The phosphorus dilemma in organically farmed grasslands – are legume presence and phytodiversity incompatible?

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ABSTRACT

Organic grassland farmers are caught between the pressure to operate profitably and the need to ensure ecosystem services like nature conservation. Phosphorus (P) management is at the core of this dilemma: grassland legumes, which are vital for nitrogen (N) inputs, forage quantity and quality, have high P requirements. However, rising soil P contents lead to decreased phytodiversity. We examined this ‘P dilemma’ based on data of a study in typical organically managed grasslands in north-eastern Germany. We found a slight, but not significant negative correlation between the soil P contents and phytodiversity. Such a negative correlation was much clearer in broader ecological studies of European grasslands, probably due to a scale effect. Despite comparably small P concentrations, we detected only moderate species richness. Thus, other factors besides P concentrations are limiting phytodiversity here, probably N concentrations. The P concentrations in the biomass of various legume species indicated P limitations for most of them. We conclude that a bundle of adapted management measures could reconcile economic and environmental requirements of organic farmers to a certain degree and thus lessen the conflict between the two ecosystem services ‘biodiversity’ and ‘forage production’.

Introduction

Grassland is of great importance for the agricultural sector in Europe as grasslands and rangelands cover one-third of the European agricultural area (Peeters 2009). Grassland provides a wide range of ecosystem services including marketed services (forage production for livestock) and non-marketed services (e.g. biodiversity conservation, climate regulation, recreation) (Hopkins and Holz 2006; Lamarque et al. 2011). Forage production is about quantity as well as quality. The livestock sector is dependent on grassland and strongly influences the appearance and quality of grassland (Hopkins and Holz 2006). Due to a continuing strong expansion of grazing land and intensification of grassland management over the past decades, the livestock sector is responsible for land degradation and habitat loss and associated loss of biodiversity worldwide. On the other hand, pastures in Europe, but also elsewhere, have developed into locations of various ecosystem types, many of which are now threatened by abandonment (Isselstein et al. 2005; Hopkins and Holz 2006). According to Mace et al. (2012), biodiversity plays a decisive role at all levels of ecosystem services hierarchy by acting (1) as regulator of ecosystem processes, (2) as a final ecosystem service itself or (3) as a good. That means, that (1) biodiversity controls ecosystem processes (e.g. nutrient cycling or biomass production), which in turn form the basis of ecosystem services; (2) biodiversity (e.g. genetic or species diversity) is the specific goal of ecosystem management for providing goods; or (3) biodiversity is a good in itself with a definite value for humans (Mace et al. 2012).

The Common International Classification of Ecosystem Services (Haines-Young and Potschin 2018) has three main categories of ecosystem outputs or ‘final ecosystem services’: provisioning, regulating and cultural services which grassland can provide all. Following this classification, grassland provides biomass by harvesting wild plants for ruminants (provisioning); it regulates physical, chemical and biological conditions by regulating basement flows and soil quality as well as protecting habitats and gene pools (regulating); and grassland offers outdoor interactions with the natural environment (cultural service). Besides biodiversity, provisioning forage for ruminants is the final ecosystem service, which is the focus of this study. According to the cascade model by Potschin and Haines-Young (2011), biodiversity can be a final ecosystem service as well as a supporting or intermediate service if understood in its broadest sense, namely as basis of all services that depend on living systems and as source of many processes. Generally, biodiversity can be defined and understood in different ways: biodiversity within species, between species and of ecosystems, as well as genetic...
diversity, taxonomic diversity, functional diversity, diversity of fauna and/or flora. Ecosystem functions and thus ecosystem services of grassland depend especially on functional diversity (Lamarque et al. 2011). The nitrogen (N)-fixing legumes represent one important functional group in grassland ecosystems and therefore an important part of the biodiversity in these ecosystems.

Loss of biodiversity could decrease ecosystem function (Loreau et al. 2001; Lamarque et al. 2011), to the contrary, a higher biodiversity could increase ecosystem functions (Mace et al. 2012). The intensity and type of grassland management mainly influences the quality and nature value of grassland and thus the scope of its ecosystem services. Therefore, fertilization could have a great impact on the ecosystem services delivered by grassland.

