Nitrous oxide emissions from winter oilseed rape cultivation

Reiner Rusera, Roland Fußb, Monique Andréc, Hannes Hegewaldd, Katharina Kesemierera, Sarah Köbkef, Thomas Räbigerf, Teresa Suarez Quinonesg, Jürgen Augustinc, Olaf Christend, Heinz Stichnothej, Heinz Flessab

a Institute of Crop Science, Department Fertilization and Soil Matter Dynamics, University Hohenheim (340i), Fruwirthstraße 20, D-70599, Stuttgart, Germany
b Thünen Institute of Agricultural Technology, Bundesallee 50, D-38116, Braunschweig, Germany
c Institute of Crop Science, Department Biobased Products and Energy Crops, University Hohenheim (340b), Fruwirthstraße 23, D-70599, Stuttgart, Germany
d Institute of Agricultural and Nutritional Sciences, Martin-Luther-University Halle-Wittenberg, Betty-Heimann-Straße 5, D-06120, Halle, Germany
e Institute of Applied Plant Nutrition, Georg-August-Universität Göttingen, Carl-Sprengel-Weg 1, D-37075, Göttingen, Germany
f Institute of Landscape Biogeochemistry, Leibniz Centre for Agricultural Landscape Research (ZALF), Eberswalder Strasse 84, D-15374, Müncheberg, Germany
g Institute of Landscape Biogeochemistry, Leibniz Centre for Agricultural Landscape Research (ZALF), Eberswalder Strasse 84, D-15374, Müncheberg, Germany
h Institute of Crop Science, Department Fertilization and Soil Matter Dynamics, University Hohenheim (340i), Fruwirthstraße 20, D-70599, Stuttgart, Germany
i Thünen Institute of Agricultural Technology, Bundesallee 50, D-38116, Braunschweig, Germany

ABSTRACT

Winter oilseed rape (Brassica napus L., WOSR) is the major oil crop cultivated in Europe. Rapeseed oil is predominantly used for production of biodiesel. The framework of the European Renewable Energy Directive requires that use of biofuels achieves GHG savings of at least 50% compared to use of fossil fuel starting in 2018. However, N2O field emissions are estimated using emission factors that are not specific for the crop and associated with strong uncertainty. N2O field emissions are controlled by N fertilization and dominate the GHG balance of WOSR cropping due to the high global warming potential of N2O. Thus, field experiments were conducted to increase the data basis and subsequently derive a new WOSR-specific emission factor.

N2O emissions and crop yields were monitored for three years over a range of N fertilization intensities at five study sites representative of German WOSR production. N2O fluxes exhibited the typical high spatial and temporal variability in dependence on soil texture, weather and nitrogen availability. The annual N2O emissions ranged between 0.24 kg and 5.48 kg N2O-N ha−1 a−1. N fertilization increased N2O emissions, particularly with the highest N treatment (240 kg N ha−1). Oil yield increased up to a fertilizer amount of 120 kg N ha−1. N2O emissions were not significantly correlated with oil or protein concentration in the seeds. Consequently oil yield remained constant at higher N fertilization. Since, yield-related emission also increased exponentially with N surpluses, there is potential for reduction of the N fertilizer rate, which offers perspectives for the mitigation of GHG emissions. Our measurements double the published data basis of annual N2O flux measurements in WOSR. Based on this extended dataset we modeled the relationship between N2O emissions and fertilizer N input using an exponential model. The corresponding new N2O emission factor was 0.6% of applied fertilizer N for a common N fertilizer amount under best management practice in WOSR production (200 kg N ha−1 a−1). This factor is substantially lower than the linear IPCC Tier 1 factor (EF1) of 1.0% and other models that have been proposed.

1. Introduction

In the context of biofuel production especially nitrous oxide (N2O) contributes to high GHG emissions during the step of biomass production (Dufossé et al., 2013; Hong, 2012). N2O is a climate relevant trace gas that absorbs light in the IR spectrum and therefore reduces the atmospheric transparency to thermal radiation from the earth’s surface (Granli and Bøckman, 1994). The atmospheric N2O concentration in the last decade increased by 0.73 ppb a−1 and with a mean concentration of 328 ppb in 2015 it exceeded the pre-industrial level by about 21%...
Nitrous oxide ($\text{N}_2\text{O}$) contributes 7.4% (0.17 W m$^{-2}$) of the total anthropogenic radiative forcing (IPCC, 2013); it has a high heat absorption capacity, a long atmospheric lifetime of more than 100 years and has a 296 fold higher global warming potential (IPCC, 2001; RED, 2009) compared to the same mass of carbon dioxide (CO$_2$). Besides its contribution to the greenhouse effect, $\text{N}_2\text{O}$ also contributes to stratospheric ozone depletion (Crutzen, 1981; Ravishankara et al., 2009). Approximately 60% of anthropogenic $\text{N}_2\text{O}$ emissions are released by agricultural soils (Clais et al., 2013). There is general agreement that nitrification and biological denitrification are the main sources for $\text{N}_2\text{O}$ production in soils (Bremner, 1997), whereas the contribution of other processes such as nitrifier-denitrification is currently under discussion (Wragle et al., 2001; Shaw et al., 2006; Butterbach-Bahl et al., 2013).

All processes of $\text{N}_2\text{O}$ production in soils rely on mineral N (i.e. Ruser et al., 2001; Zebarth et al., 2008). Therefore, $\text{N}_2\text{O}$ emissions from agricultural soils generally increase with increasing N fertilization as it provides the substrates ($\text{NO}_3^-$, $\text{NH}_4^+$) for $\text{N}_2\text{O}$ production (i.e. Stehfest and Bouwman, 2006). Furthermore, $\text{N}_2\text{O}$ emission is correlated with N surpluses (N fertilization – N removal) in arable systems (Kaiser and Ruser, 2001; Van Groenigen et al., 2004) as well as in horticultural systems (Plab et al., 2011).

Oilseed rape ($\text{Brassica napus}$ L.) is the major oil crop in Europe, accounting for more than 70% of the European oilseed volume in 2012 (Carré and Pouzet, 2014). In 2014, oilseed rape covered 9.1 × 10$^6$ ha or approximately 8.5% of the total European arable land (FAO, 2016). The corresponding mean grain yield was 3.17 Mg ha$^{-1}$. In the same year, the mean grain yield in Germany was 4.48 Mg ha$^{-1}$ on 1.4 × 10$^6$ ha (German Federal Statistical Office, 2017), showing both the high potential for winter oilseed rape (WORS) cultivation as well as the reason for Germany’s leading position (together with France) regarding WORS production in the EU.

The acreage of WORS in the European Union more than doubled between 2003 and 2014 (FAO, 2016), which went along with the increase of biodiesel contributing more than 75% of the transport biofuels in Europe (Hamelink et al., 2013). This increased production is also a result of the Renewable Energy Directive (RED, 2009), in which the European Union mandates a share of 10% from renewables in the transport energy sector by 2020. The RED also defined sustainability criteria for biofuels, which were updated in 2015 (EU, 2015). According to these criteria, biofuels can only be considered and consequently subsidized as such if they contribute to a total reduction of greenhouse gas emissions (GHG) of 35% (current reduction value) and, starting from 2018, of 50% (for production plants that became operational before October 2015) and by 60% (for new production plants) in comparison to the use of fossil fuel.

