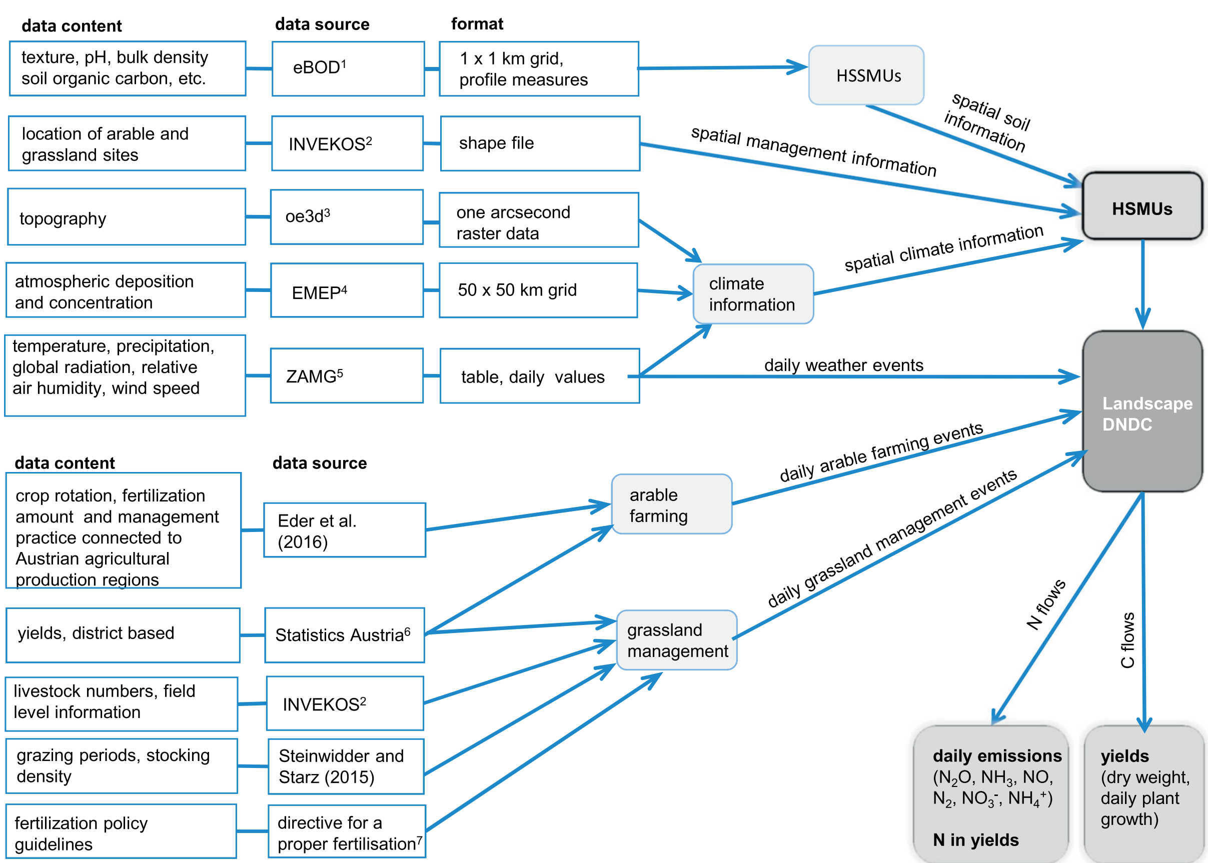

Fig. 1. Data sources and information flow for the calculation of N emissions arising from soils using the LandscapeDNDC model. [1]Digital soil map of Austria (eBOD) developed by the Federal Research Centre for Forests (BFW, 2007), [2]Integrated administration and control system (INVEKOS) coordinated by Agrarmarkt Austria (AMA, 2014), [3]digital surface model of Austria (oe3d, 2016), [4]The European Monitoring and Evaluation Program (EMEP, 2015), [5]Data of local weather stations, provided by the Austrian Central Institute for Meteorology and Geodynamics (ZAMG, 2017), [6]Austria's national statistics institute (Statistics Austria, 2016), [7]National guideline according to the Austrian ministry of life (BMLFUW, 2006).

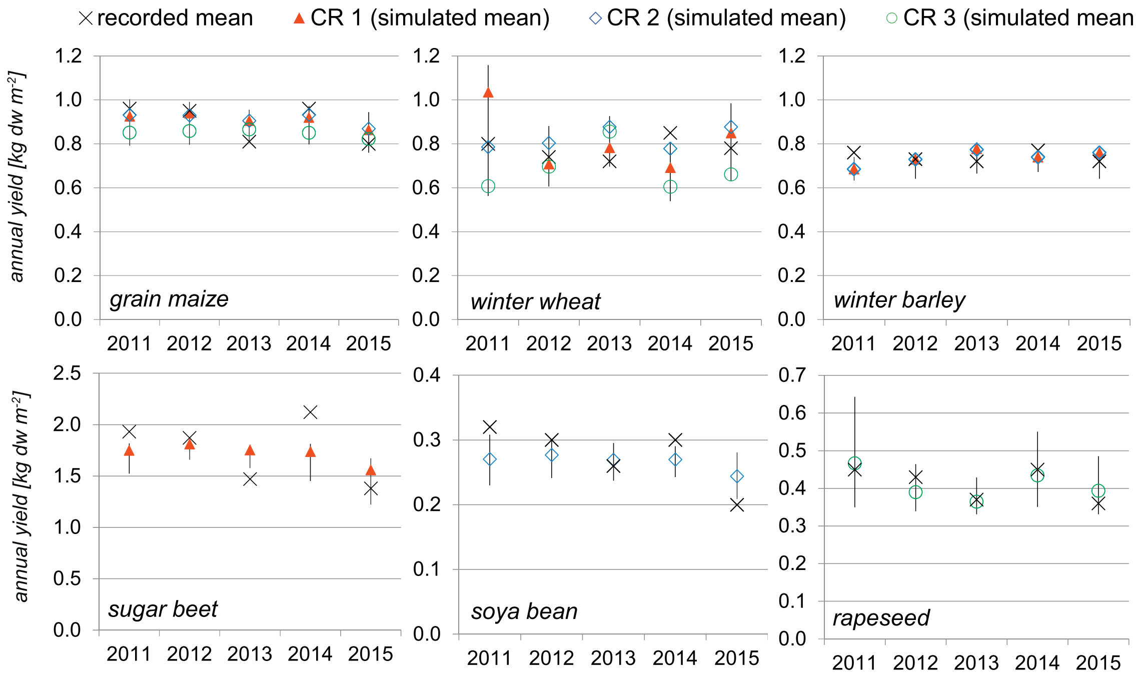

Fig. 2. Annual results of simulated arable yields compared to recorded yields (dw = dry weight). The black lines show the range of simulation results of the HSMUs (min to max). Plotted means are the area-weighted results.

year variations. Furthermore, we synchronize the model outcome with typical C:N values for the plant compartments corn and straw, as well as with corn:straw ratios at harvest according to literature. Adjusted LandscapeDNDC growth parameters, as well as the outcome of simulated corn:straw ratios and all year-to-year variations can be retrieved from the Supplementary material, Section 6.