Phosphorus (P) is an essential plant nutrient, which is growth-limiting in many habitats including agricultural ones (Sattari et al. 2016). Therefore, mineral and organic fertilizers containing P have been increasingly applied to soil since the middle of the twentieth century (Haygarth et al. 2014). This agricultural intensification in the last decades with increasing partial imbalances between P input and output has already resulted in excessive P accumulation in many soils (De Bolle et al. 2013). This trend may be amplified as the world population continues to grow. In general, the global demand for food, including ruminant-based products, is expected to double by 2050 (Koning et al. 2008). Thus, requirements for both fodder quality and quantity will also increase strongly (Koning et al. 2008). Sattari et al. (2016) calculated that P inputs in existing grasslands will have to increase more than fourfold by 2050 in relation to 2005 to realize the predicted 80% larger required forage production, assuming a constant total grassland area. They concluded that the use of mineral P fertilizer would have to amount to 24 Tg per year, an amount comparable to the overall 23 Tg mineral P fertilizer applied to global grasslands from 1970 to 2005.

Independently of the P retention capability of the soils, P surpluses increase the risk of P losses via erosion, subsurface runoff, leaching and drainage (Amery and Schoumans 2014) and, thus, the risk of surface water eutrophication that is mainly caused by P from agriculture (Schoumans et al. 2013). Furthermore, elevated soil P contents negatively affect phytodiversity (Janssens et al. 1998; Wassen et al. 2005; Ceulemans et al. 2014) and species composition in grasslands (Wassen et al. 2005; Hejcman et al. 2012), partly as a result of increased productivity and resulting competition for light (Wassen et al. 2005; Hautier et al. 2009). According to Ceulemans et al. (2014), the existing P fertilizer recommendations for grasslands often already exceed threshold soil P concentrations for phytodiversity. Even with reduced P fertilization, they did not expect highly diverse swards; hence, they concluded that P fertilization should cease in grasslands.

However, especially grassland legumes with their numerous known agronomic benefits for forage production and quality as well as for the environment (Graham and Vance 2000; Hopkins and Holz 2006; Phelan et al. 2014) have high P requirements. P fertilization can improve their vitality, increase their N2-fixing capacity and their proportion in mixed swards as well as their yields (Davis 1991; Højh-Jensen et al. 2002; Divito and Saldras 2014). This is especially crucial for organically managed grassland farming, which depends on the N-input by N2-fixing legumes (Römer and Lehne 2004). Thus, in contrast to the above suggestions for phytodiversity, P fertilization is agronomically recommended for legume vitality and N2-fixation, especially in organic grassland farming.

Consumers generally expect that organically produced foods also guarantee sustainability of the production system and protection of ecological resources (Fess and Benedito 2018). Simultaneously, organic farming is regarded as a general tool for maintaining biodiversity at the farm and landscape scale (Van Elen 2000). For instance, an Action Plan uniting organic agriculture with nature conservation movements was announced (Stolton et al. 2000). The plan recognizes the role of organic agriculture in conserving biodiversity and suggests a number of policy options for achieving high biodiversity at farm level. However, since organic products have left the regional direct market niche and conquer the shelves of discount stores throughout Europe (Hill and Lynchenaun 2002), the price pressure on grasslands’ main products milk and meat is growing also in the organic food segment. To withstand this growing price pressure and to maintain their market shares, organic grassland farmers have to produce superior product qualities. In the case of pasture-based meat production, the most widespread use of semi-natural grassland (Kruse et al. 2016), as well as dairy production, this aim is hard to achieve without diets rich in energy and protein. Therefore, a further increase of legume proportions as a provider of protein is indispensable for economic sustainability of modern organic grassland farming.

These trends in mind, organic grassland farmers are caught between the economic pressure to enhance forage value and the society’s high expectation to simultaneously maintain phytodiversity of threatened grassland sites. The question of P management stands in the middle of this heated debate since soil P content influences both sides: it has the potential to enhance grassland legumes and thus N yields but it can also be responsible for a marked decline in phytodiversity.
Therefore, we conducted a survey examining grassland phytodiversity as one important part of biodiversity and legume cover as functional diversity, as well as soil and plant P contents in north-eastern Germany. These sites were farmed organically and simultaneously designated as areas where the federate state wants to attain high biodiversity due to the high natural value (HNV) potential of these sites. Additionally, we decided to survey these sites as they have not been fertilized with P so far and could show the current P potential prior to intensification.