WORS is a crop demanding high amounts of N fertilizer to build up efficient photosynthetic leaf tissue (Hegewald et al., 2016). Maximum dry matter yield is achieved with N rates exceeding 200 kg N ha$^{-1}$ whereas N removal with the seeds as well as the N harvest index are low, thereby resulting in high N surpluses of up to 90 kg N ha$^{-1}$ a$^{-1}$ (Henke et al., 2007; Sieling and Kage, 2010). It has also been reported that large amounts of crop residues (petals and leaves), which can be mineralized easily, are returned to the soil after flowering (Sieling and Kage, 2010). Furthermore, N uptake by WORS plants ends early and increases in N content in seeds during pod filling is more the result of N translocation from vegetative plant parts than from N uptake from soil (Malagoli et al., 2005); both will result in enhanced soil mineral N contents during or shortly after the harvest period. Winter wheat ($\text{Triticum aestivum}$ L.) is the predominant succeeding crop for WORS in crop rotations. The N uptake of winter wheat before winter is approximately 20 kg N ha$^{-1}$ and as such markedly below the N release after WORS cultivation (Sieling and Kage, 2010). Both, the N surpluses as well as the high soil nitrate contents have the potential of fueling $\text{N}_2\text{O}$ production in soils.

Due to the high global warming potential of $\text{N}_2\text{O}$, the assessment of $\text{N}_2\text{O}$ emissions with a reliable emission factor is of vital importance for the calculation of GHG balances of biofuels, such as biodiesel produced from WORS. Results from life cycle analysis (LCA) suggest that direct and indirect $\text{N}_2\text{O}$ emissions account for between 20 and 40% of the total GHG emission associated with the production and consumption of biodiesel (Hong, 2012; Dufossé et al., 2013). For a bioethanol production system, the choice of different available $\text{N}_2\text{O}$ emission factors in LCAs might result in completely contrasting results and conclusions, as Smith and Searchinger (2012) remarkably demonstrated. Following IPCC guidance, they set the emission factor to 1.5% (including direct and indirect emissions) and the corresponding emission reached the 35% GHG reduction goal. Using the distinct higher emission factor of 4%, as suggested by Crutzen et al. (2008), based on their so-called “top-down” approach, the reduction potential for wheat-based bioethanol was completely eliminated.

In order to assess fertilizer-induced $\text{N}_2\text{O}$ emissions, different $\text{N}_2\text{O}$ emission factors have been proposed. The IPCC (2006) guidelines suggest a constant direct $\text{N}_2\text{O}-\text{N}$ loss of 1% of N applied and N in crop residues. This default emission factor was modified from a global data set for wheat and grassland sites originally provided by Bouwman (1996) and, as mentioned by Bouwman, does not consider crop type or site-specific effects. A further drawback of this emission factor is that $\text{N}_2\text{O}$ emissions do not necessarily correlate linearly with N fertilizer amounts and that $\text{N}_2\text{O}$ emissions increase over-proportionally when high N fertilizer doses exceed plant demand (McSwiney and Robertson, 2005; Hoben et al., 2011; Kim et al., 2013).

The Joint Research Centre (JRC) of the EU provides an online tool (the so-called Global Nitrous Oxide Calculator, GNOC) to assess GHG emissions from biofuels in EU legislation (Edwards et al., 2013). This tool calculates $\text{N}_2\text{O}$ emissions based on the approach of Stehfest and Bouwman (2006). It uses an exponential algorithm that considers site and management specific characteristics such as soil texture, climate, soil organic matter, pH and vegetation. In this model, WORS was originally in the vegetation class “other” but the JRC recently moved it into the same class as “cereals” without refitting the model (Edwards et al., 2016). This resulted in a calculative reduction of the $\text{N}_2\text{O}$ emissions from WORS.

The decision to move WORS to the cereals group in the GNOC tool is supported by Walter et al. (2015) who used data sets on $\text{N}_2\text{O}$ emissions from WORS fields to run a meta-analysis. They also used an exponential model for fertilizer-derived $\text{N}_2\text{O}$ emission from WORS, which resulted in even lower $\text{N}_2\text{O}$ emissions than the GNOC tool.

In regions with strong frost-thaw cycles, high $\text{N}_2\text{O}$ fluxes can occur during thawing periods (Flessa et al., 1995; Röver et al., 1998). These high thaw pulses can account for more than 50% of the annual $\text{N}_2\text{O}$ budget from agricultural soils (Kaiser and Ruser, 2001; Jungkunst et al., 2006). Due to these high $\text{N}_2\text{O}$ winter fluxes, annual measurements are a prerequisite for the reliable quantification of $\text{N}_2\text{O}$ emissions. Consequently, the duration of the period of trace gas measurements was a criterion for the inclusion (measurements covering > 300 days) or exclusion of data sets in the review by Walter et al. (2015), and only 12 studies with 18 annual datasets (43 data points in total) fulfilled this criterion. Additionally, the small dataset showed a high variability of the $\text{N}_2\text{O}$ emissions among study sites and also among experimental years.

The main aims of our investigations were therefore:
(i) to determine direct annual $\text{N}_2\text{O}$ emission from WORS fields over a broad range of production sites, representing areas with a high proportion of WORS within the crop rotations, thereby extending the currently available data substantially,
(ii) to quantify the effect of N fertilization on $\text{N}_2\text{O}$ fluxes and on yield-related $\text{N}_2\text{O}$ emission, and
(iii) to deduce a fertilizer-related emission factor (FRE) specific for the production of winter WORS.

2. Materials and methods

2.1. Study sites, experimental design and management

Trace gas measurements were conducted at five study sites located
in representative areas with a high share of WORS in the crop rotation. Three sites were located in (1) northern Germany representing approximately 55% of the total German WOSR production area (Hohenschulen, University Kiel; Dedelow, ZALF Müncheberg; Berge, ATB Potsdam-Bornim/Humboldt University Berlin), (2) one site in central Germany (Merbitz, University Halle-Wittenberg) representing 30%, and (3) one site in southern Germany (Hinger Hof, University Hohenheim) representing 15% of the total German oilseed area. The main characteristics of the study sites are shown in Table 1.

At each site, a randomized split-plot experiment with four replicated blocks was established in 2012. The crop rotation was identical at all sites. All crops of the rotation, winter oilseed rape (var. ‘Visby’), winter wheat and winter barley were managed according to best agricultural practices without any further differentiation within the crop. Plot size varied slightly over the study sites due to different farming machinery; the minimum size was 3 × 9 m (27 m²).

