Different Land Use Intensities in Grassland Ecosystems Drive Ecology of Microbial Communities Involved in Nitrogen Turnover in Soil

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Abstract

Understanding factors driving the ecology of N cycling microbial communities is of central importance for sustainable land use. In this study we report changes of abundance of denitrifiers, nitrifiers and nitrogen-fixing microorganisms (based on qPCR data for selected functional genes) in response to different land use intensity levels and the consequences for potential turnover rates. We investigated selected grassland sites being comparable with respect to soil type and climatic conditions, which have been continuously treated for many years as intensely used meadows (IM), intensely used mown pastures (IP) and extensively used pastures (EP), respectively. The obtained data were linked to above ground biodiversity pattern as well as water extractable fractions of nitrogen and carbon in soil. Shifts in land use intensity changed plant community composition from systems dominated by s-strategists in extensive managed grasslands to c-strategist dominated communities in intensive managed grasslands. Along the different types of land use intensity, the availability of inorganic nitrogen regulated the abundance of bacterial and archaeal ammonia oxidizers. In contrast, the amount of dissolved organic nitrogen determined the abundance of denitrifiers (nirS and nirK). The high abundance of nifH carrying bacteria at intensive managed sites gave evidence that the amounts of substrates as energy source outcompete the high availability of inorganic nitrogen in these sites. Overall, we revealed that abundance and function of microorganisms involved in key processes of inorganic N cycling (nitrification, denitrification and N fixation) might be independently regulated by different abiotic and biotic factors in response to land use intensity.


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Introduction

Soils provide a large number of ecosystem services including plant growth, carbon sequestration, degradation of xenobiotics and safeguarding of drinking water resources. Most of these functions are closely linked to the soil microbiome and its activity pattern [1,2,3]. Therefore many attempts were made to identify soil borne microbial communities as key drivers of ecological processes and describe factors that drive the abundance and diversity of selected functional communities [4]. Despite the high heterogeneity of soil microbes in time and space, it has become possible to figure out one general conclusion from these studies: Besides site-specific parameters, for example soil texture or climatic conditions, the type of land management and land use intensity has been identified as a major driver for microbial performance in soil [5,6,7,8].

Recently, the effects of land use changes have been studied mainly focusing on (i) conversion of grassland to forest or vice versa [9], (ii) alterations in tillage management [10,11,12], (iii) changes in crop rotation [13] or (iv) modifications in fertilizer quality [14]. However, studies addressing questions related to consequences of changes in land use intensity on the soil microbiome are rare, although in many parts of the world we are facing a tremendous increase in land use intensity, due to the demands of bioeconomy (production of food, feed, fuel and fiber).

This intensification is also frequently observed in grassland ecosystems. While in the past sites have been used extensively as pastures, nowadays up to four times per season, the same areas are managed as meadows for hay production and silage, entailing an intensive application of organic and inorganic fertilizers. Differences in intensity of agricultural practice like mowing, grazing and fertilization lead to changes in plant composition [15,16,17,13], microclimate, soil quality and hence to changes on macro- as well as micro-scale habitats. For some soil animals the impact of such changes is well known [18,19,20,21] but data on microbial communities in soil is rare. For example, [6] compared diversity pattern of microbial community involved in nitrogen fixation, denitrification and nitrification in grassland ecosystems under...
different management intensities. This study clearly demonstrated changes in diversity pattern of single functional groups involved in nitrogen transformation on low diverse grassland sites. However, this study did not address questions how land use intensity influences the abundance and activity of selected functional groups of microbes in soil and thus changes turnover processes and rates.

The aim of the present study was to characterize microbial communities responsible for key processes in the inorganic nitrogen cycle (nitrification, denitrification and N-fixation) in grasslands of different land use intensity and relate these results to the aboveground biodiversity of plants as well as important belowground properties (water extractable carbon and nitrogen fractions). Overall, we postulated that nitrogen cycling at extensively used sites is mainly driven by nitrogen fixation and internal nitrogen turnover is highly efficient resulting in low denitrification rates. In contrast, denitrification may play a more important role in nitrogen turnover at intensely used sites due to higher amounts of nitrogen available in soil and intensive plant growth (resulting in higher root exudation rates and increased microbial activity).