The aim of our case study was to investigate the status quo concerning the described ‘P dilemma’. The main questions of this study are:

1. Is the reported generally negative correlation between soil P content and phytodiversity found in broad-scaled ecological studies also valid for typical site conditions of organically farmed grasslands at a regional scale?
2. Are the grassland legumes sufficiently supplied with P?
3. Can we determine a conflict between the ecosystem functions forage production and biodiversity from our results and confirm the P dilemma?
4. Which P management options remain to reconcile economic sustainability of organic grassland farming with ambitious environmental goals regarding phytodiversity?

Methods

Our case study was conducted on five representative mown pasture sites under organic management in north-eastern Germany (Figure 1). The area is affected by a thermoclimatic gradient from north-west to south-east, which means the study sites are located within a transition area from the maritime to the submaritime climate zone (IfL 2007–2013). It is therefore particularly interesting with regard to climate change. Within this area, we have chosen our study sites because, on the one hand, they are farmed organically and organic farming per se is considered to be a protector of biodiversity and good manager of resources. On the other hand, we have selected the sites because the federal state designated them as areas with a high biodiversity as development target, due to their HNV potential. So the study sites can be seen as examples of sustainable use, where both agricultural and nature conservation objectives are pursued.

Figure 1. Map of surveyed grassland sites under organic management in north-eastern (NE) Germany. site 1: mineral soil; site 2: transition sequence from mineral to fen-peat soils; site 3: fen-peat soil; site 4: fen-peat soil; site 5: strongly degraded fen-peat soil.
The selected mown pastures were associated in the phytosociological community Molinio-Arrhenatheretum (Schubert et al. 2001). They are located in the Weichselian glaciation moraine landscape with its characteristic variety of soil types that also change in small spatial scales. Our study sites were defined by different soil parent materials: one site was on sandy mineral soil, one site covered a transition sequence from mineral to fen-peat soils and three sites were on fen-peat soils, including a strongly degraded one (Figure 1). The site on the mineral-to-organic soil transition sequence was divided into three zones for all investigations: one on mineral soil, one in the transition zone and one on fen-peat soil. As usual for such sites in this area, none of them was additionally fertilized. Therefore, we assumed a P-poorish of the sites, which in turn speaks for a high potential of phytodiversity, but at the same time it could mean a P deficiency for legumes. Additionally, it could show the current P potential prior to intensification.

In summer 2014 and summer 2015, we investigated 48 plots of 2 m × 2 m in total. Plots were chosen randomly at each site, following a diagonal transect across each site, and with legume presence as a precondition. The total number of plots per site was chosen in relation to pasture size and heterogeneity. The distance between adjacent plots was 5 and 50 m. On all sites, legumes were patchily distributed, covering small proportions (<15%). The vascular plant species number at each plot was recorded as alpha-diversity according to Whittaker (1972), hereafter also called species richness or phytodiversity (as part of biodiversity).

According to Liebisch (2011), we combined plant and soil analyses to assess the P nutrition status of our grassland sites. Soil samples were collected with an auger (20 mm diameter under either legumes or accompanying vegetation) to a depth of 10 cm, air-dried, sieved <2 mm and milled. P of all soil samples was extracted by double lactate (PDL) (Riemh 1948) and determined by ICP-OES (Optima 8300, Perkin Elmer, Waltham MA, USA). For comparison with data of Ceulemans et al. (2014), we additionally determined $P_{Olsen}$ (0.5 M NaHCO$_3$) of the soil samples of the accompanying vegetation according to Olsen et al. (1954). Soil pH of all soil samples was measured in a soil–water-suspension (1:5) using a glass electrode. Plant material of both accompanying grasses and dominating legume species was harvested separately in each plot. The herbs were neglected following Liebisch (2011) who recommended analysing only the grass fraction for evaluating the P status of agricultural grassland sites. The N concentration of the dried and milled plant material was determined using ICP-OES (Optima 8300, Perkin Elmer, Waltham MA, USA). For analysing the P concentration of the same material, a reaction with HNO$_3$ and H$_2$O$_2$ as well as microwave digestion had been conducted beforehand.