WORS was sown at all sites between end of August and the first two weeks in September (40–45 grains m⁻², inter-row width was 0.36 m). In early spring, 90 kg S ha⁻¹ were applied as kieserite (MgSO₄·H₂O) to avoid S deficiency in all WORS treatments including the unfertilized control. After harvest, in the period between mid-July and early August, the soil was ploughed to a depth of 25 cm and winter wheat was subsequently sown at the end of September or in early October. Crop protection and further management measures were conducted according to site-specific agricultural practice. At Berge WORS straw was removed after harvest whereas it remained on the field at the other study sites. This removed about 20 kg N ha⁻¹ a⁻¹ (2.8 Mg C ha⁻¹ a⁻¹, C/N = 105) from the site Berge (median of all years and treatments).

The treatments of WORS relevant for results described hereafter, were an unfertilized control and treatments fertilized with 60, 120, 180, or 240 kg N ha⁻¹ a⁻¹ for yield determination. Typical WORS fertilization targets in Germany are in the range from 180 kg ha⁻¹ a⁻¹ to 210 kg ha⁻¹ a⁻¹. The 180 kg N ha⁻¹ a⁻¹ fertilization treatment represents a typical fertilization target value of 200 kg N subtracting Nₘₐᵢₙ contents after winter (approximately 20 kg N ha⁻¹). At all sites we measured N₂O fluxes also in additional treatments such as biogas residue application or soil tillage variants. However, these will be discussed in subsequent publications. Trace gas fluxes were measured in every year and at every site in the 120 kg N ha⁻¹ and in the 180 kg N ha⁻¹ treatment and in some further N treatments (including some of the unfertilized controls) in single years (Table 2). N fertilization to WORS was split into two equal doses with a first application at the beginning of the growing season and the second application in BBCH-stage 5 (in florescence emergence, Meier, 2001) approximately four weeks after the first N application. We used calcium ammonium nitrate (CAN) for all N applications.

### Table 1

<table>
<thead>
<tr>
<th>Study site</th>
<th>Coordinates</th>
<th>MAP</th>
<th>MAT</th>
<th>Soil type</th>
<th>Soil texture</th>
<th>pH</th>
<th>Cₑₙₑ</th>
<th>Nₑₙₑ</th>
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<tbody>
<tr>
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<td></td>
<td>[mm a⁻¹]</td>
<td>[°C]</td>
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<td>Clay [%]</td>
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<td></td>
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<td></td>
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<td>Sand [%]</td>
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<td></td>
<td>0.01</td>
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<td>16.8</td>
<td>6.6</td>
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<tr>
<td></td>
<td>E 8°55’26”</td>
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<td>8.6/10.4/10.1</td>
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<td>3.2</td>
<td>78.2</td>
<td>16.8</td>
<td>6.6</td>
</tr>
<tr>
<td>Hogenschulen</td>
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<td>8.9</td>
<td>Haplic Luvisol/Anthrosol</td>
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<td>60.1</td>
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</tr>
<tr>
<td></td>
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<td>8.1/9.6/8.8</td>
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<td>10.5</td>
<td>29.4</td>
<td>60.1</td>
<td>5.9</td>
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<tr>
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<td>Haplic Chernosem</td>
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<td>67.8</td>
<td>16.4</td>
<td>6.6</td>
</tr>
<tr>
<td></td>
<td>E 11°91’12”</td>
<td>700/456/429</td>
<td>9.1/10.7/10.4</td>
<td></td>
<td>15.8</td>
<td>67.8</td>
<td>16.4</td>
<td>6.6</td>
</tr>
</tbody>
</table>

MAP: Long-term mean annual precipitation and annual precipitation in the single experimental years; MAT: Long-term mean annual air temperature (2m) and annual mean air temperature in the single experimental years.

1 measured in the top soil (0–30 cm).


### 2.2. Flux measurements

Using the closed chamber method (Hutchinson and Mosier, 1981), flux measurements were conducted at least once a week starting in January or February 2013 and ending in December 2015. Chambers were placed between the seed rows of WORS, but included the plants for measurements in cereals. In order to place the chambers between the plant rows, the chambers’ dimensions were 71 cm length, 27 cm width and 10 cm height. Chamber material was white opaque PVC (PS-plastic, Eching, Germany). They were equipped with rubber sealing, a pressure vent and a ventilator. For measurements, the chambers were anchored on their frames using elastic straps. The frame height was approximately four weeks after the first N application. We used calcium ammonium nitrate (CAN) for all N applications.

### Table 2

<table>
<thead>
<tr>
<th>Study site</th>
<th>Exp. year</th>
<th>N fertilization [kg N ha⁻¹ a⁻¹]</th>
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<tr>
<td>Berge</td>
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<tr>
<td></td>
<td>2014</td>
<td>–</td>
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<tr>
<td></td>
<td>2015</td>
<td>–</td>
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<tr>
<td>Dedelow</td>
<td>2013</td>
<td>–</td>
</tr>
<tr>
<td></td>
<td>2014</td>
<td>–</td>
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<tr>
<td></td>
<td>2015</td>
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<tr>
<td></td>
<td>2013–2015</td>
<td>–</td>
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<tr>
<td></td>
<td>2014–2015</td>
<td>–</td>
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<tr>
<td>Hinger Hof</td>
<td>2013</td>
<td>–</td>
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<td></td>
<td>2014</td>
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<td></td>
<td>2013–2015</td>
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<tr>
<td>Hogenschulen</td>
<td>2013</td>
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<tr>
<td></td>
<td>2013–2015</td>
<td>–</td>
</tr>
<tr>
<td>Merbitz</td>
<td>2013</td>
<td>–</td>
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<tr>
<td></td>
<td>2014</td>
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<td>2015</td>
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<tr>
<td></td>
<td>2013–2015</td>
<td>–</td>
</tr>
</tbody>
</table>

- Not determined/not calculated.
- Mean values (only given for treatments with 3 years of measurements).
13 cm and they were installed in soil to a depth of 5 to 10 cm. During flux measurements, the chambers were kept closed for one hour; gas samples were taken every 20 min using vacuumtainers or stopcock vials, resulting in four gas samples per flux measurement. Chamber temperature was recorded for each gas sample. Gas samples were analyzed for N\(_2\)O and CO\(_2\) concentrations in the laboratories of the participating research groups by various gas chromatographs equipped with electron capture and flame ionization detectors as well as automatic samplers. Lab inter-comparability was verified by conducting blind inter-comparison measurements between the labs in the beginning of the study. Each laboratory achieved a coefficient of variance below 2% on ten repeated measurements of an ambient N\(_2\)O standard gas (data not shown).

2.3. Environmental, soil, and plant analyses

A climate station was installed next to the experimental plots at each of the study sites. We measured precipitation and air temperature (2 m and 5 cm height). Additionally, soil temperature in one of the four replicated main plots was recorded in 5, 10, and 20 cm soil depth (Logtac, TRIX-8, CIK solutions, Karlsruhe, Germany).

Simultaneously to each gas sampling, soil samples were taken from 0 to 30 cm depth with an auger. The soil from three insertions per replicate plot was pooled over the four replicates, sieved (< 5 mm) and stored frozen until further analysis. Additionally, in early spring and after harvest of the WOSR N\(_{\text{min}}\) was determined in 0–30, 30–60, and 60–90 cm depth. These soil samplings were carried out for each plot separately.