Materials and Methods

Experimental Setup and Sampling

Experiments were carried out in the frame of the German Biodiversity Exploratories [22], which form an ideal platform for such type of studies, as here for the first time gradients in land use intensity were defined on a large scale in three regions in Germany. For the present study soil samples was taken in 2008 in such type of studies, as here for the first time gradients in land use intensity were defined on a large scale in three regions in Germany. The mean annual precipitation in this area ranged from more than 45,000 ha of the state of Baden-Württemberg in SW-Germany. The mean annual precipitation in this area ranged from 938–963 mm, whereas the annual mean temperature was around 7°C. All sampled soils have been described as a Rendzic Leptosols with a clayey or loamy texture and a pH-value between 5.7 and 6.9. A more detailed soil description is given in table 1. Field work permits were given by the responsible state environmental offices of the state of Baden Württemberg (according to § 72 BbgNatSchG).

Nine different grassland sites (AEG 1 – 9 with the given coordinates: N48°23'36" E9°20'31"; N48°23'15" E9°28'22"; N48°24'29" E9°32'2"; N48°23'38" E9°25'5"; N48°23'48" E9°26'21"; N48°24'11" E9°26'31"; N48°23'33" E9°22'57"; N48°26'13" E9°29'32"; N48°23'44" E9°30'10") were sampled categorized as follows: intensely used meadows (AEG 1 – 3 = IM, three times manure application and two times mown per season), intensely used mown pasture (AEG 4–6 = IP, grazed by cattle and horses, mown once a year and two times manure application per season) and extensively used pasture (AEG7 - 9 = EIP, unfertilized but infrequently grazed by sheep). All sites were sampled in early spring (April) as well as in summer (July) to assess the effect of season.

From each site five sampling replicates were taken from the upper 10 cm; each replicate consist of five pooled bulk soil cores (d = 5.5 cm) taken with a soil auger. All samples were cooled directly after sampling for DNA based analyses at −20°C and at 4°C for measurements of enzyme activities and soil parameter.

Plant Diversity

At all investigated sites the vegetation has been recorded on an area of 4 × 4 m. Plants were identified on taxa level and their percentage cover was estimated separately for the shrub layer (0–5 m woody species) and the herbaceous layer (including phanerophyte seedlings), respectively (data here not shown). The ecological strategy type of each vascular plant according to [23] was determined using the “Bioflor” data base (www.bioflor.de). This concept describing the general limits to ecology and evolution based on the trade-off that organisms face when the resources they gain from the environment are allocated between either growth, maintenance or regeneration – known as the universal three-way trade-off: (C) the survival of the individual using traits that maximise resource acquisition and resource control in consistently productive niches, (S) individual survival via maintenance of metabolic performance in variable and unproductive niches, or (R) rapid gene propagation via rapid completion of the lifecycle and regeneration in niches where events are frequently lethal to the individual. Based on this assumption the csr triangle has been

<table>
<thead>
<tr>
<th>Site</th>
<th>Soil typea</th>
<th>Horizonb</th>
<th>Soil depth</th>
<th>Texturec</th>
<th>ph</th>
<th>Corg</th>
<th>N</th>
<th>C/N</th>
</tr>
</thead>
<tbody>
<tr>
<td>IM 1d</td>
<td>Rendzic Leptosol Ah</td>
<td>16</td>
<td>40</td>
<td>540</td>
<td>420</td>
<td>6.7</td>
<td>68.7</td>
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<td>6.9</td>
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<tr>
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<td>450</td>
<td>530</td>
<td>6.4</td>
<td>51.8</td>
<td>5.3</td>
</tr>
<tr>
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<td>80</td>
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<td>5.2</td>
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<td>480</td>
<td>6.1</td>
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<td>190</td>
<td>7.2</td>
<td>40.4</td>
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<tr>
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<td>600</td>
<td>6.5</td>
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<td>40</td>
<td>680</td>
<td>270</td>
<td>6.7</td>
<td>67.6</td>
<td>5.9</td>
</tr>
</tbody>
</table>

aWorld Reference Base for soil resources, IUSS Working Group WRB.
bHorizon designation according to Guidelines for profile description, FAO.
cSoil texture was determined according to Schlichting & Blume (1966).
dIntensely used meadow.
eIntensively used pasture.

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defined, in which each plant could be integrated according to its individual life type strategies. Furthermore, site conditions were characterized by calculating the mean indicator values based on plant species composition according to [24]. Ellenberg indicator values represent a well-established method for bio-indication of a range of environmental parameters.