In order to find management options for farmers to escape the ‘P dilemma’ we did a target-oriented literature review in addition to our study.

Statistical analyses and figures were prepared with R (Version 3.3.1, R Core Development Team 2016). All variables were tested for normal distribution using the Shapiro–Wilk-test and converted to normal distribution by logarithmic transformation if possible.

Significant differences among the nutrient concentrations of different legume species were detected by a post hoc test (Tukey HSD) following either an analysis of variance (ANOVA) or the Kruskal–Wallis-test, depending on the results of the Shapiro–Wilk-test on normal distribution of the data. We used N:P ratios of plant biomass as indicators of N or P limitation. For analysing the relation between number of species per plot and N:P ratio of the above plant tissue of the accompanying vegetation, simple linear models including the different parent materials were calculated. For this, we neglected the N:P ratio of the legumes as they are inherently high in N and have significantly larger P concentrations than grasses when P is limiting (Liebisch et al. 2013). The relationship between the mean values of $P_{Olsen}$ contents and the mean values of species richness of the five sites was determined by a Pearson’s correlation. We additionally used an ANOVA followed by the F-test to evaluate the effect of $P_{Olsen}$ soil contents and parent material on phytodiversity with species number as response variable and the interaction between $P_{Olsen}$ and parent material as factor. As the surveyed plots within the five mown pastures could be considered as pseudo-replicates, the ANOVA model considered the nested effect of $P_{Olsen}$ soil content at the site scale by adding an additional error term (Equation (1)). The significance level of all tests was set at $p < 0.05$.

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\text{Model} : \ y_{ijk} = \mu + d_i + f_j + k_{(ij)} + e_{(ijk)} \quad (1)
$$

- $\mu =$ the general mean value of alpha-diversity
- $d_i =$ the effect of the $i$ soil $P_{Olsen}$ content (logarithmized) as covariate
- $f_j =$ the effect of the $j$ parent material
- $k_{(ij)} =$ the effect of the $k$ site nested in $j$ parent material
- $e_{(ijk)} =$ residual error

**Results**

In total, 121 vascular plant species were recorded at the five organically managed grassland sites, 25 of which were grasses, 12 sedges/rushes, 9 legume species and 75 herbs. We detected *Trifolium repens*, *Trifolium pratense*, *Lotus corniculatus*, *Lotus pedunculatus* and *Lathyrus pratensis* as dominating legume
Table 1. Dominating legume species, other legume species and most frequent grasses, herbs and sedges with their frequency and coverage at the organically managed grassland sites of the case study.

| Dominating legume species | Frequency | Coverage (%) |
|---------------------------|-----------|--------------|
| Trifolium repens          | 14/48     | 4–28         |
| Trifolium pratense        | 12/48     | 4–13         |
| Lotus pedunculatus        | 9/48      | 3–13         |
| Lathyrus pratensis        | 9/48      | 3–20         |
| Lotus corniculatus        | 4/48      | 5–15         |

Other legume species

| Lathyrus palustris         | 4/48      | 0.3–2        |
| Vicia cracca              | 4/48      | 0.3          |
| Trifolium dubium          | 3/48      | 0.3          |
| Trifolium medium          | 1/48      | 0.3          |

Most frequent grasses on mineral soils

| Lolium perenne            | 15/15     | 3–79         |
| Dactylis glomerata        | 9/15      | 3–34         |
| Poa pratensis             | 9/15      | 0.3–4        |
| Festuca rubra             | 8/15      | 0.3–13       |

Most frequent herbs on mineral soils

| Taraxacum officinale      | 15/15     | 3–27         |
| Achillea millefolium      | 13/15     | 0.6–17       |
| Plantago lanceolata       | 12/15     | 1–18         |