For the quantification of mineral N contents, 80 g of soil were extracted with 200 ml of a 0.0125 M CaCl\(_2\) solution. Concentrations of NO\(_3\)\(^-\) and NH\(_4\)\(^+\) in the extracts were determined using flow-injection analyzers. The analyzers used for that purpose were tested for comparability in an inter-laboratory test. A further aliquot of the soil was used to determine soil moisture by drying at 105 °C for one day.

Before and after soil management events, bulk density of the top soil was determined using stainless steel cylinders (100 ml).

Fresh matter yield was determined by cutting WOSR plants from 1 m\(^2\). The green cut was separated into straw and pods which were flailed subsequently. Moisture was determined after drying for three days at 60 °C. Aliquots of the milled straw and grains were analyzed for C and N using an elemental analyzer (vario Max CN, Elementar Analysensysteme, Hanau, Germany). The oil content of the rapeseeds was determined with NIRS (NIRSystem 5000, Foss, Hamburg, Germany).

2.4. Calculations and statistical analyses

2.4.1. Flux calculation

Molar gas concentrations were transformed into mass concentrations according to the ideal gas law taking chamber temperature and standard pressure into account. We used several criteria to select the most appropriate flux calculation model and to evaluate the reliability of calculated fluxes. The Akaike information criterion (AIC) was used to decide between flux calculation by the HMR model (Pedersen et al., 2010) and robust linear regression (Huber, 1981). The HMR estimate was used if its AIC value was smaller than the AIC from linear regression and if its kappa value, which controls the curvature, was smaller than 20 h\(^{-1}\). Restricting kappa this way avoids strong overestimation of fluxes due to outliers of the first concentration−time point, which can result in an excellent fit of the nonlinear model but extreme curvatures and flux estimates. A linear regression was applied if only three gas samples were available for flux determination.

The resulting gradients at time zero were multiplied with chamber volume divided by chamber area to derive the flux estimates. For this, the height of the frame was determined after any changes, such as re-installation after tillage measures. Snow was considered part of the soil and not part of the chamber headspace.

We used the generally clear and significant increase of CO\(_2\) concentration in closed chamber at temperatures above the freezing point to check for accurate diffusive gas accumulation, which can be affected in particular by high wind speed and changing pressure conditions (Hutchinson and Mosier, 1981; Hutchinson and Livingston, 2001). Measured fluxes were subjected to a rigorous quality check since many different people were involved in the comprehensive gas sampling (about 60,000 gas samples were taken in the whole project during the three experimental years) and occasionally different anchoring of the chambers could have resulted in small leakages at the rubber sealing. Thus, the Pearson correlation coefficient between CO\(_2\) concentration and closing time was used as an indicator of the reliability of diffusive gas accumulation. If the Pearson coefficient of the CO\(_2\) flux was smaller than +0.85 and air temperature was above 0 °C, we rated the calculated diffusive flux as considerably biased by other processes and the corresponding flux measurement was excluded from analyses.

Only a few measurements indicated exceptionally high N\(_2\)O uptake rates of more than 100 μg N m\(^{-2}\) h\(^{-1}\). These fluxes, which were usually associated with abnormally high initial N\(_2\)O concentrations, were considered unreliable given that negative diffusion gradients are limited due to low concentration in the atmosphere. Single N\(_2\)O flux estimates with extraordinary high uncertainty, i.e., standard errors above 100 μg N m\(^{-2}\) h\(^{-1}\) (above 25 μg N m\(^{-2}\) h\(^{-1}\) for uptakes larger than 50 μg N m\(^{-2}\) h\(^{-1}\)) were also rated as highly unsure and removed. The 90% quantile of all flux standard errors was 11 μg N m\(^{-2}\) h\(^{-1}\) (median: 1.5 μg N m\(^{-2}\) h\(^{-1}\)). Thus, the flux detection limit was lower than approximately 2 \(\times\) SE = 22 μg N m\(^{-2}\) h\(^{-1}\) for 90% of the flux measure.

After these rigorous quality checks approximately 10,000 measured N\(_2\)O flux rates were analyzed further for the results presented hereafter. Only a very small number of sampling dates were completely lost through the quality check but it resulted in some sampling dates with a reduced number of replicates. Multiple imputation (Honaker et al., 2011) was employed to fill these gaps in plot specific N\(_2\)O time series for subsequent statistical analysis. Imputation was done between the replicate N\(_2\)O time series group-wise by sites, treatments and years. To improve the performance of imputation, data of each group was transformed as log(flux − min(flux) + 1). The number of multiple imputations was set to 25 and linear time effects were used to account for autocorrelation. The imputed fluxes were then transformed back and cumulated fluxes calculated by linear interpolation. Finally, the median of the multiple imputations was calculated and used as cumulated flux estimate.

2.4.2. Cumulative N\(_2\)O emissions and emission factor

For the calculation of annual N\(_2\)O emissions we cumulated N\(_2\)O fluxes between 1st January and 31st December for each experimental year. We defined this time period at the beginning of our investigations based on the following expectations: except for the sowing, this period covered all soil management and N fertilization measures of WOSR cultivation as well as the time period with presumably increased soil mineral N contents following WOSR harvest and under subsequent winter wheat. Since there was no N fertilization in autumn, we did not expect increased fluxes after rapeseed sowing before winter. Consequently, autumn and early winter fluxes during WOSR cropping were neglected for calculation of annual N\(_2\)O emissions.

The FRE was derived by fitting the model described by Walter et al. (2015), which is based on the methodology in Stehfest and Bouwman (2006), after including the data from this study in their dataset (12 sites from the global meta-analysis of Walter et al. (2015) and five sites from this study with three measurement years and up to five fertilization rates per site). Briefly, a linear mixed effects model (R package lme4 version 1.1–12, Bates et al. (2015), R package lmerTest version 2.0-33, Kuznetsova et al. (2016)) relating log\(_e\)-transformed annual N\(_2\)O fluxes to fertilizer N input was fitted. The model included random intercepts
for the site and year effects and a random slope for the year effect. The log_{10}-transformation accounts for the typical higher heterogeneity of larger N_2O fluxes. As a result, the modeled relationship between N_2O fluxes and fertilization rate is exponential and can be compared to previous models by Walter et al. (2015) and Stehfest and Bouwman (2006). Since the model is nonlinear, emission factors depend on the amount of N fertilizer. We report the emission factor for an N fertilization rate of 200 kg N ha^{-1} as this is the amount beyond which no further yield increases are expected under best management practices (Maidl and Limbrunner, 2008). This is also approximately the recommended and typical fertilization rate of WOSR production in Germany. Following the methodology employed by the JRC (Edwards et al., 2016) it was calculated as:

\[ EF = \frac{E_{200} - E_0}{200} \]  

(1)

where \( E_{200} \) and \( E_0 \) are emissions (kg N_2O-N ha^{-1} a^{-1}) predicted by the model at 200 kg N and 0 kg N fertilization rate, respectively. Compared to the fitted exponential relationship, this linear emission factor approach slightly overestimates emissions from lower N fertilization and underestimates emissions from higher N fertilization. However, these deviations are small as long as the amount of N fertilizer applied does not differ too substantially from typical fertilization rates. In contrast to the IPCC emission factor (EF1) but in accordance with Stehfest and Bouwman (2006), the emission factor takes into account emissions from crop residues indirectly since crop residue N was not included as N input in the model.