2.5. Identification of N_2O hot spots and emission reduction potential

The model results obtained using LandscapeDNDC quantifies N fluxes for each N species. We extract fluxes of N_2O and yield of the different HSMUs to identify hot spots of N_2O emissions. As N_2O hot spots, we identify those agricultural HSMUs, which release twice as many N_2O emissions per unit yield (dry weight) compared to other local soil types. To evaluate emission saving potential for these areas, we carry out simulation runs with 50% reduced fertilization, and zero-fertilization for respective HSMUs.

3. Results

3.1. NUEs of the specific arable and grassland management activities

To characterize the fate of Nr, we take advantage of a widely used indicator that allows providing a more systematic viewpoint. The total flow of Nr emissions, in absolute units, indicate losses of N, not captured by agricultural products and the transfer of environmentally harmful substances from soils to the atmosphere and the hydrosphere. In contrast, NUE provides information about the relative utilization of additional N applied (EUNEP, 2015; Brentrup and Palliere, 2010; OECD, 2001). It is an indicator of the level of performance of the agricultural system expressed by the ratio between N outputs and N inputs.

3.1.1.1. Arable farming. Regional established NUEs for agricultural used soils in the Upper Enns Valley are 85% for the arable area on average. Calculated NUEs, connected to the single crops and CRs are listed in Table 1.

Comparison between CRs in arable areas shows that the highest NUE level was reached for CR 2, followed by CR 3 and CR 1. The most considerable difference occurs between CR 1 and CR 2 (76 vs. 95%). Decisive is the change in cultivating soya beans instead of sugar beets. The mineral-fertilized, sugar-producing plant achieves a NUE of only 21%, compared to the leguminous soya bean delivering high protein-levels (NUE is 104%). NUE for winter wheat (96–115%) is comparatively high because this is the only crop of which straw is removed from the fields at harvest. N in straw (37–43 kg N ha^{-1}) is considered as N output respectively.

3.1.1.2. Grassland management. NUE results of grassland areas in the Upper Enns Valley show a range from 68 to 98%. The individual values of the local management types can be retrieved from Table 1.

Table 1
Calculated NUEs of the different management types in the study region. The mean of the reference period 2011–2015 is given. As explicit spatial data for the single arable crop rotations (CR) is not available, we also present the mean of CR 1–3. Management types of grassland systems are connected to localized data. Quantities of the underlying N flows are shown in the supplementary material (Table S9).

	Management	Nitrogen-use efficiency (N_{output} / N_{input}) [%]
Arable area (mean of CR 1–3)		***85***
CR 1		***76***
	grain maize	83
	winter barley	78
	sugar beet	21
	winter wheat	113
CR 2		***95***
	grain maize	83
	winter barley	78
	soya bean	104
	winter wheat	115
CR 3		***84***
	grain maize	78
	winter wheat	96
	rapeseed	78
Grassland North	4-cut hay meadows	68
	2-cut hay meadows	82
	permanent pastures	91
Grassland South	4-cut hay meadows	83
	2-cut hay meadows	94
	permanent pastures	98

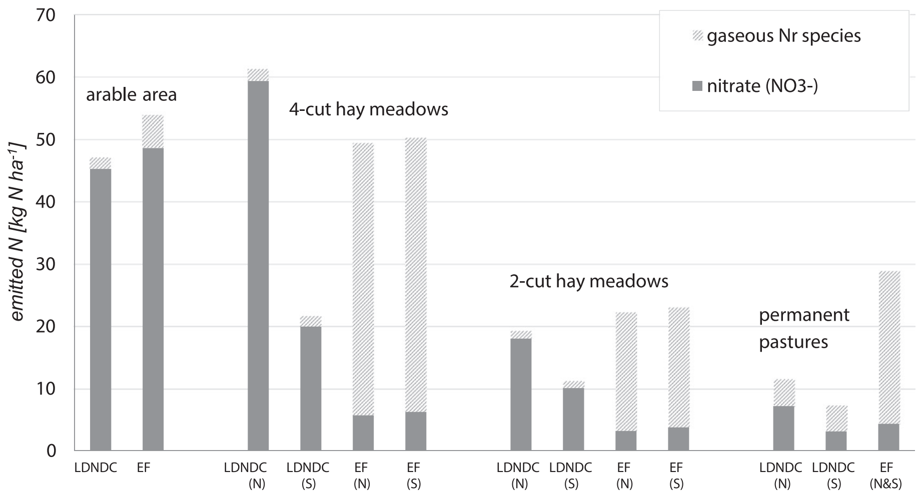

Fig. 3. Results of calculated N emissions by the two emission estimating methods (EF and LandscapeDNDC) in comparison (annual mean emissions from 2011 to 2015). For arable crops, the mean of CR 1–3 is illustrated. Fully coloured bars show the values of N leached as nitrate. Total columns (dark part and striped part together) represent the total N lost to the environment. For grassland management, (S) designates results for the southern part of the region, and (N) results for the northern part of the region. Note that the major share of NH_3 is lost during application. This flux is only covered in the EF method.

According to our calculations, grasslands in the South achieve higher NUEs than in the northern part of the test area. Permanent pastures achieve highest NUEs among the different management types. Figures close to 100% indicate a neutral soil N balance but also a high risk of N soil mining. NUE of permanent pastures is closely followed by NUEs of hay meadows, where a higher frequency of cutting is accompanied by lower NUEs.

3.2. Full N balance and established N flows of the study area

In consideration of a full N balance, the LandscapeDNDC model is used to provide information on different fluxes and species of N, both for grassland and arable land. Figs. 3 and 4 display the summarized results. Explicit numbers of emission values can be obtained from Table S8 in the Supplementary material. This table also provides also information on interannual variability which has not been further evaluated. For comparison, results obtained by applying the EF method are also shown.

3.2.1.1. Arable farming. For the arable region, the annual sum of released N emissions derived from LandscapeDNDC modelling is below the EF method result by about 13% (LandscapeDNDC: 47.12 kg N ha^{-1}, EF: 53.93 kg N ha^{-1}). Most of all, calculated NO_3^- leaching and NH_3 emission rates cause the disparity of results between the two applied methods. For these flows, estimates by LandscapeDNDC are lower than the ones by the EF method, especially regarding the absolute amounts (7%; or 3 kg N ha^{-1} less for NO_3^- and 98% or almost 3 kg N ha^{-1} less for NH_3). N_2O emissions are also below the EF results (22% below the EF estimates; or about 0.4 kg N ha^{-1}). Similar to NH_3, NO in LandscapeDNDC make up only a small share of emissions estimated by the EF method (83% less than the EF estimate; or about 0.36 kg N ha^{-1}). N_2 fluxes, accomplishing the N balance produced by LandscapeDNDC, account for 2.35 kg N ha^{-1}. This flow is not covered in the EF method.

3.2.1.2. Grassland management. Figures of N emissions from grasslands vary considerably depending on the calculation method applied but also depending on the evaluated management types. Furthermore, LandscapeDNDC computes considerable differences between the two separated grassland regions as well (see Figs. 3 and 4).

