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N$_2$O emissions and NO$_3^-$ leaching from two contrasting regions in Austria and influence of soil, crops and climate: a modelling approach

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Abstract National emission inventories for UN FCCC reporting estimate regional soil nitrous oxide (N$_2$O) fluxes by considering the amount of N input as the only influencing factor for N$_2$O emissions. Our aim was to deepen the understanding of N$_2$O fluxes from agricultural soils, including region specific soil and climate properties into the estimation of emission to find targeted mitigation measures for the reduction of nitrogen losses and GHG emissions. Within this project, N$_2$O emissions and nitrate (NO$_3^-$) leaching were modelled under spatially distinct environmental conditions in two agricultural regions in Austria taking into account region specific soil and climatic properties, management practices and crop rotations. The LandscapeDNDC ecosystem model was used to calculate N$_2$O emissions and NO$_3^-$ leaching reflecting different types of vegetation, management operations and crop rotations. In addition, N input and N fluxes were assessed and N$_2$O emissions were calculated. This approach allowed identifying hot spots of N$_2$O

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emissions. Results show that certain combinations of soil type, weather conditions, crop and management can lead to high emissions. Mean values ranged from 0.15 to 1.29 kg N_2O–N ha^{-1} $year^{-1}$ (Marchfeld) and 0.26 to 0.52 kg N_2O–N ha^{-1} $year^{-1}$ (Grieskirchen). Nitrate leaching, which strongly dominated N-losses, often reacted opposite to N_2O emissions. Larger quantities of NO_3^- were lost during years of higher precipitation, especially if winter barley was cultivated on sandy soils. Taking into account the detected hot spots of N_2O emissions and NO_3^- leaching most efficient measures can be addressed to mitigate environmental impacts while maximising crop production.

Introduction

Despite the topic having been a focus for several decades now, the reduction of greenhouse gas (GHG) emissions has not lost its actuality. On the contrary, this issue is more prevailing than ever since we are confronted more and more with extreme climatic conditions and their severe consequences for agricultural production (Lobell and Gourdji 2012). Besides this, other ecosystem services and human health are also seriously affected (Fowler et al. 2013). The cause of this development can be, among others, attributed to the excessive release of reactive nitrogen into the environment from the agricultural sector. However, N fertilization is crucial to feed the global population (Steffen et al. 2015) and too little N input causes systems to be limited and stressed. On the other hand, too much N input will likely lead to N losses to the environment (Galloway 1998). They range from eutrophication to global acidification and the loss of stratospheric ozone (Gruber and Galloway 2008). Nitrous oxide (N_2O) is one of the most important GHG contributing to climate change since it is the main source of stratospheric NOx, which is significantly involved in the stratospheric ozone depletion (Heincke and Kaupenjohann 1999). Nitrogen is lost from agricultural soils via nitrate (NO_3^-) leaching and the gaseous release of N_2O, ammonia (NH_3) and nitric oxide (NO). Especially N_2O emissions have been the focus of many studies and the rising awareness induced efforts to estimate N_2O emissions from agro-ecosystems with different approaches and methods.

The average N surplus on agricultural land in Austria is estimated at around 34 kg ha^{-1} $year^{-1}$ (Umweltbundesamt 2015). Compared to other countries, this value is rather low. This development is partly due to the rising awareness in the agricultural sector to produce crops with higher nitrogen use efficiency (NUE) (Amon et al. 2014; Küstermann et al. 2010). In general, over the last 20 years Austrian agriculture reduced its GHG emissions by approx. 11% (Amon et al. 2014). However, the amount of fertilizer N application may vary largely depending on the crop grown, with values of 210 kg ha^{-1} $year^{-1}$ and more (BMLFUW 2006).

The quantification of N_2O emissions from soils is still difficult since various processes are involved in the complex interaction of N_2O production, consumption and release from soils (Butterbach-Bahl et al. 2013). Moreover, microbial activity, soil physical properties, nutrient availability, water content and movement as well as land management, climate, and vegetation influence these processes (Li 2000; Schmidt et al. 2000; Haas et al. 2013) where N_2O is predominantly caused by the microbial processes of nitrification and/or denitrification (Firestone and Davidson 1989). Under aerobic conditions nitrification is the main source of NO and N_2O release (Li 2000) whereas denitrification is promoted by denitrifying bacteria under anaerobic conditions (Li 2000; Butterbach-Bahl et al. 2013). Soils play a predominant role in the wide range of processes involved in N cycling as they are the principle location of reactive nitrogen (Nr) transformations and main site of denitrification back to N_2 (Fowler et al. 2013).

The N_2O soil flux itself is determined by N_2O production, consumption, transport and emission from the reacting system. In general, movement of gases in soils is based on molecular diffusion and advection (Heincke and Kaupenjohann 1999). Factors like soil porosity, temperature, moisture and clay content have to be taken into account when calculating the diffusion rate of N_2O in the soil matrix (Li 2000).

The quantity of N_2O accumulated in soils depends on the amount, quality and availability of fertilizer applied (Heincke and Kaupenjohann 1999). Thereby,

soil permeability is also an important factor, since the gas is mainly transported via macropores whereas management and cultivation methods influence the natural soil structure, water distribution and nutrient availability as well as the microbiological processes (Heincke and Kaupenjohann 1999). Besides the water content (Li et al. 2000; Butterbach-Bahl et al. 2013) also temperature exerts influence with regard to gas diffusion, solubility of N_2O (Heincke and Kaupenjohann 1999) and microbiological activity. Denitrifying and nitrifying bacteria are active over a wide temperature range (-2 to 60 °C) and formation of N_2O increases until the optimum temperature of 35–40 °C (Schmidt et al. 2000). Nevertheless, high peaks of N_2O fluxes may occur during thawing as a result of N_2O formation under frozen top soil layers (Wolf et al. 2012) or driven by microbial metabolism (de Bruijn and Butterbach-Bahl 2010).

However, despite many attempts it is still difficult to get exact estimates of N_2O emissions due to the complexity of the N cycle itself. Like other trace gasses, N_2O emissions have large temporal and spatial fluctuations (Li 2000). As stated in Haas et al. (2013) this is due to the fact that the microbial processes are strongly influenced by environmental factors and agricultural management operations.

N_2O emissions are assessed by on site measurements (e.g. research station in Scheyern in South Germany, Peter et al. 2011). However, field measurements often do not measure the whole year nor entire regions with all their heterogenic environmental and management influencing factors (Butterbach-Bahl et al. 2004). Modelling approaches are therefore used in addition to measurements. Li (2000) points out the challenges of modelling trace gas emissions since these emissions evolve from different sources and are produced and consumed simultaneously in the soil where various environmental variables and reactions are involved. The advantage of calculating N_2O emissions from soils with process-based biogeochemical models lies in the possibility to simulate processes and the gas flux for different terrestrial ecosystem types (Haas et al. 2013). A better understanding of the systems can be achieved thereby providing the opportunity to develop site specific strategies for improved NUE (Haas et al. 2013; Molina-Herrera et al. 2016). Butterbach-Bahl et al. (2004) and Molina-Herrera et al. (2016) compared observed and simulated N_2O emissions and showed that the simulated

N_2O emissions were well within the span of the observed N_2O emissions. DNDC was first established in 1996 by Li and since then extensively advanced and validations have increased the reliability of the model (Gilhespy et al. 2014; Molina-Herrera et al. 2016). Measurement data provide the basis to develop DNDC with the aim to extrapolate results from site to regional and even global scale (Li 2000). The process-based DNDC model is able to scale ecosystem processes from site to regional simulation domains (Haas et al. 2013) and has therefore the potential to calculate regional inventories of N losses from agricultural soils (Butterbach-Bahl et al. 2004). As stated in Haas et al. (2013) results of LandscapeDNDC were compared to measured annual N_2O emissions of different ecosystems (forests, arable and grassland site) proving that LandscapeDNDC is able to simulate N_2O emissions across differing ecosystems. Recent validation studies of Molina-Herrera et al. (2015, 2016) have proven the capabilities of the model to predict the N cycle in agroecosystems.