**Chemical Analyses of Soils**

CaCl₂ extracts of the soil samples were prepared to analyze carbon and nitrogen contents in water extractable fraction. Therefore soil was mixed with 0.01 M CaCl₂ at a ratio of 1:2 [25] and homogenized by overhead shaker for 2 h followed by a filtration step. Water extractable organic carbon (WEOC) and total nitrogen bound (TNb) were measured using a DIMA-TOC 100 (DIMATEC Analysentechnik, Essen, Germany), nitrate and ammonium by using the commercial kits Nanocolor N 100 (DIMATEC Analysentechnik, Essen, Germany), nitrate and ammonia oxidizing bacteria and archaea, nitrogen fixing microbes as well as nitrite reducers. The marker genes used as well as the corresponding conditions for qPCR are listed in table 2.

In a pre-experiment a dilution of the DNA of 1:64 turned out to be best suited avoid inhibition of PCR, e.g. by co-extracted humic substances (data not shown). Quantitative real-time PCR was performed in 96-well plates (Applied Biosystems). Reaction mixtures with total volume of 25 μl were composed of 12.5 μl of Power SybrGreen Master Mix (12.5 μl), BSA (15 μg), template (2 μl) as well as primer and DMSO in a final concentration as referred in the table.

**Real-time PCR Assay**

Quantitative real-time PCR (qPCR) was conducted on a 7300 Real-Time PCR System (Applied Biosystems, Germany) using SybrGreen as fluorescent dye to estimate the abundance of ammonia oxidizing bacteria and archaea, nitrogen fixing microbes as well as nitrite reducers. The marker genes used as well as the corresponding conditions for qPCR are listed in table 2.

**Statistical Analysis**

All results were related on the basis of one gram of soil dry weight (g dw⁻¹). Using the R environment for statistical computing (http://www.R-project.org) univariate and multivar-
icate methods were applied to test the effect of land use and season. Assumptions were tested by checking the residuals (equal variance in the groups, no outliers, normal distribution). To account for the existence of replicate groups (n = 5) in each case linear mixed-effect models (lme in package nlme) were fitted. Two factor lines with pairwise comparisons were used to construct a table of p-values. The integrative multivariate data analysis was based on a between group analysis (between in package ade4).

To investigate differences in plant diversity data a one factor ANOVA (lm in R) and subsequent pairwise t-tests with adjusted p-values (pairwise.t.test with method holm) were applied.

Results

Plant Diversity

The number of plant species on the investigated sites ranged between 17 and 58. Significantly lower numbers were observed on the intensively managed plots (24 respectively 25 for IM and IP) compared to the EP plots where 49 different species were found (p = 0.019). The coverage of vascular plant was between 102 and 170%, with highest plant coverage on the IM plots. The plant species composition showed a clear dependency on the land use intensity level (table S1). While grasses were dominant on the intensively used sites were classified as indicators of moderate to well water conditions in soils.

On the extensively used sites were classified as indicators for nitrogen deficiency, whereas plants of the collected plants we found a clear trend for plants more linked to r-strategists on the extensively used sites (up to 96% of coverage) indicating that grasses are more adapted to the disturbance caused by mowing. Considering the different ecological strategy types of the collected plants we found a clear trend for plants more linked to c-strategists on the extensively used sites (up to 96% of the coverage).

The analysis of plants with indicator values for abiotic sites properties according to Ellenberg revealed a strong response of the plant community structure towards nutrient availability and soil water content (table 3). Most plant species from EP sites could be classified as indicators for nitrogen deficiency, whereas plants of intensively used sites were classified as indicators of moderate to strong nitrogen concentrations in soil. Additionally plants grown on the EP plots were better adapted to water shortage and matched well with plants being an indicator for moderate water shortage compared to plants obtained from the IP and IM plots which are indicators for moderate to well water conditions in soils.