2.4.3. Water-filled pore space

Water-filled pore space (WFPS) was calculated as described by Ruser et al. (1998) using the bulk density measured in the top soil of the study sites and assuming a particle density for the soil of 2.65 g cm^{-3}.

2.4.4. N surplus and oil yield-related N_2O emissions

N surplus was calculated by subtraction of N removed from the field by harvest (dry matter concentration of WOSR seed yield multiplied by N concentration of the seeds) from the respective N fertilizer amount. Oil yield-related N_2O emissions for the respective fertilization treatment were calculated by relating annual N_2O emissions to the amount of oil yield, which was the product of WOSR seed yield and oil concentration in the seeds. For the study site Berge, removal of the straw was also taken into consideration.

A linear mixed effects model of log_{10}-transformed oil yield-related N_2O emissions was used to investigate differences between years and N surplus. Year was included as a fixed effect and site as a random intercept.

Finally, total GHG savings of biodiesel produced from the 180 kg N ha^{-1} WOSR treatment were calculated using the Biograce-I (version 4d, www.biograce.net) excel tool.

2.4.5. Statistical analysis

For each site and for each year we separately ran a Kruskal Wallis One Way Anova on Ranks to detect differences between the treatments concerning oil yield-related emissions. Significant differences were determined using a pairwise multiple comparison procedure (Student-Newman-Keuls, \( p < 0.05 \)).

We calculated simple Spearman Rank Order Correlation Coefficients to test for a relationship between the N_2O and CO_2 release in the postharvest period.

All other statistical analyses were conducted using the R language and environment for statistical computing (version 3.3.2, R core, 2016). Mixed-effects models were fitted using package lme4. Confidence intervals of parameters were estimated using parametric bootstrap. Parameter \( p \)-values were derived using Satterthwaite’s approximation for degrees of freedom (Satterthwaite, 1946, R package lmerTest).

Relationships between nitrous oxide fluxes and explaining variables were investigated using Generalized Additive Models (GAM, R package mgcv version 1.8-16, Wood, 2011), which can model non-linear relationships such as the optimum curve typically observed for N_2O emissions vs. soil moisture.

3. Results and discussion

3.1. Meteorological conditions and seasonal N_2O fluxes

3.1.1. Meteorological conditions

Compared to the long-term mean air temperature at every single site, the annual temperature was higher in all experimental years and, except for Hohenschulen, at all study sites (Table 1). Highest annual air temperatures were measured at all sites in the second year of our investigations (2014), followed by the third year (2015). Additionally, considering annual precipitation, which was, depending on site, lowest in the second or third year of measurements, it became obvious that the climatic conditions during our experiment covered a year representing or slightly exceeding long-term conditions (2013), a year with average precipitation and higher temperatures and one year with rather drier and warmer conditions.

3.1.2. N_2O fluxes during the growing season

Spatial and temporal variability of N_2O fluxes was very high (Fig. 1). At all sites, increases of the N_2O fluxes were often detected after N fertilization in conjunction with rainfall events. The highest N_2O flux rate (670 μg N_2O-N m^{-2} h^{-1}) during the whole investigation period in the treatment with 180 kg N ha^{-1} a^{-1} was measured at the Merbitz study site following a heavy rainfall event (42 mm d^{-1}) one week after the second N fertilization in mid May 2013. Prior to this high N_2O pulse, precipitation after two weeks without rainfall had stimulated N_2O release after the first N fertilizer application at the same site.

Although the magnitude of N_2O flux rates differed between study sites and years, we frequently observed short-term N_2O pulses at all study sites after heavy rainfall events throughout the whole growing season as well as after rewetting of solidly dried soil in summer (Fig. 1).

Similar patterns of the N_2O release from arable soils with increased flux rates after N fertilization and rainfall were often reported and explained with enhanced denitrification due to (i) an increased availability of nitrate as substrate for N_2O production, and (ii) due to the formation of anaerobic conditions as a result of lower gas diffusivity in soil water and thus of a reduced O_2 diffusion into the soil combined with O_2 consumption by soil microbes (Flessa et al., 1995; Corre et al., 1996; MacKenzie et al., 1997).

During the first six weeks postharvest we also frequently observed increased N_2O fluxes following rainfall. Monthly fluxes following harvest were often comparable or even exceeding fluxes after N fertilization (Fig. S1). In the post-harvest period we could not see any differentiating effect on the N_2O flux rates, independent of whether the WOSR residues were incorporated into the soil or remained on the surface. Although we did not include WOSR residue incorporation or surface application in our experimental design, we had study sites where we measured increased N_2O fluxes after rainfall when the residues remained on the surface and also after later incorporation (i.e. Dedelow 2014).

Nett et al. (2015) compared the effect of the incorporation of N-rich cauliflower residues on N_2O fluxes with a treatment where the residues remained as mulch on the soil surface. They did not find significant differences of the N_2O release between these two treatments. Baggs et al. (2003) and Escobar et al. (2010) even reported higher fluxes when legume crop residues remained on the soil surface instead of being incorporated.

In this postharvest period after WOSR, nitrate contents in the top soil were generally elevated (Fig. S2). Mosier et al. (1983) reported a threshold of 10 mg nitrate N kg^{-1} soil above which denitrification rates were independent of the soil nitrate concentration. Nitrate-N
concentrations in our study often reached this level immediately after harvest. We therefore concluded that soil nitrate was not a limiting factor for the denitrifying community after harvest and consequently we assume that C availability and corresponding microbial activity and oxygen consumption played a major role as the driver for postharvest N₂O release. This assumption was supported by the statistically highly significant and positive correlations between the N₂O and CO₂ flux rates in the time between harvest of WOSR and seeding of the succeeding winter wheat (Spearman rank correlation coefficients in the time between harvest of WOSR and seeding of the succeeding winter wheat (Spearman rank correlation coefficients). Induced N₂O pulses increase with increasing duration of frost periods (90 kg N ha⁻¹) each. Note: different y-axis scaling.

In case of the surface located WOSR residues this C must have been leached from the crop residues during rainfall into the soil or solubilized in regions of the residues with direct soil contact. An alternative explanation could be N₂O production directly from the decomposing plant material (Flessa et al., 2002). Müller et al. (2003) compared the turnover dynamics of different plant residues for modeling purposes.