Since nitrate losses to the groundwater represent a potential source of environmental problems in intensive farming systems, our study also considers NO_3^- leaching. Nitrate is very mobile in soils and according to Küstermann et al. (2010) leaching may occur for the NO_3^- quantities that are not organically fixed (Klatt et al. 2017).

Within our study the overall aim was to model site specific N_2O emissions and NO_3^- leaching losses from soils considering soil properties, climate, crop rotation and management, and to identify potential hot spots and hot moments. The following hypotheses were proposed: (1) sites in regions with less precipitation (Marchfeld, MF) release less N_2O than sites in regions with higher precipitation (Grieskirchen, GK). (2) The release of N_2O emissions is influenced by crop rotations and management (e.g. N-input). (3) Higher nitrate leaching occurs in lighter soils and N_2O emissions are enhanced in heavy soils.

Materials and methods

LandscapeDNDC

The model used in this study is the LandscapeDNDC model version 0.36.1 (Haas et al. 2013). It is a process-based model to simulate the biogeochemical C and N

cycling in forest, arable, and grassland ecosystems at site and regional scale. It combines submodules for plant growth, soil and vegetation micrometeorology and water cycling as well as detailed routines for physico-chemical-plant and microbial C and N cycling and exchange processes with the atmosphere and hydrosphere of terrestrial ecosystems. LandscapeDNDC is based on a generalization of the soil biogeochemistry of the agricultural DNDC (DeNitrification-DeComposition) and Forest-DNDC (Li et al. 2000) and is based on 3 major soil carbon pools: Recalcitrant, labile and very labile soil carbon, each with an adjacent litter pool. Additionally, soil carbon is considered in microbial biomass and dissolved organic carbon. Each carbon pool has a proper nitrogen pool along defined via its carbon-to-nitrogen ratio. Inorganic nitrogen compounds e.g. like NH_4, NO_3^- and others are considered directly. The soil biogeochemistry module simulates biomass decomposition, nitrification, denitrification, chemo-denitrification as well as transport processes such as gas diffusion and solute transport via water fluxes for various compounds. The plant growth module is based on the DNDC approach, simulating three different plant growth stages for arable systems: Shooting, flowering and maturing based on a temperature degree sum approach. Recently a Farquhar-photosynthesis based approach became available for LandscapeDNDC as well (Kraus et al. 2015) but was not available at project start and therefore not used. At the end, LandscapeDNDC simulates the ecosystem carbon and nitrogen balance for various ecosystem types (forest, arable and grassland) including yields, GHG emissions and nutrient losses to surface waters on site and regional scale (Haas et al. 2013). Since the LandscapeDNDC model concept is processes based, Molina-Herrera et al. (2015, 2016) has proven its applicability across different climate zones, soil types and management systems for forest, arable and grassland ecosystems. Other studies e.g. for South Korea (Kim et al. 2014) focusing on yields, N_2O emissions and NO_3^- leaching or for soil NO emissions for Saxony, Germany by Molina-Herrera et al. (2017) confirm the model's accuracy and usability.

Selected sites

Different sites in two contrasting regions of different agricultural practice and environmental conditions

were selected to calculate annual N_2O losses from agricultural soils in Austria (Amon et al. 2014). Marchfeld (MF) is an area of 900 km^2 in the North-East of Austria characterised by fertile soils, mainly chernozems, and a dry climate with a mean annual precipitation of 525 mm and a mean annual temperature of 10 °C (long term average 1971–2000). The second region, Grieskirchen (GK) has a size of 250 km^2 and is located in the centre of Upper-Austria. Fertile cambisols and luvisols are predominant and the precipitation reaches 890 mm and the temperature 8.5 °C (long term average 1971–2000). Both regions are intensively used by agricultural farming.

Model input data

In this study we present typical agricultural land by selecting predominant soil types (predominant by covering area) and crop rotations of the major crops. Each site is exhibiting a homogeneous soil type and climatic condition. Data on vegetation, meteorology and human activity were generated on municipality level. Soil parameters were site specific. The required input parameters from several data providers in Austria were gathered to establish a database of model input data for the two selected regions. Input data include daily temperature and precipitation (see Table 1), soil type, texture, bulk density, organic carbon content (C_{org}), pH and hydrological properties (see summary in Table 2). In particular data were provided by BMLFUW (Federal Ministry for Agriculture, Forest, Environment and Water: Invekos – Integrated Administration and Control System of the EU; crop distribution/land use data base), UBA (Environmental Agency Austria: BORIS soil data base), BAW (Federal Agency for Water Management) and BFW (Austrian Research Centre for Forests: eBOD soil data base). For most crops management data were provided by AGES (Austrian Agency for Health and Food Safety: management data base) and supplemented with official data and expert knowledge of agricultural advisors. The application rates of N fertilizer were obtained from AGES in accordance with N demand recommendations (BMLFUW 2006) and supplemented by expert knowledge of the Chamber of Agriculture of Lower Austria. The timing for management operations (ploughing, planting, application of N fertilizer, harvest) was provided by AGES. Regarding the agricultural management, details on the

Table 1 Precipitation and temperature values of the two regions Marchfeld (MF) and Grieskirchen (GK) from 2006 to 2011

Year	Marchfeld		Grieskirchen	
	Sum of precipitation (mm)	Mean annual air temperature (°C)	Sum of precipitation (mm)	Mean annual air temperature (°C)
2006	574	13	973	11
2007	715	14	1008	13
2008	639	14	863	12
2009	681	14	1065	12
2010	791	12	883	11
2011	503	14	801	12

Table 2 Soil type, texture and physico-chemical properties for the two sites Marchfeld (MF) and Grieskirchen (GK)

Site	Clay (%)	Silt (%)	Sand (%)	C_{org} (%)	pH	Soil type	Wilting point (mm m^{-3})	Field capacity (mm m^{-3})	Bulk density (g/cm^3)
MF 1*	21	60	19	1.5	7.3	zL	214	394	1.4
MF 2*	33	52	15	2.7	7.2	zCL	120	474	1.2
MF 3*	28	48	24	3.1	7.4	L	100	474	1.2
MF 4*	33	49	18	3.4	7.5	zL	84	373	1.4
MF 5*	14	27	59	0.6	5.8	sL	30	127	1.5
GK 1**	19	71	10	3.3	4.5	zL	250	396	1.3
GK 2**	21	64	15	1.3	5.5	zL	140	262	1.5
GK 3**	13	45	42	1.2	5.2	L	200	349	1.4
GK 4**	23	65	12	1.4	6.7	zCL	250	405	1.3
GK 5**	21	62	17	1.2	5.7	zCL	270	417	1.3

Soil types: *zL* silty loam, *zCL* silty clay loam, *L* loam, *sL* sandy loam

*MF 1–5: soil types investigated in Marchfeld **GK 1–5: soil types investigated in Grieskirchen

average crop rotation, crop type, dates of seeding and harvest, tillage, fertilization and irrigation were generated to represent average conditions across both sites. Remaining process parameters were used as provided by the LandscapeDNDC parameter library. Daily weather data were generated with the climate interpolation and generation tool DAYMET (Thornton et al. 1997) for both sites based on Austrian wide climate station data of the ZAMG (Zentralanstalt für Meteorologie und Geodynamik). Daily weather data (minimum and maximum temperature and precipitation) for the years 2006–2011 were interpolated to cover the projected regions. Precipitation ranged from 500 to 790 mm per year (mean 640 mm) in MF and from 800 to 1070 mm per year (mean 940 mm) in GK. Mean annual air temperature between 2006 and 2011 ranged from 12 to 14 °C (mean 13 °C) in MF and from 11 to 13 °C (mean 12 °C) in GK. Regarding the years considered in this project, especially precipitation was above the long term average values (Table 1).