Labile Soil Nitrogen and Carbon Pools

Soil ammonium content ranged from 0.19 to 4.1 μg N g⁻¹ dw and was up to ten times higher in spring than in summer (p = 0.0444). In July values were consistently low (0.19 and 0.53 μg N g⁻¹ dw) in dependent from the site, whereas in April (0.62 to 4.1 μg N g⁻¹ dw) differences between sites were more pronounced, with a tendency for higher values in soil samples from EP and IM compared to IP (p = 0.2976). Nitrate concentrations were in the range of 1.61 and 54.5 μg N g⁻¹ dw in April and 1.10 and 18.2 μg N g⁻¹ dw in July. No significant influence of land use and time was detected (p = 0.2162; p = 0.1246), however samples taken in April showed a tendency for higher values than those taken in July. Water extractable nitrogen (TNb) contents were strongly linked to the nitrate concentration with values from 4.85 to 62.6 μg N g⁻¹ dw in April and in 4.09 to 23.3 μg N g⁻¹ dw in July. WEOC values ranging from 30.6–57.4 μg C g⁻¹ dw were neither affected by land use intensity (p = 0.9706) nor by season (p = 0.6837). Data are summarized in table 3 and 4.

Potential Nitrification and Denitrification Rates

The highest PNR values were measured in samples derived from IM sites. However transformation rates between the three investigated sites (AEG1 - 3) were highly variable with values from 0.26 up to 2.55 μgNO₂⁻ N g⁻¹ dw h⁻¹ (figure 1 and table 4). PNR in samples from plots characterized as IP ranged between 0.26 and 1.91 μgNO₂⁻ N g⁻¹ dw h⁻¹ and rates were neither significant different to IM nor to EP (p = 0.2944) plots. Again a high variation of the values, comparing the three sites under investigation (AEG1 - 6), was visible. The lowest PNR was measured in samples derived from EP (0.06–0.31 μgNO₂⁻ N g⁻¹ dw h⁻¹), whereas a significant different between EP and IM was observed (p = 0.0327). Overall a significant decrease of PNR from intensively to extensively managed sites was found (p = 0.0350). The sampling time point did not influence PNR in the investigated samples (p = 0.4882).

In contrast DEA showed an interdependence between the sampling time point and the land use intensity (p = 0.0346; figure 1 and table 4). In spring the values for DEA in samples derived from IM and EP were up to the factor of three times higher (ranging from 0.31 to 2.39 μg N₂O N g⁻¹ dw h⁻¹) compared to summer (0.25–1.46 μg N₂O N g⁻¹ dw h⁻¹) indicating a higher denitrification activity in soil at the beginning of the vegetation period. In samples from IP the contrary phenomenon was observed. Here higher values were measured in summer (0.99 to 1.34 μg N₂O N g⁻¹ dw h⁻¹) than in spring (0.55 to 1.22 μg N₂O N g⁻¹ dw h⁻¹). Overall a significant positive influence of land use intensity was proven (p = 0.0009).

Abundances of Functional Genes

Gene copy numbers of bacterial amoA genes (AOB) increased significantly from the spring to the summer sampling at all intensively managed sites (IM and IP), ranging in April from 4.4×10⁶ to 5.5×10⁷ copies g⁻¹ and in July from 2.2×10⁶ to 2.1×10⁸ copies g⁻¹ (figure 2). Gene copy numbers in samples derived from the extensively used plots were in the range of 7.7×10⁶ and 8.7×10⁶ g⁻¹ and hence significantly lower compared to the intensively managed sites (table 4). In addition, no clear seasonal effect could be described for samples derived from EP sites, as on two plots (AEG7 and AEG8) a decrease from April to July was observed whereas on the third one an increase was found. Overall, AOB community size was influenced by season (p = 0.0293) and by land use intensity (p = 0.0004). Gene copy numbers for the archaean amoA gene (AOA) ranged from 3.0×10⁶ to 2.5×10⁷ g⁻¹ in April and from 8.5×10⁶ and 4.5×10⁸ copies g⁻¹ in July; no statistically significant seasonal effect were proven (p = 0.5872), but a significant increase from intensively to extensively used plots was observed (see table 4). As for AOB also for AOA an increase from April to July of amoA gene abundances on the intensively used plots and a decrease on the extensively used sites was visible determined. Considering the ratios of AOA to AOB except the site AEG6 (AOA:AOB ratio < 1) AOB ratios between 2.1 and 16 were found indicating a dominance of AOA over AOB. However in soil samples from the intensively used sites the ratio decreased significantly from April to July, whereas in samples from EP in two plots no changes were observed between the two sampling time points and a increase from 2.2 to 16 was found in the 3rd plot (AEG7).

