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 N₂O and NO.
• Focusing abatement measures towards identified hot-spots will maximize their effects.
• Approach offers easy linkages to dynamic decision models.

GRAPHICAL ABSTRACT

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 (N₂O) hot spots in the study region, and we quantify the N₂O emission saving potential if focusing reduction efforts to such hot spots. Reducing fertilization of hot spots by half could remove 14% of N₂O 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 (Bencsikere et al., 2015; Bouwman et al., 2013). Excessive N not used by plants leads to elevated ammonia volatilization (NH3), nitrous oxide (N2O) emissions to air and N run-off to groundwater and surface water as nitrate (NO3−) and ammonium (NH4+). Specifically looking at N2O, agriculture currently accounts for 56–81% of global anthropogenic emissions and several emission scenarios project a doubling of those emissions up to 2050 (Davidson and Kaner, 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; Wininwater and Rydpal, 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 different management practices for a cropland site in northern China. Moreover, applications of soil models at different management practices for a cropland site in northern China. Moreover, applications of soil models at different management practices for a cropland site in northern China.

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 (Gaub et al., 2009).

The region extends over an area of ~1400 km², covers 17 municipalities (status as of 2018) and is situated at the border of the Austrian provinces of Styria and Upper Austria. About 1070 km² 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 (Gaub et al., 2009; Steiber, 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 km² of the agricultural area have been abandoned (−0.5% per year). In the same period, managed forests increased by 1400 km² (+0.4% per year), croplands lost about 200 km² (−0.16% per year), and agriculturally used grassland showed a decrease of 4760 km² (−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., Balligar 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 N₂O emissions following IPCC (2006). Direct N₂O 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 NO₃⁻ 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 NO₃⁻-N per kg N of added fertilizers and left crop residues for arable land, and 0.021 kg NO₃⁻-N per kg N of added manures for grassland. NH₄ emissions are estimated according to the CORINAIR methodology (EMEP/EEA, 2013) with national values for the content of total ammoniacal nitrogen (TAN) in manure (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 (Freihauser and Kalschmitt, 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 (BMFLUW, 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-

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

The model results obtained using LandscapeDNDC quantifies N fluxes for each N species. We extract fluxes of N\textsubscript{2}O and yield of the different HSMUs to identify hot spots of N\textsubscript{2}O emissions. As N\textsubscript{2}O hot spots, we identify those agricultural HSMUs, which release twice as many N\textsubscript{2}O 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 N\textsubscript{r}, we take advantage of a widely used indicator that allows providing a more systematic viewpoint. The total flow of N\textsubscript{r} 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 soybean instead of sugar beets. The mineral-fertilized, sugar-producing plant achieves a NUE of only 21%, compared to the leguminous soybean 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\textsuperscript{-1}) is considered as N output respectively.

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.

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.
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. 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\(_3\))

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).

3.3. N\(_2\)O hot spots and emission reduction potential

3.3.1. Identification of N\(_2\)O 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\(_2\)O emissions per unit yield
(dry weight) compared to other local soil types are assigned as N2O 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 N2O/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 N2O/yield ratios of the N2O hot spot areas when reducing fertilizer inputs to 50% or even forgo fertilization are displayed in Fig. 7 for the arable soil types and Fig. 8 for grassland HSMUs. The sum of N2O 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% N2O emissions when N2O 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% N2O 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 N2O 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, N2O emissions decrease from 3.59 to 1.06 t N2O-N per year. Respective HSMUs make up only 10% (801 ha) of the total arable area but N2O emission reduction amount to 21% of N2O emissions released from arable soils as a whole under business as usual (BAU) (11.9 t N2O-N per year released by the whole arable area under BAU; 2.53 t N2O-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 N2O-N savings and 4.74 kt yield loss.

3.3.2.2. Grassland management.

The assumed 50%-fertilizer reduction leads to decreased N2O 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 N2O-N per year is achieved. For the entire grassland region, the simulation indicates a potential of 12% N2O emission saving, when N2O 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 N2O-N per year under BAU; 2.16 t N2O-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 N2O-N savings and 8.29 kt yield loss.

**Fig. 4.** Calculated results of N released as ammonia (NH3), nitric oxide (NO) and nitrous oxide (N2O) 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 NH3 is lost during application. This flux is only covered in the EF method, while the release of N2 is only available from the LandscapeDNDC model.

**Fig. 5.** N2O emission rates per unit yield dry weight of the simulated HSMUs. Hot spot areas are shown with background shades.
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.

Fig. 6. Geographic outline of simulated area and of N2O hotspots (red).
4.1.1. 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 N2-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.