Information on soil characteristics obtained from BFW and BAW include soil characteristics like soil type, texture, pH, SOC, bulk density (Table 2).

The Invekos data base allowed a calculation of total area of crop land as well as the share of each crop covered. The arable land in Austria has a size of around 1,330,000 ha. The cultivation of winter wheat (19%), corn (16%), winter barley (7%), silage corn (6%), soya (5%), summer barley (4%), triticale (4%), sugar beet (3%) and rapeseed (3%) cover the largest share. Several kinds of crops are grown in the chosen regions, and the most important crop types were taken into account, covering at least 50% of the cultivated agricultural land. The main crop types of MF and GK with their average N fertilizer rates applied are shown in the table below (Table 3).

Cultivation and management practices differ between MF and GK. Corn is predominant in the hilly Western area of GK, and winter crops are predominant in the flat terrain of MF in the North-East of Austria. Irrigation takes place for sugar beet ($40 \, l \, m^{-2}$ in May, $35 \, l \, m^{-2}$ in the middle of August, $40 \, l \, m^{-2}$ at the end of August and $40 \, l \, m^{-2}$ in mid-September), onion ($30 \, l \, m^{-2}$ in the beginning of May, $30 \, l \, m^{-2}$ at the end of May and $40 \, l \, m^{-2}$ in August), and for corn ($35 \, l \, m^{-2}$ in July and $35 \, l \, m^{-2}$ in August).

Crop rotations

Two cash crop rotations (CR) (Fig. 1a, b) were constructed to represent crop cultivating characteristics of both areas. Input regarding the formation of the CR was gathered from crop production experts and from Austrian stakeholders. Their advice was considered practical and in line with the state of the art (Amon et al. 2014). Relevant crops in the two regions are winter wheat (WIWH), sugar beet (BEET), barley (BARL), corn (CORN), onion (ONIO), winter barley (WBAR), rape seed (RAPE), silage corn (SICO), legume-hay (LEHA) and the catch crop mustard (MUST).

In the set up, the first two years (2004–2005) were used as lead time to get the same starting situation for each soil. For each simulation year, the CR were offset by one year forming the set of crop rotations (CR1–CR7), each starting with the subsequent crop (e.g. MF CR1 starts with WIWH followed by MUST, BEET, BARL, CORN etc.; MF CR2 starts with BEET followed by BARL, CORN etc.). By simulating the entire simulation time span 7 times for 6 simulation years and finally aggregating the simulation results, artefacts generated by the crop rotation (different levels of yearly N input) were compensated. The two fist years were regarded as spin up in order to stabilize modelled soil C and N dynamics. This enables to generate more reliable estimates of mean soil N_2O emissions (Butterbach-Bahl et al. 2004). Five different soil types were chosen per region and 7 simulations runs were performed for each of the soils which amount to a total of 70 simulation runs.

Important parameters which affect ecosystem N cycling and associated N trace gas emissions are: rates and timing of fertilizer application, crop specific parameters (e.g. optimal biomass yield, residues left on the field after harvest), vegetation phenology, nitrogen efficiency, and other information on agricultural practices such as timing of seeding, harvest or tilling (Haas et al. 2013).

Model validation

Available data of crop yields provided by AGES from the two selected regions were applied to simulate realistic crop yields and to validate the model. In general LandscapeDNDC has been validated for plant growth, soil respiration and trace gas emission (Kraus et al. 2015; Molina-Herrera et al. 2015, 2016) and NO_3^- leaching (Dirnböck et al. 2016). Site-specific input data of climate, soil, vegetation, and farming practices have to be provided to be able to simulate the processes on cultivated land (Molina-Herrera et al. 2016). During the project time LandscapeDNDC was
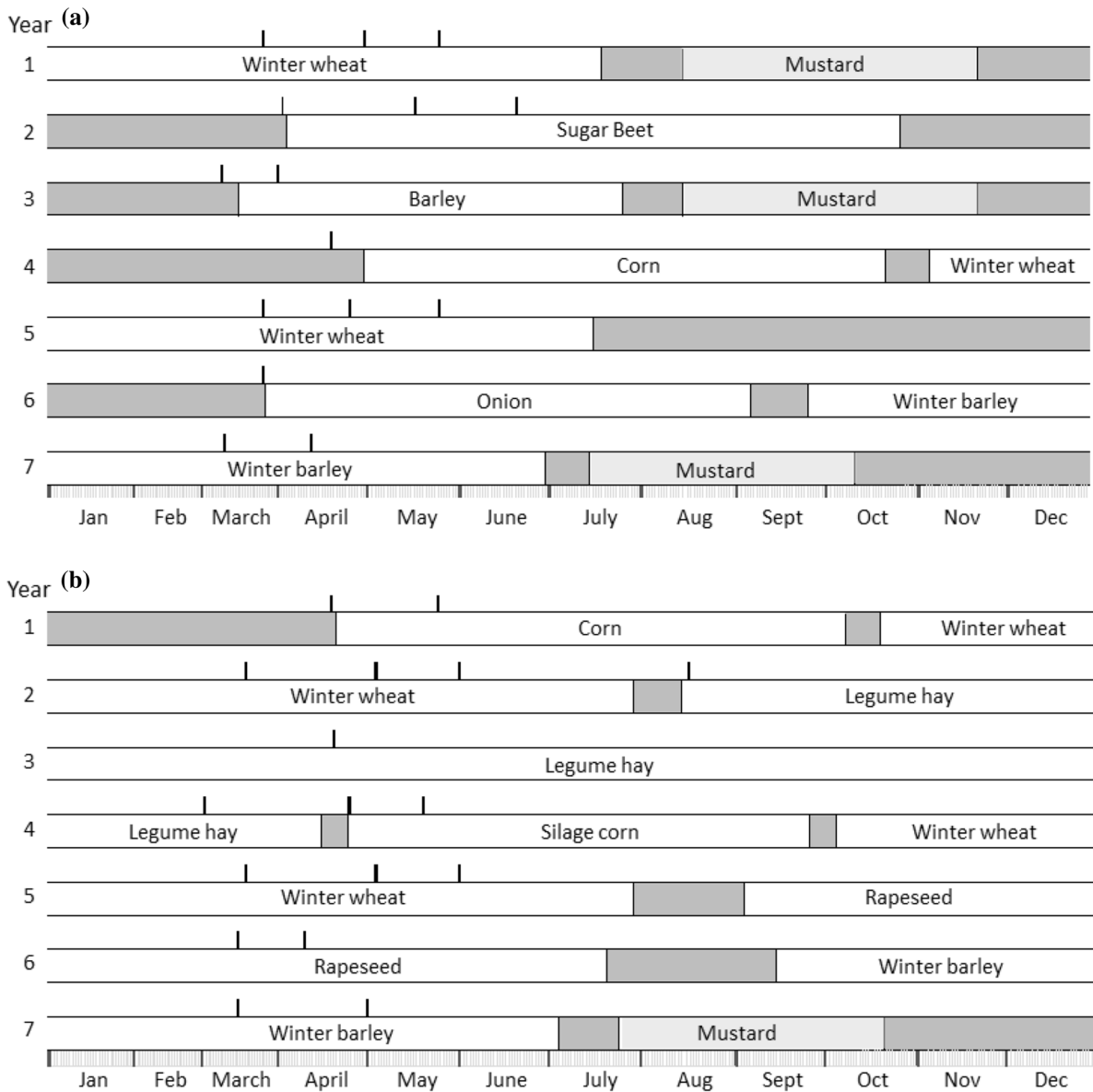
Table 3 Crop type and amount of N fertilizer applied in Marchfeld (MF) and Grieskirchen (GK), Austria