Gene copy numbers for nirK ranged in April from 8.9×10⁶ to 3.7×10⁹ copies g⁻¹ and increased in July ranging from 5.0×10⁸ to 3.4×10⁹ copies g⁻¹ (p<0.0001; figure 2 and table 4). In April all plots of one land use category showed comparable abundance levels for nirK, whereas in July AEG3 (belonging to IM) revealed up to ten times lower copy numbers than AEG1.
### Table 3. Soil parameters and plant analysis.

<table>
<thead>
<tr>
<th>Site</th>
<th>WEOCb April/July</th>
<th>Total WENc April/July</th>
<th>Nitrate April/July</th>
<th>Ammonium April/July</th>
<th>extracted DNA April/July</th>
<th>Ellenberg indicator values for N</th>
<th>moisture (%)</th>
<th>Number and cover sum of vascular plants</th>
<th>Number and cover sum of legumes</th>
</tr>
</thead>
<tbody>
<tr>
<td>IMd1SDa</td>
<td>53.3/49.0</td>
<td>32.4/23.3</td>
<td>34.6/18.2</td>
<td>0.7/0.2</td>
<td>6.0 10^6/1.3 10^3</td>
<td>4.9</td>
<td>4.4</td>
<td>26/111</td>
<td>4/20</td>
</tr>
<tr>
<td></td>
<td>±3.8/±4.1</td>
<td>±8.4/±2.9</td>
<td>±12/±2.2</td>
<td>±0.2/±2.0 10^-2</td>
<td>±1.1 10^-6/1.3 10^-6</td>
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<tr>
<td>IMd2SDa</td>
<td>39.2/56.7</td>
<td>50.8/23.0</td>
<td>44.7/17.2</td>
<td>0.6/19 10^-2</td>
<td>4.7 10^5/1.5 10^5</td>
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<td>5.4</td>
<td>17/126</td>
<td>1/8.0</td>
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<tr>
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<td>±4.5/±7.7</td>
<td>±9.4/±3.9</td>
<td>±7.7/±3.1</td>
<td>±0.2/±9.0 10^-2</td>
<td>±1.2 10^-6/3.3 10^-6</td>
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<td>33.0/37.2</td>
<td>15.3/39.8</td>
<td>17.4/6.05</td>
<td>3.0/26 10^-2</td>
<td>3.9 10^6/1.1 10^6</td>
<td>5.7</td>
<td>5.4</td>
<td>28/171</td>
<td>3/31</td>
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<td>±5.7/±4.3</td>
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<td>±8.7/±1.2</td>
<td>±0.2/±7.0 10^-2</td>
<td>±5.7 10^-6/1.7 10^-6</td>
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<td>18.9/11.4</td>
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<td>1.8/1.10</td>
<td>1.6/28 10^-2</td>
<td>4.0 10^4/6.6 10^4</td>
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<tr>
<td>EPd3SDa</td>
<td>42.7/44.5</td>
<td>42.7/44.5</td>
<td>1.6/1.94</td>
<td>0.8/23 10^-2</td>
<td>3.5 10^9/7.8 10^-4</td>
<td>2.9</td>
<td>3.5</td>
<td>58/119</td>
<td>6/5.5</td>
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<td></td>
<td>±4.7/±6.1</td>
<td>±2.4/±1.0</td>
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<td>±6.6 10^5/7.6 10^-5</td>
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*standard deviation.
water-extractable organic carbon.
water-extractable nitrogen.
intensely used meadow.
intensely used mowed pasture.
intensely used pasture.
doi:10.1371/journal.pone.0073536.t003
and 2 and AEG8 (belonging to EP) up to five times higher numbers than the other both plots of this land use category. Overall nirK gene abundance increased significantly with increasing land use intensity. The occurrence of nirS genes was significant lower in spring than in summer (p = 0.0315). Copy numbers ranged from $2.3 \times 10^6$ and $1.4 \times 10^7$ copies g$^{-1}$ in
<table>
<thead>
<tr>
<th>Soil data</th>
<th>Ammonium</th>
<th>Nitrage</th>
<th>Total nitrogen</th>
<th>Organic carbon</th>
<th>Potencial nitrification</th>
<th>Potential denitrification</th>
<th>DNA</th>
<th>nirK</th>
<th>nirS</th>
<th>amoA</th>
<th>AOA</th>
<th>amoA AOB</th>
<th>nirH</th>
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<tr>
<td>Land use</td>
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<td>0.1246</td>
<td>0.1515</td>
<td>0.9706</td>
<td>0.0350*</td>
<td>0.0009*</td>
<td>0.0059</td>
<td>0.0048*</td>
<td>0.0498*</td>
<td>0.0064*</td>
<td>0.0004*</td>
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<tr>
<td>Time point</td>
<td>0.0100*</td>
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<td>0.0993</td>
<td>0.6837</td>
<td>0.4882</td>
<td>0.2668</td>
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<td>&lt;0.0001*</td>
<td>0.0315*</td>
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<td>0.0293*</td>
<td>0.656</td>
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Pairwise test