In contrast to earlier investigations on N₂O flux rates from arable fields or grassland in Germany (Flessa et al., 1995; Kammann et al., 1998; Röver et al., 1998; Kaiser and Ruser, 2000; Ruser et al., 2001), we did not observe considerable N₂O pulses during thawing of frozen soil. We assume that the mild winters in all three experimental years without any severe frost periods were probably the main reason for the low N₂O flux rates observed in our study. It was often shown that frost/thaw induced N₂O pulses increase with increasing duration of frost periods and with severity of the soil freezing (Teepe et al., 2004; Wagner-Riddle et al., 2007; Risk et al., 2013; Xu et al., 2016). Except for the first two weeks of our measurements in 2013, soil temperatures in 10 cm depth did not drop below −2 °C for more than one week at all sites (not shown) and hence the conditions during our field experiment did not enable distinct frost/thaw induced N₂O pulses.

Over the whole data set, we could explain 27% of the variability of the N₂O flux rates (in the treatment with 180 kg N ha⁻¹) with the generalized additive model (Table S1). All smooth terms with a statistically significant contribution to the model results were recorded at the study sites Merbitz, Ihinger Hof, and Hohenschulen, whereas we did not find significant correlations between the smooth terms and the log₁₀-transformed N₂O flux rates at Berge and Dedelow (Fig. S3). We found a relationship between the temperature and soil moisture and the N₂O flux rates for the sites Merbitz, Ihinger Hof, and Hohenschulen. The moisture optimum appeared to be around 50% WFPS. The fluxes at Merbitz were also related to the nitrate contents of the top soil (p < 0.001). Enhanced N₂O flux rates with increasing soil moisture and partly with increasing nitrate contents suggest denitrification as a major source for the N₂O released at the sites Merbitz, Hohenschulen, and Ihinger Hof.

3.2. Effect of N fertilization on the N₂O fluxes

Over all sites combined, increasing N fertilization significantly enhanced N₂O flux rates (p < 0.001, Table 3). This effect was more apparent at sites with higher N₂O flux level (Table 2). In contrast, N fertilization effects did not appear at Berge and Dedelow, the sites with the lowest flux levels. Following N fertilization, nitrate and ammonium contents generally increased with increasing N amounts. The increased nitrate contents after N fertilization served as available substrate for N₂O production under conditions supporting denitrification. In contrast, N₂O flux rates were negligible under conditions favorable for nitrification (high ammonium concentrations and soil moisture contents below field capacity).
times higher than in Berge and Dedelow, respectively (Table 1). For
the period between 1st January and the sowing of wheat in autumn of
the first experimental year, our flux chambers covered bare soil. The CO2
flux rates (which are only a rough estimate due to the chamber closing
times being optimized for N2O flux measurements) are therefore an
indicator for C mineralization. The mean CO2 flux rate at Hohenschulen
in that period was 86.9 mg CO2-C m–2 h–1. It was 1.6 and 2.3 times
higher than the corresponding mean flux rates at Berge and Dedelow.
We therefore presume that the higher soil respiration rates at Ho-
henschulen decreased O2 availability, thus favoring anaerobic con-
tions and N2O production during denitrification whereas O2 supply in
the soils in Berge and Dedelow was sufficient to impede nitrate re-
duction. This would explain the higher N2O fluxes compared to the
other sandy sites (Table 2).

### 3.4. Inter-annual variability of the N2O fluxes

The median annual N2O flux at all sites exhibited a high variability
and the effect of the experimental years was statistically significant and
of the same order of magnitude as the site effect (Table 3). The highest
inter-annual variability was measured at the study site Ihinger Hof in
the treatment with 180 kg N ha–1 a–1. At this site, the flux was 5 times
higher in 2013 than in 2015 (Table 2).

A main driver for the inter-annual variability of the annual N2O flux
rates was rainfall shortly after N fertilization or harvest. In the treat-
ment with 180 kg N ha–1 a–1, the annual N2O flux rate decreased at all
experimental sites in the same order as the annual precipitation de-
creased (Fig. 2). This clearly indicates that local weather conditions and
in particular the amount of rainfall (especially in months with increased
mineral N availability as, for example after N fertilization; Fig. S2), play
a key role in determining the magnitude of N2O flux rates.

A high inter-annual variability of N2O fluxes has often been reported
in field studies with N2O measurements (Dobbie et al., 1999; Pfäb et al.,
2011; Reeves and Wang, 2015) as well as in modeling approaches from
sites with different climate conditions (Leip et al., 2011) or with dif-
f erent climate scenarios (Ben Aoun et al., 2016). Despite a uniform
management (N fertilization, crop type) annual N2O emission varied by
up to factor seven between the single experimental years. These dif-
f erences also resulted from different weather conditions during the
study, with rainfall being one of the dominant drivers for N2O release
from soils (Smith et al., 1998; Dobbie et al., 1999; Laville et al., 2011).

### 3.5. Cumulative N2O emissions and fertilizer-related N2O emissions

Due to high variability of N2O fluxes, cumulative N2O emissions
were also scattered widely over the study sites and experimental years.
For the treatment fertilized with 180 kg N ha–1 a–1, annual N2O
emission varied between 0.24 kg N2O-N ha–1 a–1 (Berge, 2014) and
5.48 kg N2O-N ha–1 a–1 (Merbitz, 2013) (Fig. 3). The magnitude of an-
nual N2O emissions was in the same range as those assembled by
Walter et al. (2015) in their meta-analysis on the effect of N fertilization
on N2O emissions from WOSR fields. For N fertilizer amounts approx-
imately in the same range as our 180 kg N ha–1 a–1 treatment Walter et al.
(2015) reported annual N2O emissions from WOSR fields ranging between 0.31 and 5.61 kg N2O-
N ha–1 a–1.

Since N2O flux rates were stimulated with increasing N fertilizer
amount, cumulative N2O emissions also increased with N fertilization
(Fig. 3). We supplemented the dataset of Walter et al. (2015) with our
data, thereby doubling the number of data points, and following their
methodology, derived an exponential model relating N2O emissions to
N fertilization (Table 3). The model confirmed a strong impact of study
sites and years on annual N2O fluxes. A nonlinear response of N2O
emissions to N fertilization has often been reported and explained ei-
ther with an increased N supply strongly exceeding N demand of the
crop or with extended periods of increased mineral N supply for N2O
Fig. 2. Median N\textsubscript{2}O flux and annual precipitation at the five study sites. Note: different y-axis scaling and non-equidistant x-axis units.

Fig. 3. Mean measured cumulative annual N\textsubscript{2}O emission ($n = 4$, ± standard deviation) at different nitrogen fertilization rates. Lines depict site-specific and mean N\textsubscript{2}O emissions modeled with the mixed effects model described in Table 3.
production (van Groeningen et al., 2010; Hohen et al., 2011; Shcherbak et al., 2014).