Crop type	MF	GK
	N fertilization (kg N ha^{-1})	
Winter wheat (WIWH)	39 + 38 + 39	60 + 50 + 34
Sugar beet (BEET)	45 + 35 + 30	–
Barley (BARL)	55	–
Corn (CORN)	150	90 + 66
Onion (ONIO)	130	–
Winter barley (WBAR)	55 + 55	65 + 60
Rapeseed (RAPE)	–	80 + 70
Silage corn (SICO)	–	90 + 90
Legume-hay (LEHA)	–	20 + 30 + 10
Mean	112 ± 29	136 ± 38

Fig. 1 Crop cultivation and fertilization management for the crop rotation 1 in MF (**a**) and GK (**b**). Timing for fertilizer applications are marked as black dashes. Mustard (MUST) is representing the catch crops. Grey parts are times of no cultivation (NONE)

further developed (e.g. some parameters were adjusted e.g. at WIWH grain/straw proportion). Process parameters (as explained in Molina-Herrera et al. 2016) were validated for plant growth only. LandscapeDNDC produced comparable results with regard to measured versus simulated crop yields and showed no significant difference between the two regions (Fig. 2). Of all crop types a strong correlation was found between measured and simulated yield ($R^2 = 0.85$ and 0.94) for

GK and MF, respectively. Furthermore observed NO_3^- leaching (kg N ha^{-1} year^{-1}), N yield (kg N ha^{-1} year^{-1}) and dry weight yield (kg ha^{-1} year^{-1}) values from lysimeter stations (Eder et al. 2015) near GK have been compared to simulation results calculated with LandscapeDNDC (Fig. 3a–c). The model is capable to reproduce the observed yield biomass and nitrogen content as well as the substantial high NO_3^- leaching losses due to the site specific

Fig. 2 Measured versus simulated crop yields in MF and GK
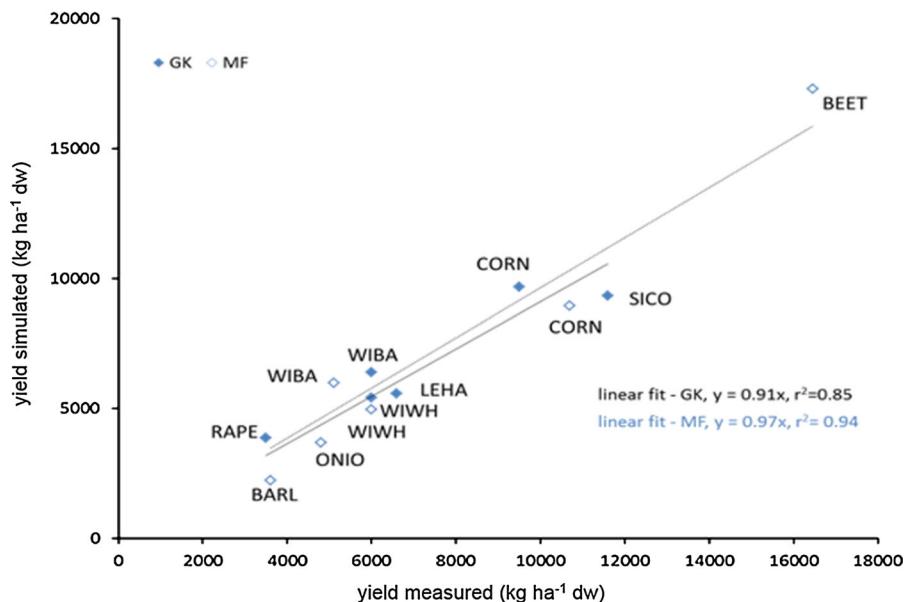
sandy soils (underestimations occurred for all soy bean cultivations while overestimation occurred for a superior high corn cultivation with high observation data uncertainties). Simulated annual N_2O emissions were less than 0.5 kg N_2O–N ha^{-1} $year^{-1}$ due to the strong nutrient leaching and the consequently low substrate availability.

Statistical analyses

Daily and/or annual fluxes of N_2O and leaching losses of NO_3^- were tested for normal distribution and variance homogeneity. We then used either the Procedure ANOVA (Student-Newman Keuls test— SNK), or, if these assumptions were not met, we used the Wilcoxon test. The effects of parameters like location, year, crop, and crop rotation on N_2O emissions or NO_3^- losses were investigated. SAS Enterprise Guide V 9.1 was used for all analysis. Significance level was $p < 0.05$ unless otherwise stated.

Results

N_2O fluxes

Soil N_2O fluxes differed substantially in space and time depending on climate (precipitation and

temperature), soil and management practices. Comparing the simulation results for the two chosen regions across all soil types (5 different soils per region) and all crop rotations (6 years), statistically significant different fluxes were found ($p < 0.001$). Mean annual fluxes were 0.53 and 0.37 kg N_2O– N ha^{-1} $year^{-1}$ at MF and GK, respectively with significant differences ($p < 0.05$) between the years (SNK) at GK (Table 4), and significant (SNK) differences between soil types (Figs. 4a, 5a) at both sites. Mean annual fluxes ranged from 0.24 (MF 5, 2006) to 0.82 (MF 2, 2007) kg N_2O–N ha^{-1} $year^{-1}$ and from 0.28 (GK 2, 2006) to 0.52 kg N_2O–N ha^{-1} $year^{-1}$ (GK 1, 2007).

Marchfeld

N_2O emissions ranged from 0.16 to 1.82 kg N_2O N ha^{-1} $year^{-1}$ in Marchfeld and were significantly different between soil types (Fig. 4a, b).