Particularly for NO_3^- leaching, LandscapeDNDC calculations substantially exceed EF results. The two separately treated grassland regions are exposed to high precipitation quantities (northern part: 1380 mm, southern part: 1217 mm). According to the model results, 61–73% of precipitation amounts percolates accompanied by high NO_3^- discharge. The most extreme difference occurs for the management type “hay meadows, four times cut” in the North, where NO_3^- leaching rates are computed ten times higher compared to EF results (59.40 vs 5.69 kg N ha^{-1}). For N_2O, LandscapeDNDC calculates lower emission rates arising from hay meadows and permanent pastures than the EF method does (46–69% of the EF amounts for hay meadows, 23–25% of the EF amounts for permanent pastures).

Regarding the two separately treated grassland regions, LandscapeDNDC indicates a three times higher response of NO_3^- leaching of northern grasslands compared to the southern ones. Next to high percolation rates, the comparatively low N uptake (or low NUE) of the northern grassland region enhances this N flow. In contrast, the EF method tends to calculate slightly higher emissions for the southern part of the study region than for the northern part. This tendency applies not only for NO_3^- but also for N_2O (10–17% more for NO_3^- and the same range for N_2O emissions; or about 0.6 kg N-NO_3^- ha^{-1} and about 0.3 kg N-N_2O ha^{-1} more).

Another issue, causing considerable differences, are estimated NH_3 flows. Here, different system boundaries of the two methods take effect. The EF method estimates NH_3 emissions including application losses. For slurry applications, 25% of applied N is assumed being volatilized as NH_3. For farmyard manure, the respective amounts are 11.85% (all factors from Environment Agency Austria, 2016). LandscapeDNDC sets system boundaries at N entering the soil, while NH_3 emissions largely occur from manure application before being integrated into the soil. As a result, LandscapeDNDC consistently provides much smaller values. For hay meadows, it ranges between 0.2 and 0.6% of related EF calculations. For pastures, the respective range is 14–15%.

Modelled NO emissions are below the EF method estimates as well (10–4.5% of the respective EF estimates). N_2 emissions, only provided by LandscapeDNDC, amount 1.80–5.83 kg N ha^{-1}.

3.3. N_2O hot spots and emission reduction potential

3.3.1. Identification of N_2O hot spots

Emission hot spots cannot be identified from the EF method, as they result from a combination of management practices, soil properties, and weather patterns. Here we use the potentials of soil modelling to quantify effects and expected impacts of measures taken. Agricultural HSMUs, releasing twice as much N_2O emissions per unit yield

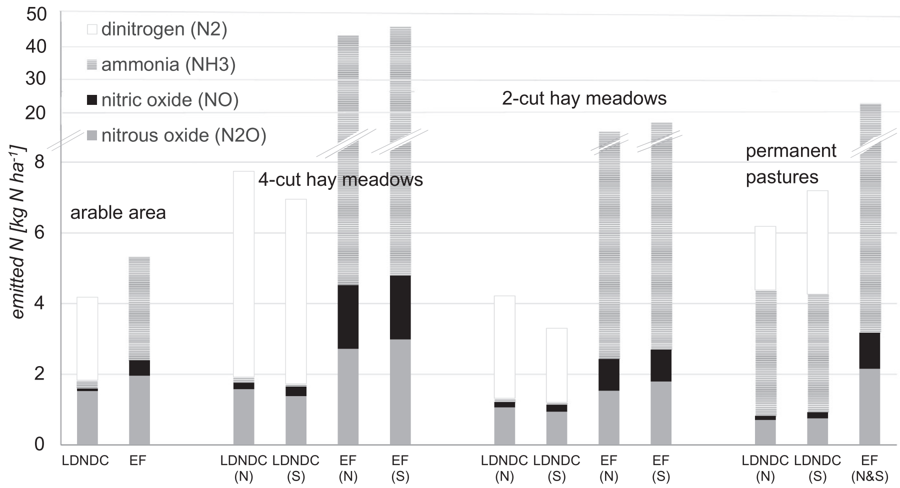

Fig. 4. Calculated results of N released as ammonia (NH_3), nitric oxide (NO) and nitrous oxide (N_2O) of the two emission estimating methods (EF and LandscapeDNDC) in comparison (annual mean emissions from 2011 to 2015). For arable crops, the mean of CR 1–3 is illustrated. Totals of the stacked columns cover the difference between N lost as nitrate and total N emitted (cf. lighter parts of the columns in Fig. 3), respectively. For grassland management, (S) designates results for the southern part of the region, and (N) results for the northern part of the region. Note that the major share of NH_3 is lost during application. This flux is only covered in the EF method, while the release of N_2 is only available from the LandscapeDNDC model.

(dry weight) compared to other local soil types are assigned as N_2O hot spots (see Fig. 5) and are spatially outlined in Fig. 6. HSMU soil type, the share within the individual management category as well as computed N_2O/yield ratios can be obtained from Table 2.

3.3.2. Emission reduction potential

As emission hot spots contribute significantly to overall emissions, these areas are likely to be the first target of optimized emission abatement attempts. The change of N_2O/yield ratios of the N_2O hot spot areas when reducing fertilizer inputs to 50% or even forego fertilization are displayed in Fig. 7 for the arable soil types and Fig. 8 for grassland HSMUs. The sum of N_2O mitigation potential for the different management categories extrapolated for the entire study area and the yield loss relating to it is visualized in Fig. 9.

The soil model predicts a reduction of 14% N_2O emissions when N_2O hot spots of the whole study region receive 50% N fertilizer. However, losses in yield arise as well (about 5% less crop yield and a loss of grassland yield by <1%). Calculations for a renunciation of fertilization show 21% N_2O savings, 8% lower yields for crops and a drop of grassland yields <1% as well.

3.3.2.1. Arable farming. At specific arable farming hot spots, the assumed 50%-fertilizer reduction leads to decreased N_2O emissions by 62–72% on average for the arable HSMUs (CR 1: 52–62%; CR 2: 44–66%; CR 3: 90–84%). For these HSMUs in total, N_2O emissions decrease from 3.59 to 1.06 t N_2O-N per year. Respective HSMUs make up only 10% (801 ha) of the total arable area but N_2O emission reduction amount to 21% of N_2O emissions released from arable soils as a whole under business as usual (BAU) (11.9 t N_2O-N per year released by the whole arable area under BAU; 2.53 t N_2O-N saving potential when applying 50% fertilizer to hot spot areas). Yields decrease by 4.9% (62.6 kt yield per year under BAU; 3.06 kt yield loss). Mineral fertilizer savings make up 56 t N per year. The simulations for the omission of fertilization results in 3.24 t N_2O-N savings and 4.74 kt yield loss.

3.3.2.2. Grassland management. The assumed 50%-fertilizer reduction leads to decreased N_2O emissions by 38–72% for the identified hot spots within the grassland HSMUs (65–70% for permanent pastures, 61–72% for hay meadows, four times cut, and 38–59% for hay meadows, two times cut). Adding up emissions released from all grassland hot spots, a decrease from 3.4 to 1.2 t N_2O-N per year is achieved. For the entire grassland region, the simulation indicates a potential of 12% N_2O emission saving, when N_2O hot spots (10%, or 2210 ha of the total grassland area) receive only half of the manure compared to the current situation (18.0 t N_2O-N per year under BAU; 2.16 t N_2O-N saving potential). According to our calculations, yields almost remain constant (107.6 kt yield per year - BAU; 5.56 tons yield loss for 50% fertilization). Respective results for the omission of fertilization are 3.16 t N_2O-N savings and 8.29 kt yield loss.