The German legislation on N fertilization (DüV, 2006) currently allows for an N surplus (N fertilization minus N removal with harvest) of 60 kg N ha$^{-1}$a$^{-1}$ (mean value of three years). Assuming a high WOSR yield of 5 Mg ha$^{-1}$a$^{-1}$ we can deduce a crop demand of 227 kg N ha$^{-1}$a$^{-1}$ (5 Mg ha$^{-1}$a$^{-1}$ × 45.4 kg N Mg$^{-1}$, cf. Table 1 of DüV, 2006). On a legislative basis, it can therefore be expected that N fertilization in German WOSR production potentially varies between 0 and 287 kg N ha$^{-1}$. Over this range of N fertilization, all proposed nonlinear models for the N$_2$O emission – N fertilization rate relationship (Fig. 4) result in lower fertilizer-related N$_2$O emissions when compared to the linear IPCC Tier 1 approach.

Comparing the impact of different emission factors for direct N$_2$O field emissions from WOSR cultivation in Poland, Syp et al. (2016) also reported higher N$_2$O emission calculated with the BioGrace approach (IPCC default values, Tier 1) compared to the GNOC. Assuming a fertilizer amount of 200 kg N ha$^{-1}$a$^{-1}$ the global FRE factor derived from the exponential model was 0.6% (CI: 0.31%–1.00%). This factor is within the uncertainty range of the EF1 IPCC emission factor (0.3%–3%), but about 40% lower than the IPCC default value and was also lower than the FRE calculated by GNOC and by Walter et al. (2015) (Fig. 4).

One reason for the lower FRE in our experiment may be the fact that two of our five study sites have sandy, well aerated soils with low C$_{org}$ contents. These were chosen because they are representative for a large part of the German WOSR production area. A second reason might be the absence of distinct frost/thaw cycles at all study sites. As a consequence, the absence of frost/thaw cycles results in N$_2$O emissions about half as high as in case of frost/thaw cycle occurrence. However, such mild winters with less frost/thaw cycles seem to have become more frequent in Germany as a result of ongoing climate change (Kreyling and Henry, 2011).

The low rapeseed FRE factor is in good agreement with results from recent studies in the UK which also observed lower FRE factors than the IPCC default factor (Bell et al., 2015). Based on multiple field experiments with different crops including WOSR a new national emission factor of 0.4% was derived for the UK (Sylvester-Bradley et al., 2015). Similarly, a recent multi-site study in France, again including WOSR in the field experiments, observed lower N$_2$O emissions than expected from the IPCC default emission factor (Le Gall et al., 2014).

Table 5: Mean oil yield-related N$_2$O emission (kg N$_2$O-N Mg$^{-1}$ oil ha$^{-1}$) as affected by study site, N fertilization and experimental year. Different letters indicate statistically significant differences between the N fertilizer treatments within one year and one study site (Student-Newman-Keuls Method used for comparison of three or more treatments, t-test for comparison of two treatments, p < 0.05).

<table>
<thead>
<tr>
<th>Study site</th>
<th>Year</th>
<th>N fertilization kg N ha$^{-1}$a$^{-1}$</th>
<th>0</th>
<th>120</th>
<th>180</th>
<th>240</th>
</tr>
</thead>
<tbody>
<tr>
<td>Berge</td>
<td>2013</td>
<td>1.17 $^b$</td>
<td>1.29 $^{bc}$</td>
<td>1.38 $^{bc}$</td>
<td>1.38 $^{bc}$</td>
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</tr>
<tr>
<td></td>
<td>2014</td>
<td>1.63 $^c$</td>
<td>1.92 $^{bc}$</td>
<td>2.28 $^{bc}$</td>
<td>2.42 $^a$</td>
<td>2.56 $^a$</td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>0.97 $^c$</td>
<td>1.50 $^b$</td>
<td>1.74 $^a$</td>
<td>1.87 $^a$</td>
<td>1.85 $^a$</td>
</tr>
<tr>
<td>Mean</td>
<td></td>
<td>1.26 $^{bc}$</td>
<td>1.57 $^a$</td>
<td>1.80 $^a$</td>
<td>1.89 $^a$</td>
<td>2.04 $^a$</td>
</tr>
<tr>
<td>Dedelow</td>
<td>2013</td>
<td>2.36 $^a$</td>
<td>2.62 $^a$</td>
<td>2.73 $^a$</td>
<td>2.79 $^a$</td>
<td>2.78 $^a$</td>
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<td></td>
<td>2014</td>
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<td></td>
<td>2015</td>
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<td>2.50 $^a$</td>
<td>2.53 $^a$</td>
<td>2.54 $^a$</td>
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<tr>
<td>Hühner Hof</td>
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<td>1.78 $^a$</td>
<td>2.01 $^a$</td>
<td>1.96 $^a$</td>
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</tr>
<tr>
<td></td>
<td>2014</td>
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<td>1.76 $^{bc}$</td>
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<td>1.76 $^{bc}$</td>
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<tr>
<td></td>
<td>2014</td>
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<td></td>
<td>2015</td>
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<td>2.29 $^{bc}$</td>
<td>2.38 $^{bc}$</td>
</tr>
<tr>
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<td>1.88 $^{bc}$</td>
<td>1.96 $^a$</td>
</tr>
<tr>
<td></td>
<td>2014</td>
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<td>1.61 $^c$</td>
<td>2.09 $^b$</td>
<td>2.23 $^b$</td>
<td>2.38 $^b$</td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>1.11 $^b$</td>
<td>1.30 $^e$</td>
<td>1.67 $^a$</td>
<td>1.78 $^a$</td>
<td>1.64 $^a$</td>
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<tr>
<td>Mean</td>
<td></td>
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<td>1.55 $^a$</td>
<td>1.88 $^a$</td>
<td>1.96 $^a$</td>
<td>2.00 $^a$</td>
</tr>
</tbody>
</table>

- not determined/not calculated.
- Mean values were only calculated for treatments with 3 years measurements.
and 4.03 Mg ha\(^{-1}\)a\(^{-1}\) in 2013, 2014, and 2015, respectively. These yields follow the pattern of mean German WOSR yields, which were 3.96, 4.48, and 3.91 Mg ha\(^{-1}\)a\(^{-1}\) in 2013, 2014, and 2015, respectively (German Federal Statistical Office, 2017). Research studies commonly achieve better yields than commercial farms, but the identical annual pattern emphasizes the representativeness of our compilation of study sites for German WOSR production.

The reason for the high yield at Dedelow remains unclear. A possible explanation is a pool of easily mineralizable N, resulting from long-term application of organic fertilizers to silage maize and sugar beet approximately every second year before our experiment. However, C\(_{\text{org}}\) and total N contents at this site were comparatively low. Unfortunately, we did not determine N contents of the WOSR seeds in every year and at every site since we focused on oil yield as the relevant target yield. However, for the year 2015 we determined an N uptake in the seeds of 123 kg N ha\(^{-1}\) in the unfertilized treatment at Dedelow. The only study site with N uptake measurements in the unfertilized treatment in the same year was Merbitz with 59 kg N ha\(^{-1}\). Further N uptake data for unfertilized treatments at other sites than Dedelow in 2013 and 2014 varied between 54 and 93 kg N ha\(^{-1}\). The high N uptake in the rapeseeds at Dedelow was not only a result of increased biomass growth but also enhanced N concentrations (not shown). Both, high biomass and N concentration in the seed of the unfertilized treatment in Dedelow indicates a high and easily available N delivery at

![Fig. 5. Relationship between mean N surplus and mean oil yield-related N\(_2\)O emission (n = 4, ± standard deviation) as affected by study site and experimental year. At the site Berge straw removal was also taken into consideration for the calculation of the N surplus. Note: different y-axis scaling.](image-url)
this site. The median flux over all sites and years in the treatment with 180 kg N ha\(^{-1}\) a\(^{-1}\) related to grain yield was 0.22 kg N\(_2\)O-N Mg\(^{-1}\) oilseed grain.