Most frequent grasses on fen soils

| Poa pratensis             | 18/27     | 0.3–62       |
| Holcus lanatus            | 14/27     | 1–37         |
| Festuca pratensis         | 8/27      | 0.3–20       |
| Alopecurus pratensis      | 8/27      | 0.3–11       |

Most frequent herbs on fen soils

| Filipendula ulmaria       | 19/27     | 0.3–22       |
| Ranunculus acris          | 19/27     | 0.3–10       |
| Plantago lanceolata       | 9/27      | 3–10         |
| Taraxacum officinale      | 9/27      | 1–14         |

Most frequent sedges/rushes on fen soils

| Carex disticha            | 11/27     | 4–70         |
| Juncus articulatus        | 6/27      | 0.3–2        |
| Carex nigra               | 6/27      | 5–30         |

Table 2. Characterized of examined parameters of interest at all sites. pH (H2O), PDL, Olsen and species number (4 m⁻²) given as mean ± SD.

| Parent material | Diverse | Mineral | Mineral | Transition | Fen | Site 3 | Site 4 | Site 5 |
|-----------------|---------|---------|---------|------------|-----|--------|--------|--------|
| pH              | 7.0 ± 1.0 | 7.0 ± 0.8 | 4.5 ± 0.1 | 5.3 ± 0.6 | 5.9 ± 0.2 | 5.1 ± 0.3 | 6.6 ± 0.2 | 5.9 ± 0.7 |
| PDL (mg kg⁻¹)   | 32.72 ± 23.02 | 22.80 ± 18.37 | 37.03 ± 17.62 | 16.17 ± 4.29 | 24.72 ± 6.31 | 26.70 ± 7.21 | 43.25 ± 9.15 | 70.82 ± 44.38 |
| Olsen (mg kg⁻¹) | 24.52 ± 14.71 | 21.75 ± 5.85 | 41.00 ± 21.63 | 9.33 ± 4.80 | 26.00 ± 7.81 | 20.45 ± 14.64 | 24.88 ± 3.04 | 47.00 ± 18.76 |
| species number n| 15.9 ± 3.4 | 13.3 ± 2.4 | 14.3 ± 2.1 | 20.3 ± 1.1 | 19.3 ± 4.0 | 15.3 ± 3.3 | 17.8 ± 2.5 | 14.4 ± 1.9 |
| n               | 48       | 12       | 3        | 6          | 3   | 11     | 8      | 5      |

Discussion

Soil P and phytodiversity

Organic farmers use their fertilizers preferentially on arable land as this is regarded as more valuable than grassland. This behaviour probably causes the distinctly smaller range of soil P values than found in broader ecological studies (Janssens et al. 1998; Merunková and Chytrý 2012; Ceulemans et al. 2014). The soil P contents (Table 2) were comparable to those in other organically managed grassland sites in northern Germany (Wachendorf and Taube 2001; Ohm et al. 2015). In contrast, Ceulemans et al. (2014) surveyed grasslands of a wide nutrient gradient across Europe, from nutrient-poor dunes to grasslands highly enriched with nutrients species within the 48 plots. Besides these dominating legume species, we also found some other legumes, but only in small proportions (Table 1).

The soil Olsen contents (Table 2) were mostly smaller than 50 mg P kg⁻¹ and without large differences among sites, except for the strongly degraded fen-peat that had up to 69 mg P kg⁻¹ in one plot. The P content was smallest in the transition zone of site 2, where the parent material was between mineral and fen-peat soil. Soil pH values ranged from 4.4 to 7.8 (Table 2). The mineral and strongly degraded fen-peat soils showed on average smaller species numbers per plot than those on fen-peat soils or in the transition zone of site 2 (Table 2). The smallest number of species was 10 at mineral site 1 and the largest was 23 in the organic zone of site 2.

There was a negative, but not significant (p = 0.19) correlation between species richness and the mean Olsen values of all sites. The ANOVA showed an effect of the logarithmically transformed Olsen concentrations on species richness with a significance level of 0.02. There was further a slight effect of the parent material on species number with a significance level of 0.09. However, there was no significant interaction between log-transformed Olsen and parent material.