Soil types MF 1 and MF 5 (0.42 ± 0.08 and 0.33 ± 0.11 kg N_2O–N ha^{-1} $year^{-1}$, respectively), both displaying a $C_{org} < 2\%$, showed significantly lower N_2O emissions compared to MF 2–4 (0.68 ± 0.43, 0.65 ± 0.14, 0.62 ± 0.46 kg N_2O– N ha^{-1} $year^{-1}$, respectively). For MF 4 highest emissions were simulated (0.29 to 1.82 kg N_2O– N ha^{-1} $year^{-1}$).
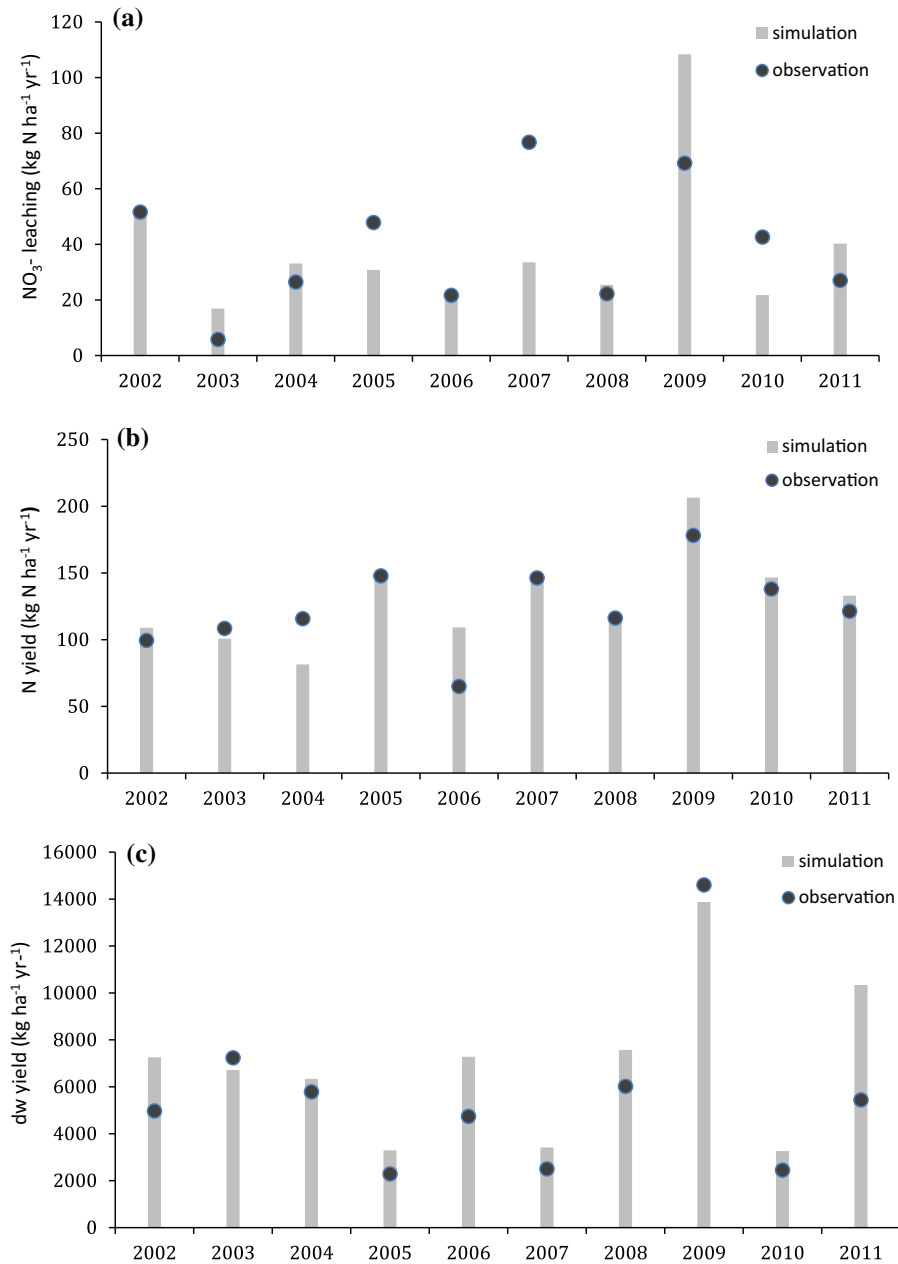
Fig. 3 **a** NO_3^- leaching (kg N ha^{-1} year^{-1}), **b** N yield (kg N ha^{-1} year^{-1}) and **c** dry weight (dw) yield (kg ha^{-1} year^{-1}) comparing observed values to simulated data calculated with LandscapeDNDC

Grieskirchen

At GK 2, the soil type with lowest field capacity, the lowest N_2O emissions (0.28 to 0.37 kg N_2O–N ha^{-1} year^{-1}; mean over all years and CR: 0.32 kg N_2O–N ha^{-1} year^{-1}) were simulated (Fig. 5a, b).

GK 1 with higher silt and C_{org} contents emitted significantly more N_2O ($p < 0.05$) (0.41–0.52; mean 0.47 kg N_2O–N ha^{-1} year^{-1}). In more detail, GK 2 had lowest average emissions in 2006 with values ranging from 0.22 to 0.45 kg N_2O–N ha^{-1} year^{-1}. GK 1 had lowest N_2O losses in 2010 with 0.32 kg

Table 4 Minimum, maximum and mean annual N_2O emissions (kg N_2O–N ha^{-1} $year^{-1}$) and NO_3^- leaching losses (kg NO_3–N ha^{-1} $year^{-1}$) for Marchfeld (MF) and Grieskirchen (GK)

| | N_2O emissions (kg N_2O N ha^{-1} $year^{-1}$) | | | | | | NO_3^- leaching losses (kg NO_3–N ha^{-1} $year^{-1}$) | | | | | |
| | Marchfeld | | | Grieskirchen | | | Marchfeld | | | Grieskirchen | | |
	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean
2006	0.2	1.6	0.5	0.2	0.8	0.3	3.3	67.6	25.0	5.6	71.9	34.7
2007	0.2	1.4	0.6	0.3	0.9	0.4	2.0	107.7	36.3	17.6	149.3	62.6
2008	0.2	1.8	0.6	0.2	0.8	0.4	3.8	76.2	31.1	11.3	64.7	33.3
2009	0.3	0.8	0.5	0.3	0.7	0.4	7.5	81.9	36.1	18.7	96.2	51.3
2010	0.2	1.4	0.5	0.3	0.8	0.3	5.5	88.2	42.8	14.1	80.4	41.1
2011	0.3	0.8	0.5	0.3	0.8	0.4	0.5	84.9	25.2	6.9	90.1	42.2

The illustrated yearly emissions are based on 35 simulation results (5 soil types cross 7 crop rotations) each

N_2O–N ha^{-1} $year^{-1}$ and highest emissions in 2007 with 0.90 kg N_2O–N ha^{-1} $year^{-1}$.

Examining the different crop types reveals that for MF, highest daily N_2O losses were simulated when CORN and BARL were grown (Fig. 6; average of all 5 soils and 7 CRs: 0.0029 and 0.0027 kg N_2O–N ha^{-1} day^{-1}, respectively), followed by BEET and NONE vegetated soils. Low emissions were released when MUST was grown. However, the crop with the lowest N_2O losses was ONIO and WIWH which emitted on average only 0.0011 kg N_2O–N ha^{-1} day^{-1} (data not shown).

In GK at each soil type, four crop types showed always significantly higher N_2O emissions. Similar to MF highest emissions were calculated for the cultivation period of CORN (average 0.0014 kg N_2O–N ha^{-1} day^{-1}), second was LEHA followed by SICO and NONE vegetated soil (Fig. 6).

Lowest N_2O was emitted when RAPE and WBAR were grown. Mean daily N_2O emissions of MF and GK separated by soil type are correlating significantly with clay and C_{org}.

NO_3^- leaching

Significantly (SNK, $p < 0.05$) higher nitrate leaching losses were modelled for GK compared to MF (Table 4). Average annual NO_3^- losses (2006–2011) were 32.8 and 44.2 kg NO_3–N ha^{-1} $year^{-1}$ for MF and GK, respectively. Especially in 2007 ($p < 0.05$) (GK) and 2010 (MF) higher NO_3^- leaching was modelled. All sites in both regions showed lower leaching losses in 2006 and 2008 (only GK) and 2011

(only MF). Average NO_3^- losses separated by soil types ranged from 25 to 62.6 kg NO_3–N ha^{-1} $year^{-1}$.

Marchfeld

Soil type MF 4 (zL) with high C_{org} showed the lowest NO_3^- leaching loss (Fig. 4b). Values ranged from 1.30 to 71.32 kg NO_3–N ha^{-1} $year^{-1}$. Soil type MF 2 (zCL) was less effective in retaining NO_3^- (5.35 to 107.68 kg NO_3–N ha^{-1} $year^{-1}$).

MF 2 and MF 5 showed highest NO_3^- leaching losses during the examined years. Overall, all five soils showed high losses in 2010.

Regarding the crop types during NONE vegetated soils significantly higher NO_3^- losses occurred (on average 0.14 kg NO_3–N ha^{-1} day^{-1}) at soil types MF1–4, followed by BARL and CORN. However, at MF5 these two crops were identified by maximal leaching losses (0.23 kg NO_3–N ha^{-1} day^{-1}). At MF, the cultivation of BEET and WIWH had the best capacity to retain NO_3^- (0.025 and 0.045 kg NO_3–N ha^{-1} day^{-1}, respectively).