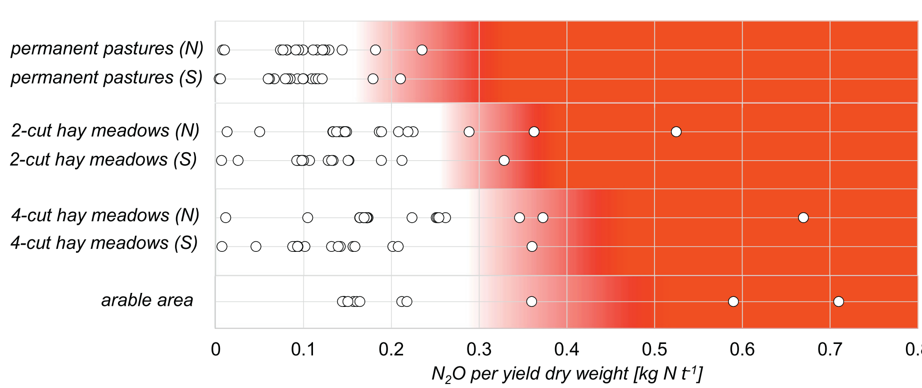

Fig. 5. N_2O emission rates per unit yield dry weight of the simulated HSMUs. Hot spot areas are shown with background shades.

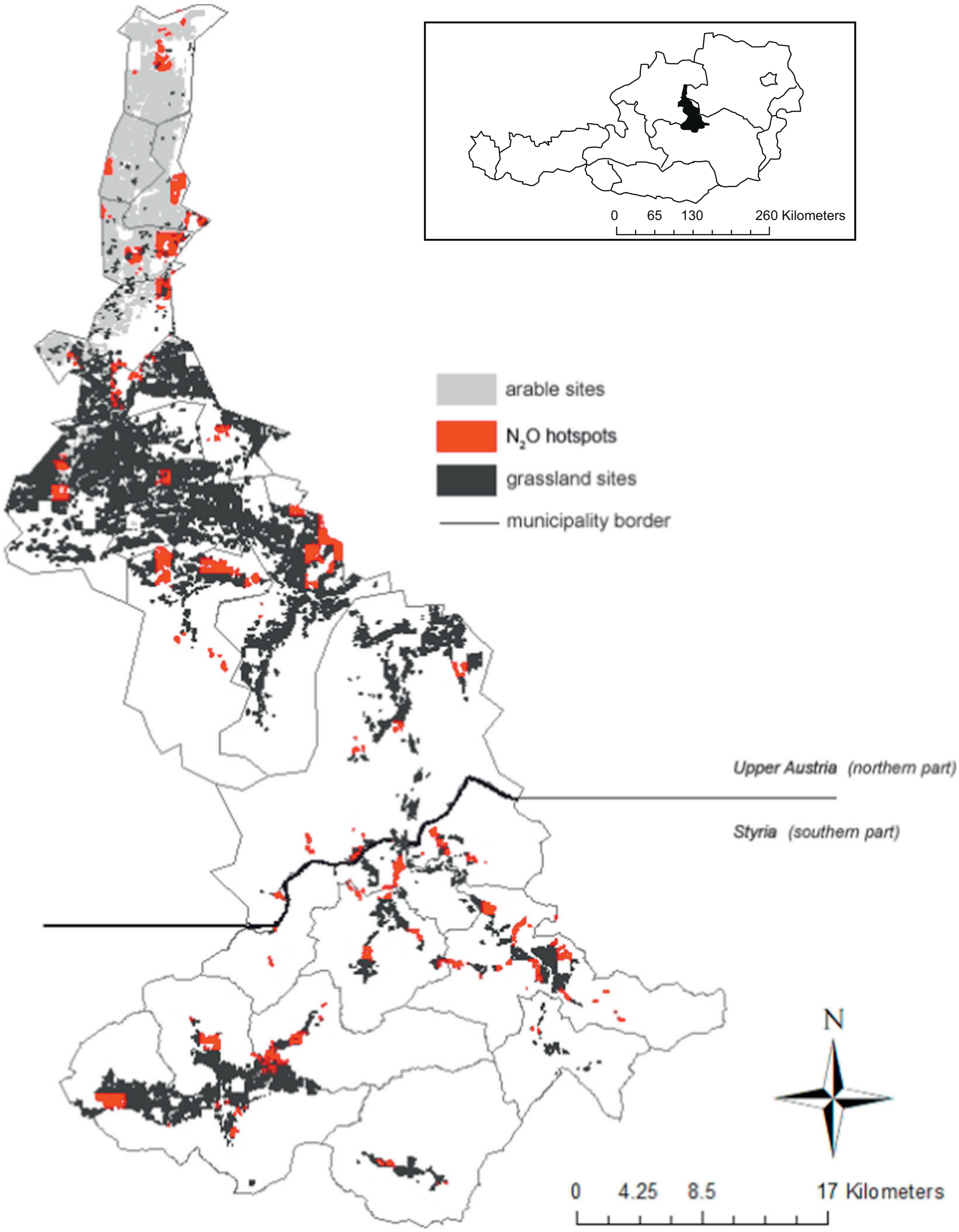

Fig. 6. Geographic outline of simulated area and of N_2O hotspots (red).

4. Discussion

4.1. NUEs of the specific arable and grassland management activities

NUE (the ratio of N output to N input) can be directly derived from the N budget as a functional value and allows for comparison between agricultural systems. Based on the national OECD N gross balance records for Austria's farming system, NUE is estimated ranging between 67 and 77% for the years 2010–2014 (data extracted from http://stats.oecd.org/, Feb. 2018). Mogollón et al. (2018) estimate 67.5–70% for the year 2005. These estimates do not distinguish between different kinds of land-use. Compared to the national values, regional calculated NUEs appear to be notably high. Regarding manure, again, there are different boundaries of the accounting methods (as within the EF method, compare to Section 3.3), primarily explaining divergences. The OECD gross balance calculation considers N in manure equal to N in excretion (gaseous N losses during accumulation, storage, and application are integrated) while the soil N budget calculation considers N in manure being equal to N entering the soil. Apart from this, OECD utilizes country totals for input quantities, here we are limited to adapt input values according to farming practice recommendations. Uncertainties of the regional established N budgets are mainly linked i) to the amount of applied manure and its N content, ii) to assumed BNF rates, and iii) to fertilization of single fields after high-yield harvests. An Austrian report on N balances on the level of groundwater bodies estimates the overall (additive) uncertainty for the established N budgets with ±28% (BMLFUW, 2013b). That study mainly uses the same N in- and output flows as we do.

Table 2
Soil types and corresponding emission rates of N_2O hot spot areas (highlighted in italics) compared to non-hot spot areas. N_2O per yield dry weight refers to area-weighted means.