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<th>Soil data</th>
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<th>Total nitrogen</th>
<th>Organic carbon</th>
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<th>Potential denitrification</th>
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<th>amoA AOB</th>
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<td>n.s</td>
<td>n.s</td>
<td>n.s</td>
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<td>0.0120*</td>
<td>0.0078*</td>
<td>0.0501*</td>
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<td>n.s</td>
<td>n.s</td>
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<td>0.0530*</td>
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<td>n.s</td>
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<th>Plant data</th>
<th>Plant diversity</th>
<th>Plant coverage</th>
<th>Legume diversity</th>
<th>Legume coverage</th>
<th>Ellenberg nitrogen</th>
<th>Ellenberg moisture</th>
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<td>0.0400*</td>
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Pairwise test

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<tr>
<th>Plant data</th>
<th>Plant diversity</th>
<th>Plant coverage</th>
<th>Legume diversity</th>
<th>Legume coverage</th>
<th>Ellenberg nitrogen</th>
<th>Ellenberg moisture</th>
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*significant at a level <0.05.

doi:10.1371/journal.pone.0073536.t004
April and between 4.0 × 10^6 and 2.9 × 10^7 copies g dw−1 in July. Also land use had a significant influence on the number of nirS gene copies. However, nirS gene copy numbers were up to the factor of 500 lower compared to nirK.

Figure 2. Box plot analysis of gene copy numbers of different genes involved in the cycling of inorganic nitrogen cycle at the two different sampling time points (April and July) with different land use intensity. All data are log-transformed. doi:10.1371/journal.pone.0073536.g002
Gene copy numbers for \( \text{nifH} \) ranged from \( \text{1.8} \times \text{10}^6 \) to \( \text{6.9} \times \text{10}^7 \) copies \text{g dw}^{-1} \) (figure 2). The data did not reveal significant differences between the sampling time points \((p = 0.639) \) but between the land use intensities \((p = 0.006) \). IP sites show significant higher \( \text{nifH} \) copy numbers \(( \text{from } \text{1.9} \times \text{10}^6 \text{ to } \text{6.9} \times \text{10}^7 \text{ copies g dw}^{-1} \)) than the other two land use categories. The lowest values were detected on the EP plots, where quantities between \( \text{1.4} \times \text{10}^6 \text{ and } \text{9.0} \times \text{10}^6 \text{ copies g dw}^{-1} \) were measured.

**Integrative Data Analysis**

Based on all data obtained a between group analysis (BGA) was performed using the mean values of all five replicates from each site (Figure 3). The BGA revealed fertilization and other types of soil disturbance (e.g. soil compaction by machines or animals) are interconnected and explain the differences between the intensely (IM and IP) and the extensively (EP) managed grassland sites to a large extent [PCA1:43% PCA2:24%]. Based on the BGA data, it can be assumed that the effects of fertilization were more dominant compared to the influence of grazing and mowing. Moreover, clear differences between the sampling time points appeared (samples from July located in the lower part, from April in the upper part). Overall abundance of \( \text{nifH}, \text{amoA} \) AOA and AOB genes as well as potential nitrification activity were more related to land use intensity, whereas ammonium content and dissolved organic carbon as well as DNA content and abundance of \( \text{nirS} \) and \( \text{nirK} \) genes were affected more by time.

**Discussion**

**Plant Diversity**

[29] was among the first, who described a close link between land use intensity of grasslands and plant species composition. These observations have been confirmed in the present study. These results can be discussed on the basis of life strategies of plants and soil microorganisms: Whereas more plant species could be assigned to the exploitation group of so called c-strategists on the intensively managed sites, with high assimilation and growth rates if high nutrient levels were present in soil, the extensively managed plots were more characterized by “csr-plants” characterized by higher tolerance level to disturbance and nutrient stress but slower growth rates [23]. In addition to differences in species composition we found a significant reduction of plant species richness in response to land use intensification which has been also found in several other studies [30,31,32,33]. These losses were comparable to an over regional inventory by [34] who revealed that increased fertilization by \( 90-130 \text{ kg N ha}^{-1} \) is linked to a 50% higher productivity and a decrease of species richness by 30%.