Grieskirchen

At GK, significantly (SNK) more NO_3^- was lost compared to MF. The different soil types influenced NO_3^- leaching losses significantly (Fig. 5b). Especially GK 1 and GK 2 are at risk of high NO_3^- losses (on average: GK 1 with 51.08 and 53.63 kg NO_3–N ha^{-1} $year^{-1}$ for GK1 and GK 2 respectively). Soil types GK 4 and GK 5 showed significantly (SNK) lower losses of NO_3^- (on average: 38.32 and 35.08 kg
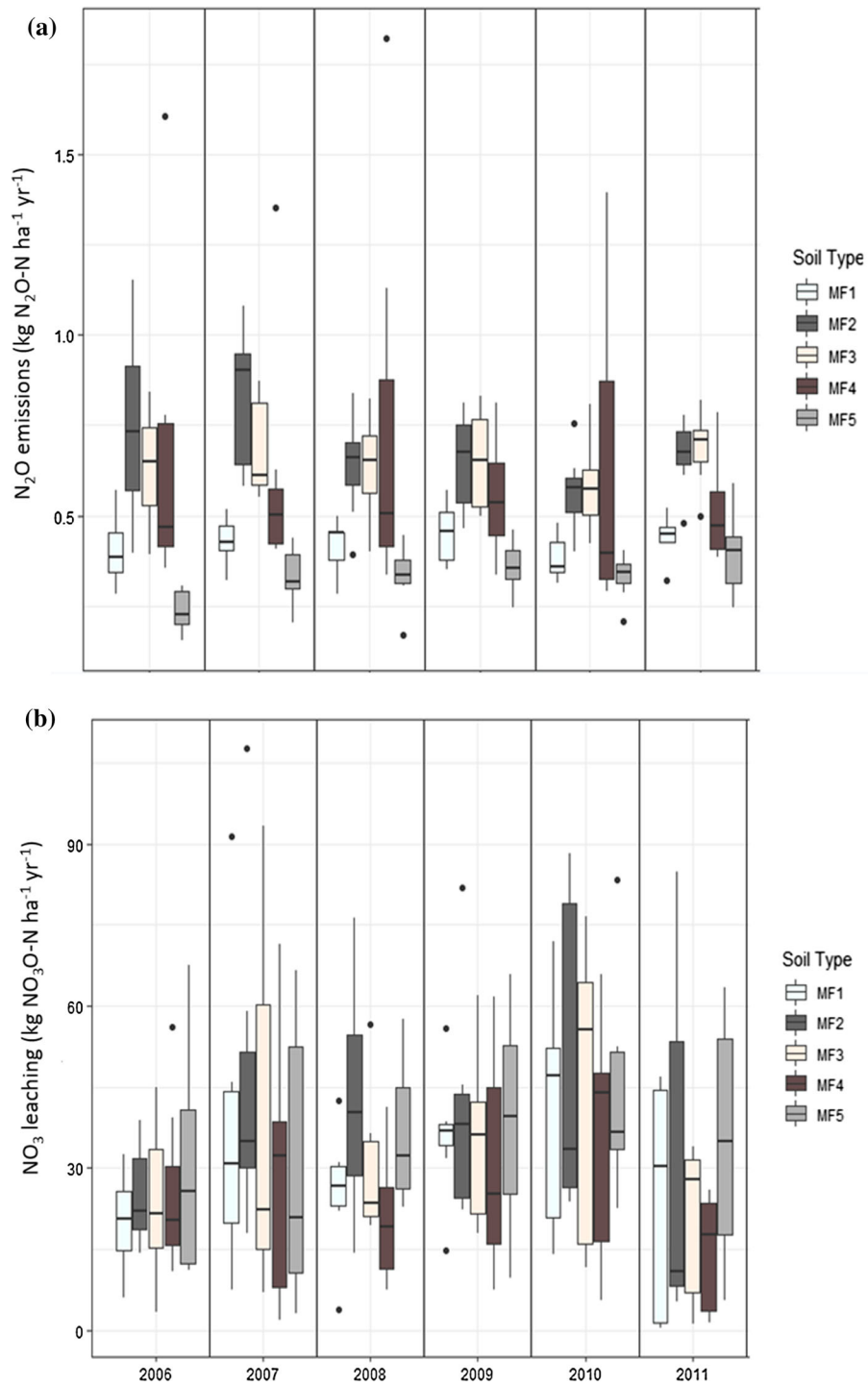
Fig. 4 **a** Annual N$_2$O emissions (kg N$_2$O–N ha^{-1} year^{-1}) and **b** annual NO$_3^-$ leaching (kg NO$_3$–N ha^{-1} year^{-1}) differentiating the 5 soils at MF (annual emissions across the 7 rotations are shown). Dots are marking the highest/lowest emission values for each soil type, bars are the standard deviation
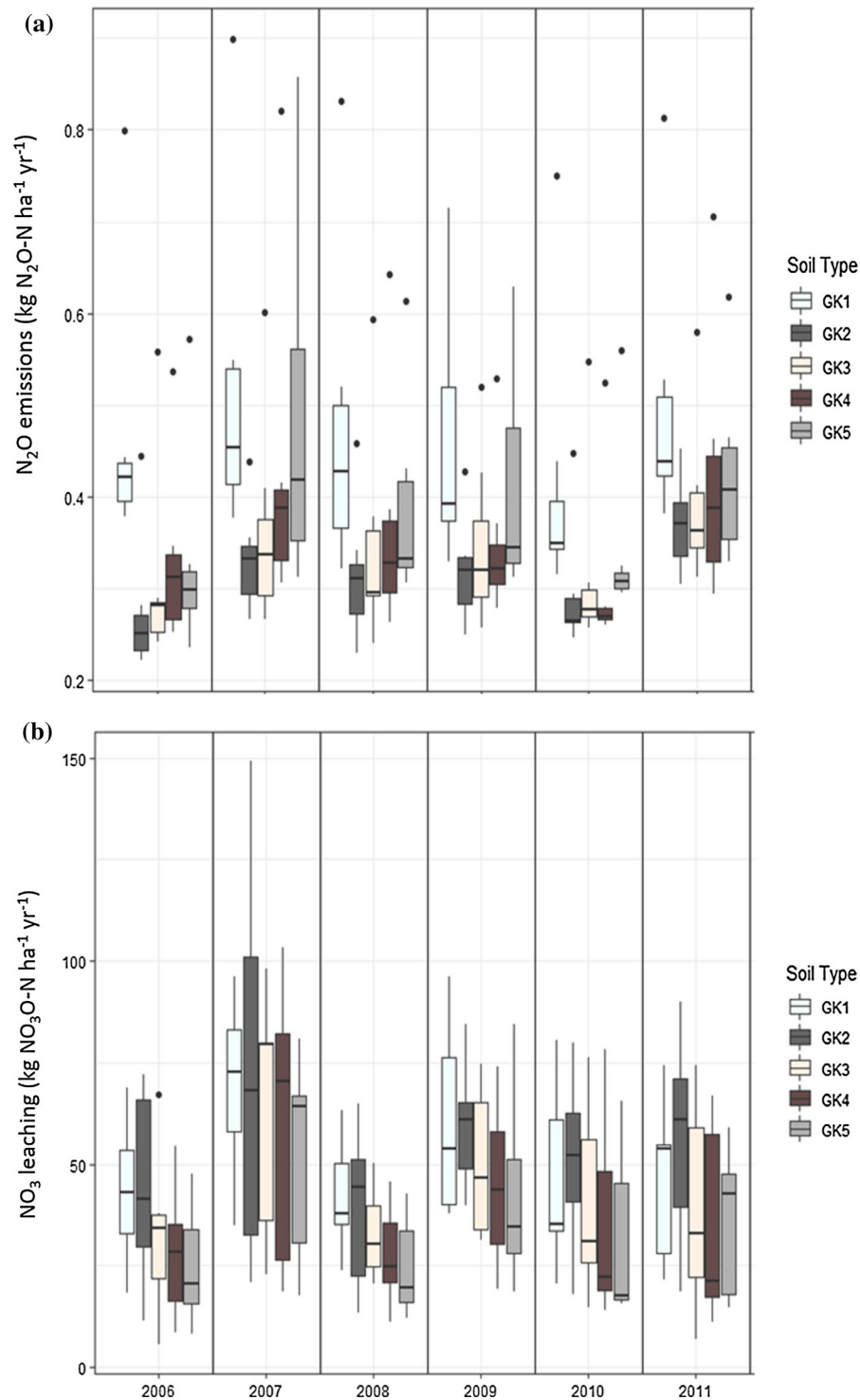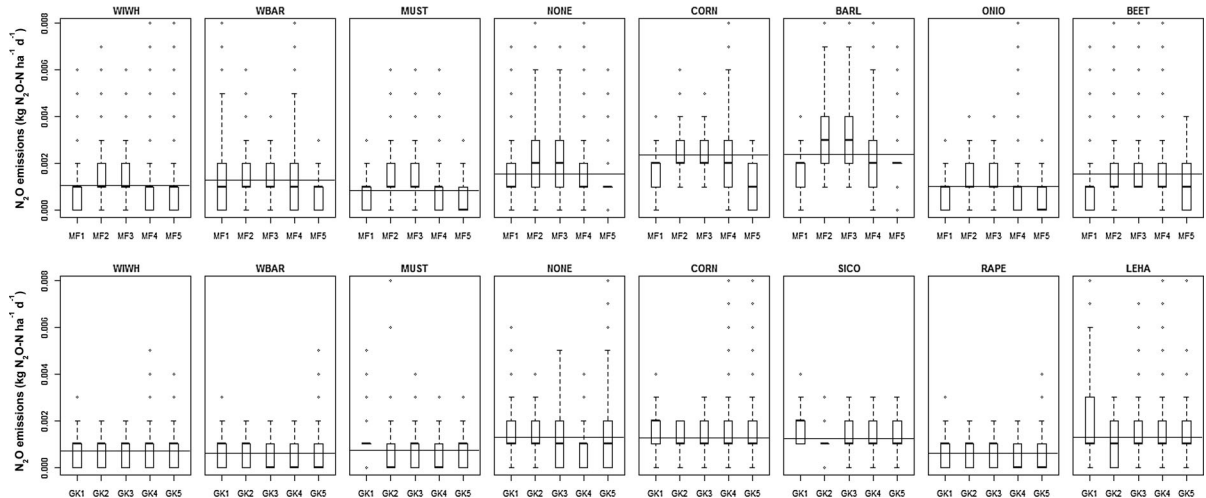
Fig. 5 **a** Annual N$_2$O emissions (kg N$_2$O–N ha^{-1} year^{-1}) and **b** annual NO$_3^-$ leaching (kg NO$_3$–N ha^{-1} year^{-1}) differentiating the 5 soils at GK (all annual emissions across the 7 rotations). Dots are marking the highest emission values for each soil type, bars are the standard deviation