Soil type (texture, pH, humus content)	Share [ha/ha]	N_2O per yield dry weight [kg N t^{-1}]		
			Avg. hot spot/other	Avg.
Arable area				
Loamy sand, slightly acid, medium humus	*0.04*	*0.73*	*0.63*	0.20
Sandy loam, slightly acid, medium humus	*0.06*	*0.62*		
Silty loam, slightly acid, rich in humus	*0.01*	*0.30*		
Silt, alkaline, medium humus	0.02	0.22	0.15	
Other soil types	0.85	≤0.21		
4-cut hay meadows (northern part)				
Loamy silt, neutral, rich in humus	*0.04*	*0.67*	*0.44*	0.20
Loamy silt, slightly acid, medium humus	*0.02*	*0.37*		
Loamy sand, alkaline, rich in humus	*0.01*	*0.35*		
Sandy loam, slightly acid, medium humus	0.05	0.26	0.19	
Other soil types	0.83	≤0.25		
2-cut hay meadows (northern part)				
Loamy silt, neutral, rich in humus	*0.03*	*0.52*	*0.42*	0.18
Loamy silt, slightly acid, medium humus	*0.06*	*0.36*		
Loamy sand, alkaline, rich in humus	0.02	0.29	0.16	
Other soil types	0.89	≤0.22		
Permanent pastures (northern part)				
Loamy silt, neutral, rich in humus	*0.03*	*0.24*	*0.20*	0.10
Loamy silt, slightly acid, medium humus	*0.05*	*0.18*		
Loamy sand, alkaline, rich in humus	0.02	0.14	0.09	
Other soil types	0.90	≤0.13		
4-cut hay meadows (southern part)				
Loamy silt, neutral, rich in humus	*0.14*	*0.36*	*0.36*	0.15
Loamy sand, alkaline, rich in humus	0.04	0.21	0.12	
Other soil types	0.82	≤0.20		
2-cut hay meadows (southern part)				
Loamy silt, neutral, rich in humus	*0.14*	*0.33*	*0.33*	0.14
Loamy silt, slightly acid, medium humus	0.06	0.21	0.11	
Other soil types	0.80	≤0.19		
Permanent pastures (southern part)				
Loamy silt, neutral, rich in humus	*0.14*	*0.21*	*0.19*	0.10
Loamy silt, slightly acid, medium humus	*0.17*	*0.18*		
Loamy sand, alkaline, rich in humus	0.03	0.12	0.06	
Other soil types	0.67	≤0.11		

4.1.1.1. Arable farming. Based on the established NUEs, we compare the common land uses in the region. Within the three arable CRs compared, the implementation of soya beans (CR 2) to arable crop rotations leads to the highest NUE. The harvest of soya beans is rich in protein (leading to high N output), and N input via plant-based N_2-fixation is comparatively low (i.e., soya beans compared to sugar beets in CR 1: crop yield C:N ratios are 9 vs 250; N input via BNF or as fertilizer is 111 vs 130 kg N ha^{-1} yr^{-1} for soya bean and sugar beet, respectively). N input incrementally achieved via BNF is successfully picked up by the plants. In contrast, classic "fertilization events" lead to higher losses of N, primarily as nitrate. However, specific local data on BNF is not available and had to be derived from external literature. The BNF rate of soya beans used for the N budgets here (111 kg N ha^{-1}) is based on the average value of a meta-analysis of 637 data sets according to Salvagiotti et al. (2008). In this meta-analysis, for 80% of these data sets the amount of N fixed was not sufficient to replace N export from the fields. The mean net soil N mining was −40 kg N ha^{-1} (this value equals the difference of N fixed in aboveground biomass minus N removed with grains; the value of our study: −22 kg N ha^{-1}). Other Austrian N balances for the agricultural sector use different values for BNF (e.g., 125 kg N ha^{-1} (Environment Agency Austria, 1998), or 65 kg N ha^{-1}(BMLFUW, 2013b)). Therefore, NUE for the cultivation of soya beans described in literature may differ distinctly but ultimately should tend towards a neutral or negative soil N balance.

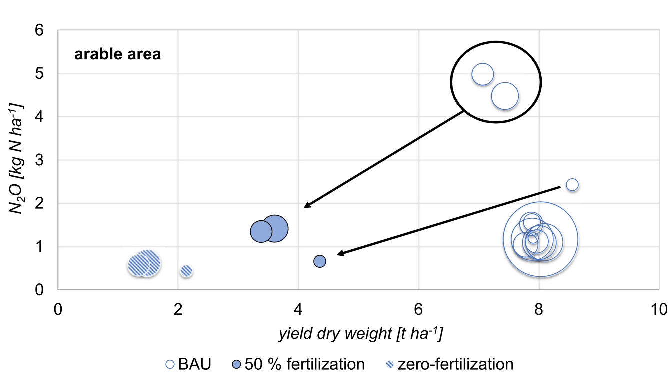

Fig. 7. N_2O emissions and yields in dry weight, for single soil types of the arable area. Delineated arrows point out the change of N_2O hot spots when they are treated with a 50% reduced fertilization compared to business as usual (BAU), or without fertilizer. Circle sizes are proportional to the relative areas of each soil type represented.

4.1.1.2. Grassland management. Regional NUEs of permanent pastures indicate a closed N balance holding risk for soil N mining (NUE close to 100%). Regarding this management category, we lack detailed input data. Local livestock head counts are available, but for grazed yields and grazing practice we need to rely on national information. Also, assuming the average grazing time of an Austrian cow (2 h per day during grazing season) might be a critical underestimation of the local grazing practice. Estimates based on averaged national data might lead to too