**Land Use Intensity and Nitrification**

Several studies which investigated the impacts of different management regimes on the nitrifying potential in soil [35,6,36] described higher PNR in response to higher intensities of land use of grassland ecosystems. These results were confirmed in our study where all intensively managed meadows showed higher PNR values compared to the extensively managed plots. Interestingly despite an increase in \( \text{amad} \) gene copy numbers in samples from the IP and IM from spring to summer no comparable increase in PNR activities was observed. This could be the result of the fact that during summer the soil pH mainly in the rhizosphere decreased due to higher exudation rates of the plants, which might reduces the ratio of ammonia to ammonium in soil. As ammonia oxidizing microbes are only able to use ammonia for oxidation [37], this fact may explain that there is no direct link between PNR and \( \text{amad} \) gene copies when data from the spring and summer sampling were compared. AOA harbor urease genes allowing AOA to use urea as alternative substrate, when ammonia is limited in soil [30]. The observed increase of AOA between spring and summer might be related to that urea utilization and a mixotrophic lifestyle of AOA. The relatively low PNR in summer (in relation to the abundance of AOA and AOB) might be also a fact of nitrification inhibitors which are excreted by many plants into soil to increase their competitiveness towards ammonia against microbes [39]. This ability of plants to reduce activity of AOA and AOB becomes even more important in those sites which have been extensively managed and nitrogen contents are very low. Obviously plants on these plots were indeed very competitive for ammonia uptake as not only PNR rates did not change between spring and summer but also the abundance of AOA and AOB did not change.

In nearly all samples a higher number of \( \text{amad AOA} \) than \( \text{amad AOB} \) copies \((\text{AOA}/\text{AOB} \text{ ratios were between 2 and 16}) \) was found. These results are in accordance with some other recent studies indicating dominance from AOA over AOB in soils [40,41,14]. Up to now it is still unclear if these higher abundance values of AOA might be related to a significant contribution to ammonification in soils [42,43,44,45,46]. However considering the correlation of different \( \text{amad} \) genes and the nitrification potential \((\text{R}_{\text{AOA}} = 0.836, \text{P}_{\text{AOA}} < 0.001 \) and \( \text{R}_{\text{AOB}} = 0.680, \text{P}_{\text{AOB}} < 0.001 \) our dataset delivers one more hint that AOA plays an important role at least for ammonia oxidation in soil, although based on molecular data still the oxidation of hydroxylamine to nitrite has not been proven for AOA. Overall it seems that AOA and AOB are similarly affected by land use intensification and it has been crystallized that nitrogen availability is the major driver for performance and abundance of the nitrifying community.

**Land Use Intensity and Denitrification**

Microbial communities in soil are generally stimulated by plant growth especially in the rhizosphere by root exudates, resulting in an increase of anoxic habitats in the rhizosphere and the need for microbes to use alternative electron acceptors like nitrate inducing denitrification [47,48]. This was confirmed in our study by reduced DEA activities in soil samples derived from EP, where plant biomass was significantly lower. However in our study DEA was not stimulated in summer compared to spring, which is surprising on the first glance but might be a result of a low nitrate concentrations found mainly in soils from intensively managed plots IP and IM. Furthermore the differences in water content and the corresponding differences in the redox potential between spring and summer sampling might have influenced DEA activity independent from land use intensities.