Fig. 6 Mean and median kg N_2O–N ha^{-1} day^{-1} emissions in MF (upper row) and GK (lower row) of each crop in all CRs from 2006 to 2011. MF 1–5 soil types investigated in Marchfeld; GK 1–5 soil types investigated in Grieskirchen. Crops: WIWH (winter wheat), WBAR (winter barley), MUST (mustard), NONE (no crop cultivated), CORN (corn) BARL (barley), ONIO (onion), BEET (sugar beet), SICO (silage corn), RAPE (rapeseed), LEHA (legume-hay)

NO_3–N ha^{-1} $year^{-1}$ for GK 4 and GK 5, respectively compared to GK1 and GK2).

The crop type had a strong impact on NO_3^- leaching losses. NONE vegetated soils released highest amounts of NO_3^- (0.18 kg NO_3–N ha^{-1} day^{-1}). Furthermore, SICO and WBAR (0.174 and 0.168 kg NO_3–N ha^{-1} day^{-1}) lost significant amounts of NO_3^-. Least losses of NO_3^- were simulated when CORN and LEHA (0.061 and 0.073 kg NO_3–N ha^{-1} day^{-1}) were grown.

Evaluating the data (N_2O versus soil temperature in 0.1 m depth), r^2 reaches up to 0.8 in MF and 0.9 in GK. Overall, the soil temperature has a greater influence on N_2O emissions in GK compared to MF. However, soil temperature is an influencing factor for N_2O emissions especially for GK 1, 2 and 3.

Regarding NO_3^- losses (NO_3^- losses versus infiltration rate) r^2 reaches 0.6 in MF and 0.5 in GK. The rate of infiltration has the highest impact on MF 1 and GK1.

Discussion

Comparing the two regions, N_2O emissions at GK were significantly lower than fluxes at MF. The emission strength of N_2O depends on the following factors (Butterbach-Bahl et al. 2013): temperature, moisture, management, crop type and on the local soil conditions, e.g. the organic carbon and nitrogen content. At MF, significantly less N_2O was lost from soils with higher sand fractions and low C_{org}. The poor soil MF 5 (sL) with C_{org} of 0.58%, clay (14%), silt (27%), and pH (5.8) values and highest sand fraction (59%) produced lowest emissions between 2006 and 2011 (from 0.087 to 0.591 kg N_2O–N ha^{-1} $year^{-1}$). On the contrary, highest emissions were calculated for soils MF 2, 3 and 4 (e.g. MF 2, zCl) which are characterized by higher clay and silt contents, high C_{org} and high field capacity. These findings are in accordance with the theory that heavy soils are more likely to develop anaerobic microsites conditions for denitrification (Butterbach-Bahl et al. 2004). We could find a higher increase in soil emissions with increasing soil temperature at MF 2 and MF 3. On the other hand, MF 5 (sandy soil, low C_{org}) did not react that strongly to temperature increases, which could relate to the fact that sandy soils tend to have a lower microbial biomass (Meyer et al. 1997).

Despite interannual differences CORN caused at MF highest emissions. This crop received the highest mineral fertilizer (150 kg N ha^{-1} $year^{-1}$) amount in one basal application, 10 days prior to the seeding. An N_2O peak occurs at the beginning of the cultivation. As stated in Fuß et al. (2011) a high amount of applied fertilizer is the dominant factor releasing N_2O. Therefore a combination of applied fertilizer and low N uptake due to initial plant growth probably

caused this peak. Furthermore, increasing temperatures also raise emissions. Especially MF 2 (zCl) has the highest potential, during CORN and BARL to release N_2O. ONIO crops and WIWH on the other hand released on average lowest N_2O emissions.

At GK, lowest annual N_2O emissions were simulated for GK 2 (zL) which has lowest field capacity (262 mm m^{-3}). GK 1 (zL), on the contrary, containing high silt (71%) and C_{org} (3.25%) content as well as a very low pH (4.5) showed highest values no matter which crop was cultivated. GK 5 showed similar results. In 2007 and 2009, precipitation was high and higher temperatures were recorded. In 2010, all soil types in GK show lowest, but in 2007, highest N_2O emissions. The overall low emissions in 2010 can be due to the very low average temperature of 10.7 °C, whereas the high emissions might be accounted for to the exceptional high average temperature of 12.6 °C in 2007 (Flessa et al. 2002). Also at GK, high N_2O emissions were caused by the cultivation of CORN, followed by LEHA, NONE and SICO whereas winter crops like RAPE, WBAR and WIWH released lowest N losses in form of N_2O.