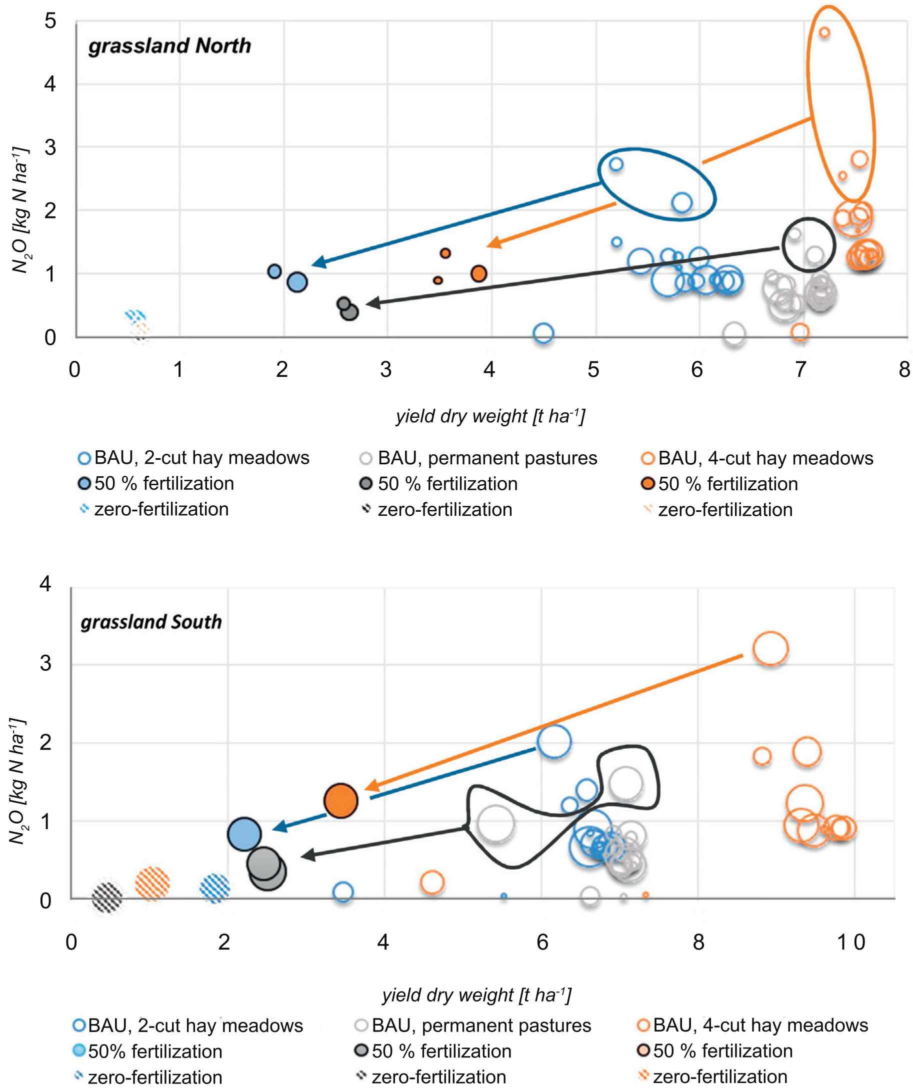

Fig. 8. N_2O emissions and yields in dry weight of single grassland HSMUs. The results of the three investigated management forms are outlined. Results of the northern region are illustrated in the upper graph, results of the southern region in the lower graph. Delineated arrows point out the change of N_2O hot spots when they are treated with a 50% reduced fertilization compared to business as usual (BAU), or without fertilizer. Circle sizes are proportional to the relative areas of each soil type represented.

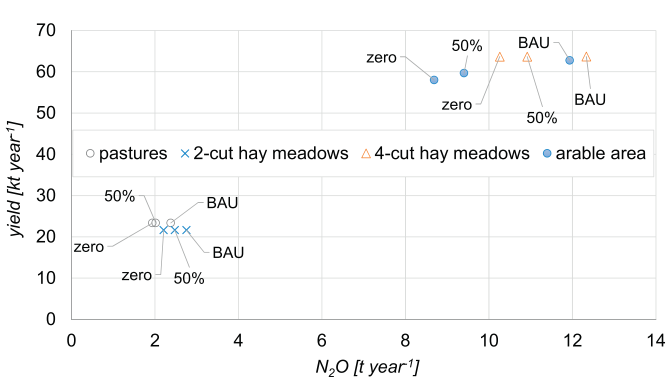

Fig. 9. Modelled N_2O emissions for the entire arable and grassland area of the study region. Data points indicated with "50%" show calculated N_2O emissions and yields when identified N_2O hotspots receive only half the amount of fertilizer compared to business as usual (BAU). Data points indicated with "zero", show emissions and yields with no fertilization of the N_2O hotspots, respectively. Large shares of the area (all that are not considered hotspots) would in both cases continue to receive BAU fertilizer levels.

short grazing times and subsequently too little N input by urine and dung of grazing animals. Single farmers report a 24-hours grazing duration in summer (and concomitant lower stocking rates during grazing), but reliable empiric data is not available yet.

Due to methodological reasons, NUE calculated includes the amount of the N input flow "N deposited by grazing animals as urine and dung" as prognosed by LandscapeDNDC. We process these flows using the model default settings. Simulated N input by grazing animals is rather low (12 and 5 kg N ha^{-1} for North and South respectively). However, the activity data we use for the EF method (9.8 kg N ha^{-1}), resting upon national average estimates as well, is in line with the outcome of the LandscapeDNDC default calculations.

Four times cut hay meadows achieve lower NUEs compared to twice cut hay meadows (see Table 1). According to the Austrian fertilization guidelines, the amount of recommended manure spreading differs by a factor of two between these two management categories but provided N seems to be not transferred to the plants in the same ratio. However, for all grassland management types, we assume the same BNF rates, which could substantially influence the presented results.

As for the arable crop soya beans, BNF values for grassland are derived from literature as well. Our estimate (50 kg N ha^{-1}; the same amount as used in BMLFUW, 2013b) correlates with the assumption of 15–20% clover when using BNF data of relevant Austrian literature (i.e., 50–350 kg N ha^{-1} yr^{-1} BNF of clover fields according to Freyer et al., 2005, or Starz et al., 2015). On the other side, estimated BNF rates could go up to 600 kg N ha^{-1} yr^{-1} for clover (BFW, 2011) or may be as low as 5 kg N ha^{-1} as an average value for intensively used grassland (Mogollón et al., 2018). One among many lysimeter field studies of grassland sites (Fu et al., 2017) shows N plant uptake succeeding organic manure input as well. This study also gives a good overview of comparable studies, indicating a high variability of the NUE from various grassland sites under different management conditions (starting from 50% up to 384% and more; calculated as a ratio of N in harvest/N input as fertilizer). In the end, BNF in grasslands depends on soil fertility, fertilization activities, and the intensity of use but the differentiation of BNF amounts regarding local practices is not feasible.

4.2. Full N balance and established N flows of the study region

4.2.1. Implementation of the full N balance

Soil modelling using LandscapeDNDC has taken a long development since first being published by Li et al. (1992). In this study, we specifically look at the N budget that can be achieved using LandscapeDNDC. This approach uses the regionally based N budget (or, primarily recorded in- and outflows of N) as an overall framework for calibration. This procedure is useful in particular when comprehensive areal field observations are unavailable or typically hard to afford. Since our results show consistency with comparable studies (see Section 4.2.2), it suggests a valid approximation to the N balance.

Compared to our approach, various studies invest a high effort in data acquisition to explicitly validate the N cycle. Typically, N flow field measurements of the compound of interest are used to calibrate or validate the model outcome (see, e.g., Rafique et al., 2011; Wang et al., 2012; W. Zhang et al., 2015, Y. Zhang et al., 2015). A number of studies make efforts to integrate observed hydraulic and thermal soil conditions, as inaccurate calculation of soil water content and soil temperature affects the dynamics of anaerobic-aerobic soil fractions, microbial growth, and activity, and thus N turnover via nitrification and denitrification as well as plant N uptake (Cui et al., 2014; Kim et al., 2015; Molina-Herrera et al., 2016; Saggar et al., 2004). Nevertheless, concerning the whole N balance, there is often insufficient reporting of crucially relevant data. Several studies proceed in maintaining the plant C:N ratio default settings of the model or typically do not report used ratios. Since the values in the model are set to the lower limits of observed plant C:N ratios, N in yields might be critically underestimated. Furthermore, articles dealing with leguminous plants, as with soya beans (Kim et al., 2015) or grassland species (Rafique et al., 2011; Saggar et al., 2004; Wang et al., 2012) neglect to report modelled N fixing values. Also, other relevant N input flows (such as N deposition values) are rarely reported. Subsequently, comparisons across studies are hardly feasible, even if the modelled system and overlapping parameters shall allow for it. Occasional studies, such as the one by Congreves et al. (2016), present a whole N balance for a region in Canada concerning climatic variability on total Nr losses. As we do, they highly recommend to include all N species to accurately determine the full N budget, trade-offs, and net N losses.