Lathyrus pratensis and Lotus pedunculatus had large P concentrations in the aboveground plant biomass although the corresponding soil P concentrations were relatively low. Trifolium repens showed relatively constant P plant concentrations over a wide range of P contents in soil. For Lotus corniculatus, we also determined relatively constant P plant concentrations, albeit over a small range of soil P concentrations (Figure 2).

The plant P concentrations of T. repens and T. pratensis differed significantly, while those of the other three legume species were intermediate. For the soil PDL contents under different legumes, no significant differences could be ascertained (Table 3).

There was a positive linear trend between the number of vascular plant species and the N:P ratio of the plant tissue of the grasses accompanying the legumes. With increasing N:P ratio of the plants, the number of plant species was also increasing (Figure 3). The linear model was best when containing all soils of all sites and could explain about 28% of the variance at p < 0.001. We calculated separate models for mineral and organic soils, but they were not as good as the model for all sites together.
due to their location (bordering arable land, polluted water bodies, redistribution by livestock), but without direct fertilization. This wider gradient probably explains the significantly negative relationship between species richness and soil P content by Ceulemans et al. (2014), missing in our data. Nevertheless, we were able to detect this negative relationship between soil P content and phytodiversity, which is reported in broad-scaled studies like that of Ceulemans et al. (2014), on our organically farmed grassland sites as well, although not so pronounced. With our plots on the studied pastures we have covered sandy mineral soils and also degraded to intact organic soils. Other soils of this geographical region, such as marsh soils and loess soils, are missing from our study. These could possibly reinforce the negative relationship between soil P and species richness.

Figure 2. P concentrations in aboveground plant biomass of different legume species and corresponding soil P concentrations (extraction by double-lactate) of five organically managed grassland sites in NE Germany.

| Plant P (g kg⁻¹ DM) | Soil P (mg kg⁻¹ atro) |
|----------------------|------------------------|
| Lathyrus pratensis   | 2.66 ± 0.97 ab         | 32.64 ± 19.69 a       |
| Lotus corniculatus   | 1.79 ± 0.12 ab         | 31.00 ± 26.32 a       |
| Lotus pedunculatus   | 2.30 ± 1.44 ab         | 27.62 ± 8.80 a        |
| Trifolium pratense   | 2.01 ± 0.59 a          | 21.57 ± 10.97 a       |
| Trifolium repens     | 2.94 ± 0.36 b          | 38.06 ± 34.02 a       |

Gilbert et al. (2009) suggested that Pₐₜₒₙₛ values below 10 mg kg⁻¹ would be best for restoring a high floristic diversity in mesotrophic grasslands. However, our plots with Pₐₜₒₙₛ values below 10 mg kg⁻¹ still had a comparatively small number of species (Figure 4). This was probably due to different sampling strategies, as Ceulemans et al. (2014) chose locations containing as many characteristic plant species of the surveyed site as possible. At our sites, this would correspond to values between the mean species number per plot and the total number of vascular plant species per site. Our average species number was also smaller than reported by Wachendorf and Taube (2001) for organically managed grasslands in north-western Germany. They investigated 102 representative plots with on average 23.4 species per 40 m². On mown pastures, they found on average 21.1 species in comparison to the 15.9 species we found. This difference is probably related to the different plot sizes, as the probability of finding more species increases with larger plot sizes (Meier and Hofer 2016). According to a general function of the area-abundance relationship (Preston 1962), we should have found on average 28.5 species using a plot size of 40 m².
Soil P and legumes

For legume growth, the P concentrations we found were mostly not optimal. *Trifolium repens*, the commonly known and used clover species in permanent grasslands of the temperate zone (Käding and Werner 2003) and our most dominating legume species, was found to require P application rates of 200–250 mg P kg\(^{-1}\) soil to attain 97% of maximum yields in pot experiments in New Zealand (Moir et al. 2003).
2.7 and 3.0 g kg$^{-1}$ Olsen in P could have sufficient P supply. P fertilization might further decrease species richness at the end of the study (Maxwell et al. 2013). Davis (1991) calculated critical P concentrations in grassland legumes based on the optimal P concentrations for 90% of their maximum yields in a field study in New Zealand. For $T. repens$, he stated 3.1 (in May) and 3.4 g kg$^{-1}$ (in December) for critical concentrations; for $T. pratense$ 2.2 and 2.8 g kg$^{-1}$, for $L. pedunculatus$ 2.7 and 3.0 g kg$^{-1}$ and for $L. corniculatus$ 2.1 and 2.3 g kg$^{-1}$. According to these numbers, the legumes of our study were not sufficiently supplied with P on most sites.