Oil yield varied between 0.97 and 2.79 Mg ha\(^{-1}\) a\(^{-1}\) (Table 4), the median oil yield was 2.00 Mg ha\(^{-1}\) a\(^{-1}\). Due to the high seed yield, oil yield was also highest at study site Dedelow, where 2.25 Mg ha\(^{-1}\) a\(^{-1}\) was achieved even in the unfertilized treatment. With few exceptions (Hohenschulen 2015 and Merbitz 2014) N fertilization higher than 120 kg N ha\(^{-1}\) did not result in statistically significant increases in oil yield.

This result is in agreement with Hegewald et al. (2016) who reported only small increases in oil yield (0.04 Mg ha\(^{-1}\) a\(^{-1}\)) when N fertilization was increased from 120 to 180 kg N ha\(^{-1}\) a\(^{-1}\) in a study on the effect of different preceding crops on WOSR yield. As mentioned by Rathke et al. (2006) N fertilization increases the crude protein content of rapeseeds at the expense of oil concentration. We also observed declining oil contents with increasing N fertilization, however, due to higher dry matter development with increasing N supply, oil yields were stable over the N fertilizer range between 120 and 240 kg N ha\(^{-1}\) a\(^{-1}\) (Table 4).

Oil yield–related \(N_2O\) emissions varied depending on site and year. The median yield-related \(N_2O\) emission over the entire data set was 0.46 kg N\(_2\)O-N Mg\(^{-1}\) oil. Cumulative annual \(N_2O\) emissions, and consequently oil yield-related \(N_2O\) emissions, were lowest at study sites Berge and Dedelow (Table 5). For 2014 and 2015 we found a tendency at Dedelow for increasing yield-related emission with increasing N fertilization.

In contrast, oil yield-related \(N_2O\) emissions were distinctly higher in all years for the site Merbitz with its silty Chernozem soil and also higher for the sites Hinger Hof and Hohenschulen (Table 5). Increasing N fertilization at these three sites by 60 kg N ha\(^{-1}\) from 120 to 180 kg N ha\(^{-1}\) a\(^{-1}\) resulted in a slight increase (approximately 0.15 kg N\(_2\)O-N Mg\(^{-1}\)) in yield-related \(N_2O\) emissions whereas the application of another 60 kg N ha\(^{-1}\) a\(^{-1}\) (in total 240 kg N ha\(^{-1}\) a\(^{-1}\) approximately doubled the yield-related emissions in the respective years with trace gas measurements indicating a threshold for strongly enhanced yield-related \(N_2O\) emissions between fertilization intensities of 180 and 240 kg N ha\(^{-1}\) a\(^{-1}\).

Different functions have been used to visualize the relation of yield-related \(N_2O\) emissions to N surplus. Van Groenigen et al. (2010) used an exponential function to describe the yield-related \(N_2O\) emissions with N surplus in silage maize. Walter et al. (2015) fitted a segmented linear function to their WOSR yield dry mass-related \(N_2O\) emissions and found a critical N surplus of 80 kg N ha\(^{-1}\) where the yield-related \(N_2O\) emissions substantially increased. We found neither a clear threshold in our experiment nor a simple global relationship. This can be attributed to the high inter-annual variability of the \(N_2O\) emissions (Fig. 5). Especially in 2013, \(N_2O\) emissions were driven by fertilization whereas emissions in 2014 and 2015 did depend less on fertilization and were generally lower. Oil yield-related \(N_2O\) emissions at the study sites Hinger Hof and Merbitz increased with a small N surplus in 2013, the year with the highest precipitation, whereas the threshold for increased yield-related \(N_2O\) emissions under drier conditions in 2014 and 2015 was approximately +50 kg N ha\(^{-1}\) N surplus. The site Hohenschulen also exhibited a distinct increase of yield-related \(N_2O\) emission with rising N surplus, whereas the low emission sites Berge and Dedelow did not respond to varying N surpluses. The removal of WOSR straw at the site Berge resulting in lower N surplus values might have additionally affected \(N_2O\) emissions.

Calculating the GHG balance of biodiesel produced from the 180 kg N ha\(^{-1}\) WOSR treatment according to current EU RED methodology (i.e., using the IPCC emission factor of 1%) resulted in GHG savings of 44% and 51% compared to the current and updated fossil fuel reference, respectively (Table 6). Note that both yield (4990 kg ha\(^{-1}\)) and N fertilization of the 180 kg N treatment were much higher than the EU RED default values (3113 kg ha\(^{-1}\) with 137.4 kg N fertilization). Substituting the EF1 IPCC emission factor with the rapeseed specific emissions factor of 0.6% for the calculation of direct \(N_2O\) field emissions from fertilizer N input improved GHG savings to 47% and 54%, respectively. If we assume that the same WOSR-specific emission factor is also valid for emissions from the crop residues, GHG savings even achieve 50% and 56%, respectively. Thus, a full update of the emission factor results in a calculative increase of GHG savings by five to six percentage points. As a result the 50% goal of the EU RED would be achieved even with the original reference value.

4. Conclusions

With our study covering three years at five study sites in representative regions of German WOSR production, we doubled the experimental base (now 86 annual flux values) for the deduction of an emission factor for fertilizer-related \(N_2O\) emissions from WOSR cropping. Our measurements confirmed the result of Walter et al. (2015) that fertilizer-related \(N_2O\) emissions from WOSR are markedly lower than the (linear) one percent default value proposed by the IPCC Tier 1 approach. For a common N fertilizer amount of 200 kg N ha\(^{-1}\) a\(^{-1}\) in German WOSR cultivation the fertilizer-related emission factor developed from our data set combined with the data assembled by Walter et al. (2015) is 0.6%. Applying this WOSR-specific emission factor to the N fertilizer amount used in our experiment (180 kg N ha\(^{-1}\) a\(^{-1}\)), which is similar to the amounts used under best agricultural management conditions, we could show, that the GHG reduction criteria stipulated by the Renewable Energy Directive for biofuels can be fulfilled for existing biofuel plants: the \(N_2O\) emissions from biofuel production achieved more than 50% GHG savings compared to fossil diesel exploration and consumption, especially if best management practices are adopted in WOSR cultivation.

Our oil yield results indicate that there is potential for reduction of N fertilization in comparison to agricultural practice recommendations.
References


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References

Without oil yield losses. This would be an excellent mitigation option due to avoiding GHG emissions during fertilization production and the reduction of direct and indirect N₂O field emissions.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2017.07.039.

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