4.2.2. Established N flows of the study area

Major findings of our study refer to differences between the results of the empirical EF method (the standard method used for national reporting of greenhouse gases to the UNFCCC) and the soil model results. In contrast to the EF method, being based on N input only, the soil model approach takes advantage of incorporating the N budget as a whole while differentiating local soil and climate condition on high spatial and temporal scales. Results show differently estimated quantities of the total Nr flow between the two methods. Further, LandscapeDNDC splits the Nr flow into fluxes of individual N species (such as NO_3^- or N_2O), according to local conditions.

4.2.2.1. Arable farming. Simulated NO_3^- leaching rates, contributing the dominant share of lost Nr (96%) from the arable soils, are found to be in line with external measurements. The arable area of the study region belongs to a designated area for nitrate-observation (BMLFUW, 2014) where lysimeter field measurements (Office of the Provincial Government Upper Austria, 2014) determined a loss of 28–40% of N additions as fertilizer by leached NO_3^- (or 50–52 kg N ha^{-1} yr^{-1}). Simulated N losses as leached NO_3^- are in the range of 19–40% (29% on average) of N added as fertilizer, depending on crop and year. The measured level of nitrate leaching of cultivated soya beans is 18 kg N ha^{-1} yr^{-1} (our simulation: 20 kg N ha^{-1} yr^{-1}). Another LandscapeDNDC study (Molina-Herrera et al., 2016) determines N_2O fluxes and NO_3^- leaching rates for six arable sites across Europe. Results show N_2O emission factors (kg N_2O per kg fertilizer N input) ranging from 0.95 to 3.15% for arable sites (our study: 1.07%). NO_3^- leaching rates range from 6.36 to 88.41 kg N ha^{-1} (our study: 45 N ha^{-1} yr^{-1}).

Klatt et al. (2016) quantify regional parameter-induced uncertainties of the LandscapeDNDC model for N_2O and NO_3^- flows. That analysis covers >4000 polygons of German arable soils. Managed crops are rapeseed, wheat, and barley. Investigated parameter-uncertainties can be transferred to our study with high confidence. Results show a 50% likelihood range (the range between the 25th and the 75th percentile) for N_2O emissions from 0.46 to 2.05 kg N ha^{-1} yr^{-1}. Average direct N_2O emissions are 1.43 kg N ha^{-1} yr^{-1} similar to the result of our study (1.51 kg N ha^{-1} yr^{-1}). For leached NO_3^-, this study reveals significantly lower rates (LandscapeDNDC average value: 29 kg N ha^{-1} yr^{-1}, 50% likelihood range: 24.5 to 36.0 kg N ha^{-1} yr^{-1}) than our study (45.3 kg N ha^{-1} yr^{-1}).

Annual NO emission estimates derived by LandscapeDNDC (0.08 kg N ha^{-1}) are rather low both compared to the EF method (0.44 kg N ha^{-1}) and other estimates. Butterbach-Bahl et al. (2009) conducted a European inventory of soil NO emissions using a modified version of DNDC. According to this, our study region pertains to an area emitting 1.0–1.5 kg N ha^{-1} for the year 2000. Other approaches (e.g. Stehfest and Bouwman, 2006) estimate emission of similar magnitude. Independently, Molina-Herrera et al. (2017) recently coupled LandscapeDNDC to a specific submodule and validated modelled NO results successfully. However, that study does focus only on NO emissions and does not consider simultaneous calibration/validation for NO_3^- and N_2O fluxes as well, neither does the study report the full N balance. W. Zhang et al. (2015) report on the challenge of simultaneous calibration and validation of NO and N_2O fluxes and report consistent simulation results for the LandscapeDNDC model.

Validation studies of NH_3 volatilization for any biogeochemical model are even more scarce, due to a lack of observations at high resolution. Our LandscapeDNDC results show for the arable area emissions of 0.23 kg N ha^{-1} yr^{-1} on average. Low NH_3 emission results can be explained by the predominating neutral to acid soils in the study region limiting the volatilization of NH_3 in DNDC. However, the EF method estimates 2% of N input volatilized as NH_3. In our case, this converts to 2.93 kg N ha^{-1} yr^{-1}, which is significantly above the LandscapeDNDC results.

4.2.2.2. Grassland management. The large difference in calculated NO_3^- leaching for managed grassland between the LandscapeDNDC and EF method results (see Fig. 3) is remarkable. The deviation increases with the intensity of land-use (permanent pastures: 4.3 vs. 7.2 and 3.1 kg N^- ha^{-1} EF vs. LandscapeDNDC for North and South, respectively, hay meadows, two times cut: 3.2 vs. 18.0 kg N^- ha^{-1} EF vs. LandscapeDNDC for the North; 3.8 vs. 10.1 kg N^- ha^{-1} EF vs. LandscapeDNDC for the South; hay meadows, four times cut: 5.7 vs. 59.4 kg N^- ha^{-1} EF vs. LandscapeDNDC for the North; 6.3 vs. 20.0 kg N^- ha^{-1} EF vs. LandscapeDNDC for the South). This discrepancy leads us to suppose that i) the applied national nitrate leaching factor is inadequate for the grassland areas in the treated study region, and/or ii) BNF rates are overestimated, especially for the more intensive used grassland areas (see also Section 4.1).

Simulated nitrate leaching of grasslands strongly reflects NUE prescribed to the soil model. This relationship gets visible when looking at the regional differences between the two distinct grassland areas (see Fig. 3 for nitrate leaching rates and Table 1 for NUEs). The distinction of grassland into the two areas "North" and "South" constitutes i) a difference in recorded yields, ii) a spatial variation in soil characteristics, and iii) different weather events and timing of management events derived therefrom. The South shows higher NUEs for all grassland types compared to the North. Above all, this is due to generally higher achieved yields in the southern region (+21% more within the five simulated years). As a consequence of high NUE, the South shows a lower response regarding NO_3^- leaching in the soil model than the North which can be explained by the competition for the available N between plant growth and soil biogeochemistry. On the other hand, the amount of N_2O fluxes are comparable between the two regions.

Molina-Herrera et al. (2016) determine N_2O fluxes and NO_3^- leaching rates for two grassland sites (and six arable sites) across Europe. Results show N_2O emission factors (kg N_2O per kg fertilizer N input) ranging from 1.43 to 3.65% for the two grassland sites (our study: 0.70–1.40%). Determined NO_3^- leaching rates range from 0.26 to 69.00 kg N ha^{-1} (our study: 3.11 to 59.40 kg N ha^{-1}).

As discussed in the previous paragraph regarding arable farming, annual NO emission estimates derived by LandscapeDNDC (here, 0.12–0.27 kg N ha^{-1} for grassland) are rather low compared to other estimates.

Within LandscapeDNDC, application steps, i.e., any processes before manure-N enters the soil, are not considered (see also Section 3.2). In the results shown here, NH_3 emissions from manured hay meadows are even lower than those from arable soils, giving a hint that the competition for Nr within the model prefers N uptake by plant growth rather than biogeochemical N cycling. In contrast to modelled hay meadows, the gap between the EF method and LandscapeDNDC results for pastures is smaller (see Fig. 4) since NH_3 emissions from urine and dung deposited by grazing animals are considered to be within the boundaries of the applied soil model.