To describe denitrifying communities, the two nitrite reductase genes \((\text{nirK} \) and \( \text{nirS} \)) were quantified. Overall, many prokaryotes from soil are able to denitrify and the proportion of denitrifiers within the soil microbial community was considered between 10% and 60% of the total bacterial and archaeal microbiota reaching values between \( \text{10}^5 \) and \( \text{10}^9 \text{ copies g}^{-1} \) [27,49,50,51,52,53]. The values found in our study were in a similar range. In all samples gene copy numbers for \( \text{nirK} \) genes dominated over \( \text{nirS} \) genes \((\text{nirS}/\text{nirK} \text{ ratio } 0.002-0.02) \). This is in accordance with data from other studies, which indicate \( \text{nirK} \) phyotypes being more dominant in the rhizosphere or in soils characterized by an intensive root development [54], or high nutrient contents due to climatic conditions (e.g. freezing and thawing, [53]). The relatively high WEOC (water extractable organic carbon) values confirmed this hypothesis (Table 3). A strong time effect \((p < 0.0001) \) was found for both types of nitrite reductase genes in all soil samples, with values up to 20 times higher in summer, which correlated to the increased overall microbial biomass values obtained in soil samples.
Figure 3. Between group analysis (BGA) using the mean values of all five replicates from each site. For calculation all data were logarithm-transformed, scaled and calculated using the R software package (www.r-project.org). The small letters a and j sign the April and July samples. Clear differentiation according to the sampling time point is visible (all April samples are on the right site in the upper part, all July on the left site in the lower part). Also the intensely used sites are clear separated from the extensively used plots whereas the separation is more clear in July. doi:10.1371/journal.pone.0073536.g003
from summer compared to spring (data not shown). The increase in abundance of the nitrite reductase over time is in contrast to the potential measurements of denitrification, but could be easily explained as all denitrifiers are facultative anaerobes and can shift their metabolism from respiration to denitrification based on the conditions present in soil. Therefore not surprisingly this missing link between denitrifying activity and genetic potential was described in some studies before [56,57]. However several studies have indicated that the plant species composition can act as a major driver for denitrifiers in soil [58] probably by providing different amounts and compositions of root exudates [59,60] mainly in summer which may explain the significant copy number increase of both genes in our study. Overall it can be concluded that abundance of denitrifiers is mainly driven by the amount and quality of carbon present in soil and consequently increase or decrease of denitrifiers is strongly linked to development of overall microbial biomass, whereas the induction of genes involved in denitrification is linked to the presence of nitrate and the redox conditions present in soil.

Land Use Intensity and Nitrogen Fixation

Surprisingly the highest amount of nifH copies was found at IP sites and not as we assumed at sites, which have been used extensively (EP). This was nitrogen limited. Despite no differences were found in the WEOC, this might be related to the fact that in soil the amount of energy equivalents needed for nitrogen fixation is too low in soil samples derived from EP [61], confirming earlier data obtained by [62] from glacier fore fields, where also low abundance of nifH as well as low potential activity for nitrogen fixation was found at the youngest sites, which had the lowest supply with nitrogen and lacked of energy equivalents. [63] as well as [64] showed in their study that intensive grazing has positive influences on nitrogen fixing communities in grassland soil, due to the high amounts of excrements which provide good sources for energy generation.

However it must be stated that the occurrence of legumes was not correlated to the nifH gene abundance, so we assume that the obtained results just reflect the occurrence of free living nitrogen fixing microbes, as we did not include nodule analysis in our study.

Conclusion

Obviously there is a strong link between land use intensity and microbial community structure in soil. Some of these effects might be direct effects and a response to e.g. increased fertilization regimes, other might be indirect effects and mediated by changes in the plant diversity and biomass (e.g. through changes in exudation). However the described data just reflect potentials for certain processes. If the same response pattern can be confirmed for the activation of certain functional traits remains an open question for future research focusing more on mRNA based studies with a more dense net of sampling time points.

Supporting Information

Table S1 List of plant species with species categorization according to ecological strategy type (Grime 1997) and Ellenberg values for moisture (M) and nutrients (N) (Ellenberg et. al. 2001).

Different ecological strategy types as given in the table are competitors (c), competitive ruderals (cr), stress-tolerant competitors (cs), stress-tolerant ruderals (sr), ccr-plants (competition is restricted by the combined effects of stress and disturbance), stress-tolerants (s) and ruderals (r). Ellenberg indicator values are normalized values on an ordinal scale from low to high values (1 to 9) whereas 1 means high tand 9 low tolerance to nutrient- and moisture stress. NA shows plants which are not categorised by Ellenberg and x signs these with a large ecoclor amplitude. Occurrence of each plant species on the different plots is given in percent of coverage.

(DOC)

Acknowledgments

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Author Contributions

Conceived and designed the experiments: MS EK IS AF VR. Performed the experiments: AM IS SB DK MS. Analyzed the data: AM AF GW. Contributed reagents/materials/analysis tools: AM VR DK IS SB. Wrote the paper: AM SM EK MS.

References