Regarding the NO_3^- loss to the groundwater, significantly higher losses were found at GK compared to MF. Especially in 2007, (high precipitation and high temperature) high losses were simulated in GK, whereas lower losses were found in 2006 and 2008. At MF, high losses occurred in 2010 (highest precipitation) and low losses were simulated in 2006 and 2011 (least precipitation). Besides the climatic impact in both regions, nitrate leaching is an interacting process of water movement and nitrate availability during critical times of the year.

Losses of NO_3^- were lowest at the silty loam (MF 4), but highest at silty clay loam (MF2) and sandy loam (MF 5). In general high losses at all soils occurred in 2010 and 2007, which could be due to the higher precipitation in both years. This is in accordance with the study of Kim et al. (2014) where high rainfall events were referred to as the source of highest NO_3^- leaching rates. During times of no crop cultivation (NONE), highest losses were simulated. Crops with high losses of NO_3^- were BARL followed by CORN. Lowest leaching losses were found during the cultivation of BEET and WIWH.

Silty loams (GK 1, 2) are at risk of NO_3^- losses whereas silty clay loams (GK 4, 5) have the potential to better retain NO_3^- from leaching. However, high

losses occurred in all soils in the year 2007 which was the hottest and wettest year. Silty loams can therefore be classified as a hot spot of NO_3^- leaching since high losses occur independent of the crop type. In this area soils with a high content of silt are dominating. The soils are deep (> 2 m) and the share of clay increases in the subsoil. However, a more prevailing hot spot is GK 1, since it not only shows high NO_3^- losses but releases also more N_2O. Overall NONE vegetated soils led to high losses followed by WBAR and SICO. As in MF, WIWH tended to retain NO_3^-. GK 1 differs from the other sites not only regarding the high silt and C_{org} contents, it also has a significant lower pH value compared to the other computed sites. Though the direct impact of pH on denitrification has been hard to established (Šlmek and Cooper 2002) whereas the content of C_{org} and texture do effect N_2O emissions (Butterbach-Bahl et al. 2004).

With regard to the climate, the year 2010 (MF) and 2009 (GK) were the wettest in the investigated period. The year 2010 was in both regions the coldest. As stated in Heincke and Kaupenjohann (1999), it has to be considered that seasonal fluctuations may exert significant influence on biological, physical and chemical factors which determine the nitrogen dynamics. Since nitrification and denitrification are influenced by soil moisture, increases in precipitation can lead to a rise in nitrate leaching, but do not necessarily induce elevated N_2O emissions (Haas et al. 2013). An increase in temperature may increase soil respiration which leads to an increase in anaerobic volume in which denitrification takes place causing an increase in N_2O emissions (Flessa et al. 2002). Clearly in 2007, emissions rose, whereas in 2010, emissions dropped. Since calculations of all soil types across all CRs are averaged, the main reason for the differences lies in the local weather conditions. Increased precipitation and rising temperature (as in 2007) may have triggered higher direct emissions. However, also less precipitation and high mean annual temperatures may induced an increase in emissions (as in 2011). Low mean annual temperatures in 2010 (about 1 °C lower than the average of 2006–2011) resulted in lowest direct N_2O. Compared to the long term climate observations, both regions received more precipitation between 2006 and 2010. Only 2011 lay in the long term average trend. Interestingly, in 2009 GK had substantial more precipitation compared to MF. On the contrary in 2010, MF had more than a third of precipitation in

2010 (> 250 mm) whereas GK received not significantly more (long term mean annual precipitation lies at 890 mm).

Furthermore, management operations contribute to N utilization and losses. Long term ground coverage provides not only a measure against soil erosion, but cultivating cash and catch crops over the vegetation period offers high N utilisation (Küstermann et al. 2010). Changes in the soil structure due to different management operations can also exert a strong influence on the N_2O gas fluxes (Flessa et al. 2002).

The N_2O and NO_3^- results display high interannual variation due to certain combinations of weather, soil properties and management operations resulting in pulses of N-loss. Especially unfavourable weather conditions for plant growth prohibit a full N utilization. Large amounts of the applied nitrogen cannot be used by the crops whereas large amounts of N which cannot be stored in the soil organic nitrogen stock are subject to gaseous or leaching losses (Küstermann et al. 2010). In our study, the largest part of nitrogen was lost via NO_3^- leaching whereas only approximately 1% of the investigated N loss was N_2O. Other studies reveal similar results (Kim et al. 2014) and point out that such NO_3^- losses are potential indirect sources of N_2O emissions. However, it has to be considered that some of the applied fertilizer can get leached to deeper soil layers or volatilize as NH_3 after application rather than causing N_2O emissions (Haas et al. 2013).

Also, the composition of microbial communities exerts an influence on the nitrogen flux e.g. number of NH_3 oxidisers present in the soil, and are strongly affected by the type of N fertilizer applied (Inselsbacher et al. 2010). Besides N mineralization, consumption and nitrification rates also the mean residence time of soil NH_4^+ and NO_3^- pools are influenced by soil activity/N transformation rates (Inselsbacher et al. 2010). In the LandscapeDNDC model these parameters are calculated reflecting the effect of temperature and moisture in combination with crop type.

As stated in Küstermann et al. (2010) a close relationship exists between N input, N surplus, N_2O emissions and N leaching since the N surplus in the soil surface reacts to the different management operations and consequently influences the N cycle mainly due to N emissions and soil organic N content. Nevertheless, N is transported from one place to another, which means a loss in one way and a gain in another (Galloway 1998). Therefore besides the advantage of large yield increases by applying nitrogen fertilizers, negative effects such as N_2O emissions and NO_3^- leaching occur which present an environmental hazard. Even though this study has shown that N losses for two sites in Austria are low, it remains important to maintain a high NUE and keep the losses as low as possible.

In our hypotheses 1 we postulated that sites in regions with less precipitation (MF) release less N_2O emissions than sites in regions with higher precipitation (GK). This hypothesis was rejected. In our study the difference in precipitation in the chosen regions influenced the release of N_2O to a lesser extent than the properties of the soils.

Hypotheses 2 said that release of N_2O emissions are influenced by crop rotations and management. Here we could show that crop rotations had a significant influence on N_2O emissions with corn releasing highest N_2O emissions and winter cereals (e.g. WIWH in MF and WBAR in GK) mitigated emissions.

In Hypotheses 3 we postulated that higher nitrate leaching occurs in lighter soils in years of high precipitation. This hypothesis was supported by the model results. In addition our study revealed negative relations between nitrate leaching and N_2O emissions, as nitrate leaching was enhanced in lighter soils and N_2O emissions in heavy soils with high C_{org} and high field capacity.

The new and innovative approach of this study was the regional approach taking into consideration combinations of influencing factors and their additive effects on N_2O-emissions as well as nitrate leaching, which at some points lead to unexpected results.

Conclusion

The results are considered reliable since a consistency between modelled estimations was confirmed in the past. This approach is able to take local and regional climate, soil and management conditions into account which are the driving forces that induce the relevant biogeochemical processes generating GHG emissions.

Applying LandscapeDNDC allowed identifying soil types with low sand content as hot spots of N_2O emissions and sandy soils as hot spots for NO_3^- leaching. Furthermore, this tool revealed the impact of

cultivation and management on N_2O emissions and NO_3^- leaching for the most common crop types in the selected regions. In nitrate intensive farming systems it is difficult to spatially and temporally optimize N fertilization. However, changes in farm management are the key to reduce NO_3^- leaching losses to the environment via demand tailored split application and reduced fertilization rates. The model results provide a basis for optimization measures, e.g. avoiding times of no crop cultivated or considering different crop and soil characteristics, which should help to improve the NUE and therefore the productivity while decreasing the environmental impacts.

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