4.3. N_2O hot spots and emission reduction potential

Reducing environmental impacts often requires to apply dedicated abatement measures. For the greenhouse gas N_2O, agricultural measures are commonly connected with reduction of fertilizer input, as the emission factor approach (IPCC, 2006) assumes a strict proportionality between N input and emissions. Specific opportunities to reduce these N_2O emissions may arise from the fact that they are patchy and may be significantly increased in areas of specific soil properties. Such areas, here called N_2O hot spots, can be preferential target areas for any measures. With higher emissions, also the emission abatement potential increases in such areas.

While mineral fertilizer certainly is a cost factor in agriculture, it is applied with good reason to maximize yields. The reduced application, therefore, requires careful balancing towards minimizing overfertilization. Such measures do not necessarily require large technological effort. Review studies are available for cropping systems (Venterea et al., 2012) and for grassland-based agriculture (Li et al., 2013) that account for the effects of modifications of fertilizer application. With key aspects especially of the interaction of environmental parameters and fertilizer supply forms (rate, source, placement, timing) still not fully understood, guidance provided beyond minimizing inputs remains limited. Hence economic evaluations of N_2O abatement measures (Winiwarter et al., 2018) are merely looking into partial aspects of application limitations, such as "variable rate technology" that allows to adequately provide fertilizer additions based on sensors providing plant growth status on sub-plot scale.

LandscapeDNDC has in the past helped to understand management impacts in the emissions. Molina-Herrera et al. (2016) presented a mitigation study based on LandscapeDNDC and reported significant potentials for the mitigation N_2O emissions and NO_3 leaching from arable and grassland sites in Europe by optimizing the arable management (like altering timings of seeding, harvesting, tilling, fertilization as well as fertilization rates and the split of singe fertilization applications).

As a contribution to existing and suggested abatement measures, we target the advantages of identifying hot spots and of focussing fertilizer reductions to such area. As demonstrated in Figs. 7 and 8, achieved yields from N_2O hot spots are simulated in the same order of magnitude as those from "other" areas, but N_2O emission per ton yield is enhanced. Applying only 50% of the regular N fertilizer will result in a clear emission decrease, which extends even further when no fertilizer is applied. At the same time, also the crop yield will be reduced. While, within the hot spot area itself, the result is largely proportional (i.e., emissions per ton yield remain rather stable) an overall effect when tackling hot spots becomes visible. The model indicates, for a reduction of fertilizer application of 50% in hot spot areas only, an overall reduction potential of 14% of the N_2O emissions for the entire study area. In return, crop yield decreases by nearly 5%, and grassland yields decrease significantly <1% (see Fig. 9). Emission reductions appear most strongly for arable areas, but also for intensive (4-cut) meadows.

Hence, first modelling results for the test area demonstrate that not only hot spots can be identified, but they can also be used to devise specific emission abatement strategies. Of course, farmers would not automatically give up fertilizing just because their plot is prone to high emissions. But economic schemes can be developed that take account of such biophysical differences, and at the same time account for all the economic effects that LandscapeDNDC covers. That may include mineral fertilizer savings, but also manure (in this example: 75 t N) otherwise applied to grassland N_2O hot spots but under a regime of measures to be distributed to arable areas to substitute further mineral fertilizer.

Agri-environmental schemes are now widespread, their success in promoting sustainable attitudinal and environmental change is being increasingly questioned. European ecologists have observed that agri-environmental schemes are having only a limited impact on species richness and abundance for example (Kleijn et al., 2001; Whittingham, 2007). Political and scientific interest in overcoming the criticism has inspired a quest for innovative agri-environmental governance arrangements. One such innovation concerns a shift from top-down vertically organised governance arrangements towards regionally organised arrangements (Böcher, 2008; Kneafsey, 2010; Prager, 2015). Future application of modelling such as LandscapeDNDC on a regional level may contribute to specific programmes of agro-environmental subsidies (such as the Austrian Agri-environmental Programme ÖPUL) to support farmers in carrying the economic risk of yield losses when managing dedicated N_2O hot spots with limited fertilizer amounts.

5. Conclusions

This paper describes the implementation of the soil model LandscapeDNDC to a study area in the Austrian Alps. Results presented are consistent with existing model application and indicate valid implementation of the model. Our approach considers a complete set of N input sources, plant yield according to national statistics in cropland and pastures as well as the distribution of nitrogen compounds to products and the environment.

In contrast to previous studies, here a major focus is to establish a complete N budget. All substantial N input flows to LandscapeDNDC are adopted from external data sources and the individual N-uptake in crops derived from the model calibration to realistic yields. NUEs, derived from external statistics, are successfully implemented to the soil model. Emissions as nitrate, NH_3, N_2O, NO or molecular N_2 are consistent with the overall budget and thus confirm the robustness of the model. Individual flows of nitrogen compounds can be assigned comprehensively and even precisely to specific areas by the soil model in contrast to the national EF method.

Nevertheless, soil model results depend on robust and sufficient local data. We face this issue regarding BNF rates, and grazing practice, where we are forced to use general literature or national instead of regional parameters. That may lead to critical uncertainties. Thus, an establishment of local or at least regional data collection would be desirable.

The study identifies several shortcomings of the LandscapeDNDC model which led to further improvement of the model. The empirical DNDC based plant growth model was recently supplemented/replaced by a Farquhar photosynthesis-based approach reproducing the physiological diurnal pattern of grassland plant growth with stronger growth rates in spring compared to late summer. The new model accounts for local changes in soil pH due to manuring as well as for a new approach to consider the soil ammonium/ammonia equilibrium and NH_3 gas diffusion for slurry application. Additionally, the fixed grassland management with given dates for cutting can be replaced by a dynamic farmer approach where cutting occurs when given biomass thresholds are exceeded. This feature diminishes the effect of long periods with mature grassland with a vanishing competition for N boosting the biogeochemical N cycling. The findings of this study led to these recent advancements of the model which are still in the phase of validation and will be made available with the next major release of LandscapeDNDC.

LandscapeDNDC proves to be an appropriate tool to assess the impacts of agricultural management on the soil N cycle. Even when no suitable field measurement data is available and total amounts of specified N flows may not be validated in detail, the model allows approximating the quantity of N flows as well as the relative change, potential trade-offs and N net losses. Model results help to identify Nr emission reductions due to specific management changes.

We use the spatially explicit setup to model effects on N_2O flows. Identifying hot spots of N_2O emissions allows to assess the impacts of hypothetical abatement measures focusing on just these areas. Based on LandscapeDNDC results, reduced fertilizer application can be a reasonable measure to decrease N_2O emissions from agricultural used soils. If halving the amount of fertilizer just in hot spot areas, N_2O emissions can be reduced by 14%, mostly from cropland, while crop production would decline by only 5% (and grassland yields decrease by <1%).

Results from this biophysical model can be linked to economic evaluation and used in agro-environmental subsidy schemes. Moreover, application in scenarios on a landscape scale may take advantage of specific management options. As it may be difficult to assess the impacts of specific land-use appropriately, again using modelling approaches is a key to quantify potentials and consequences of implementation consistently. Suitable toolsets (agent-based modelling coupled with qualitative socio-economic data) exist and merely need interfacing for a future extension of the modelling scope.

Declarations of interest

None.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2019.02.071.

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