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

<table>
<thead>
<tr>
<th>Soil type (texture, pH, humus content)</th>
<th>Share [ha/ha]</th>
<th>N₂O per yield dry weight [kg N t⁻¹]</th>
<th>Avg. hot spot/other Avg.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Arable area</td>
<td></td>
<td></td>
<td></td>
</tr>
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<td>Loamy sand, slightly acid, medium humus</td>
<td>0.04</td>
<td>0.73</td>
<td>0.63 0.20</td>
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<td>0.62</td>
<td></td>
</tr>
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<td>Silty loam, slightly acid, rich in humus</td>
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<td>0.30</td>
<td></td>
</tr>
<tr>
<td>Silt, alkaline, medium humus</td>
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<tr>
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**Fig. 7.** N₂O emissions and yields in dry weight, for single soil types of the arable area. Delineated arrows point out the change of N₂O 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.
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...
et al., 2005, or Starz et al., 2015). On the other side, estimated BNF observations are unavailable or typically hard to afford. Since our results suggest a valid approximation to the N balance. This procedure is useful in particular when comprehensive areal measurements of the compound of interest are used to calibrate or validate model results. Several studies proceed in maintaining the plant C:N ratio default set-values. Also, other relevant 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 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 Congresves 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 UNFCC) 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$_2$O), 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$_2$O fluxes and NO$_3^-$ leaching rates for six arable sites across Europe. Results show N$_2$O emission factors (kg N$_2$O 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$_2$O and NO$_3^-$ flows. Their 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 NO$_3^-$ emissions from 0.46 to 2.05 kg N ha$^{-1}$ yr$^{-1}$. Average direct N$_2$O 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 submodel 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$_2$O 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$_2$O 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$^{-1}$ yr$^{-1}$ EF vs. LandscapeDNDC for North and South, respectively, hay meadows, two times cut: 3.2 vs. 18.0 kg N ha$^{-1}$ yr$^{-1}$ EF vs. LandscapeDNDC for the North; 3.8 vs. 10.1 kg N ha$^{-1}$ yr$^{-1}$ EF vs. LandscapeDNDC for the South; hay meadows, four times cut: 5.7 vs. 59.4 kg N ha$^{-1}$ yr$^{-1}$ EF vs. LandscapeDNDC for the North, 6.3 vs. 20.0 kg N ha$^{-1}$ yr$^{-1}$ 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 NO3− 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 N2O fluxes are comparable between the two regions.

Molina-Herrera et al. (2016) determine N2O fluxes and NO3− leaching rates for two grassland sites (and six arable sites) across Europe. Results show N2O emission factors (kg N2O per kg fertilizer N input) ranging from 1.43 to 3.65% for the two grassland sites (our study: 0.70–1.40%). Determined NO3− leaching rates range from 0.26 to 69.00 kg N ha−1 (our study: 3.11 to 39.40 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, NH3 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 NH3 emissions from urine and dung deposited by grazing animals are considered to be within the boundaries of the applied soil model.

4.3. N2O hot spots and emission reduction potential

Reducing environmental impacts often requires to apply dedicated abatement measures. For the greenhouse gas N2O, 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 N2O 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 N2O 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 N2O 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 N2O emissions and NO3− 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 single 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 N2O hot spots are simulated in the same order of magnitude as those from “other” areas, but N2O 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 N2O 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 N2O 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 N2O 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,
mating the quantity of N advancements of the model which are still in the phase of validation and contrast to the national EF method. Identifying hot spots of N2O emissions allows to assess the impacts of geochemical N cycling. The mature grassland with a vanishing competition for N boosting the biodegradation 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 N emission reductions due to specific management changes. We use the spatially explicit setup to model effects on N2O flows. Identifying hot spots of N2O 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 N2O emissions from agricultural used soils. If halving the amount of fertilizer just in hot spot areas, N2O emissions can be reduced by 14%, mostly from cropland, where 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.

**Acknowledgements**

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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.

**References**

EUNEP, 2015. Nitrogen Use Efficiency (NUE) - an Indicator for the Utilization of Nitrogen in Agriculture and Food Systems. Wageningen University, NL.

ons nitrogen fluxes from European farmlands with DNF–EUROPE. Environ. Politi

Europe. European Summary Report (Project Report Task 1) of the EU-Congrt
Acted FAIR-CT96-1877 Biogenic Emissions of Greenhouse Gases Caused by Arable

Original title: “Futter- und Körnerleguminosen im biologischen Landbau.” Österrei
Agrarverlag, Leopoldsdorf: AVBUC.

Fu, G., Grange, R., Wang, B., Butterbach-Bahl, K., Kiese, R., 2017. Impacts of climate and

Gaube, V., Haberl, H., 2013. Using integrated models to analyse socio-ecological system
dynamics in long-term socio-ecological research – Austrian experiences. In: Singh,
Habert, Ch., Mertl, M., Schmid, M. (Eds.), Long Term Socio-


Gavirova, O., Jonas, M., Erb, K., Haberl, H., 2010. International trade and Austria’s livestock

Europe. European Summary Report (Project Report Task 1) of the EU-Congrt
Acted FAIR-CT96-1877 Biogenic Emissions of Greenhouse Gases Caused by Arable

Web references