To achieve high quality milk and meat production, organic farmers depend on forage that is rich in energy and protein. Thus, as organic farming rests on N- and protein-providing legumes, our data show that P fertilization should be agronomically recommended at our surveyed sites. Appropriate P fertilization can enhance legume yields and proportions and thus also N$_2$-fixation, causing positive effects for the total production of the swards, of course in interaction with other site conditions.

Additionally, with $P_{DL}$ concentrations mainly between 20 and 40 mg kg$^{-1}$, the soils of our study sites were mainly associated in Germany’s soil class B (low). Following the recommendations, class C (optimal) should be achieved, so that these grassland sites (except the degraded fen site) would still require P fertilization. Bélanger et al. (2017) pointed out that an adequate P supply especially in the early growing season is important for optimum yields of grassland swards. However, Cooper et al. (2018) reported that the majority of organically managed grasslands in Europe had sufficient soil extractable P. Interpreting data of thousands of grasslands at the field scale, the authors considered not only extensive grassland sites, as we do, but also more intensive and fertilized ones. This again stresses the necessity of organic farmers to push productivity, comprising phytodiversity. Additionally, Cooper et al. (2018) remarked that conventional soil extractants ($P_{DL}$, $P_{Olsen}$, $P_{CAL}$, $P_{AL}$, Bray) may underestimate P supply in organic farming systems as they do not assess processes mediated by microbes, soil fauna and flora. These are probably more important in organically than in conventionally managed soils. This means that even soils categorized as 'low' in P could have sufficient P supply.

**N:P ratio and phytodiversity**

Greater plant available soil P concentrations following P fertilization might further decrease species richness at our surveyed sites, depending on the extent of the input (Janssens et al. 1998; Ceulemans et al. 2014). This was already indicated by the negative trend between species number and soil P found in our study. In addition, the specificity of organically farmed grasslands, the N-limitation, might influence this relation. In conventional N-rich grasslands, phosphate usually is the limiting nutrient for plant growth (Fay et al. 2015). There, additionally added P drives biomass growth in the presence of surplus N, resulting in enhanced competition. This in turn supports the dominance of a few competitive nitrophilous species resulting in a decline of phytodiversity (Güsewell 2004). Various studies have shown that shifts in limitation generally lead to changes in vegetation composition, phytodiversity and plant traits (Güsewell 2004). We detected a slight linear trend between the number of plant species and the N:P ratio: with increasing N:P ratios, the species number also rose.

Güsewell (2004) outlined that N:P ratios up to about 10 indicated N-limitation in wetlands and dry grasslands. N:P ratios between 10 and 20 described N- or P- or co-limitation of these vegetation types. The N:P ratios in the biomass of our study were mostly between 5 and 15, showing that the majority of the plots were N-limited. Thus, P fertilization and the following N-input by legumes would probably lead to clear changes in vegetation composition. Liebisch et al. (2013) proposed an optimal N:P range from 5.5 to 9 (analysed in the grass fraction) for a sufficient P nutrition status of the grasslands. They deduced this range by analysing data of mineral agricultural grassland sites managed at different intensities in Switzerland. Our data showed a clear difference between the N:P ratios of the mineral and the organic sites, which is in line with a slight, but not significant effect of the parent material on species number (ANOVA, $p = 0.09$). The mineral plots of our study were within the optimal range of Liebisch et al. (2013), so that P fertilization should not be necessary. The N:P ratios of our organic plots were mainly above this range. This was probably due to the release of N as a result of mineralization of the organic soils. Here, an appropriate P fertilization could be recommended especially during the early growth stage in spring (Bélanger et al. 2017).

**Compatibility of measures to promote legumes with other ecosystem services**

Regarding the P management of their grasslands, the study confirmed that organic farmers are confronted with a conflict between optimizing production and sustaining phytodiversity. The study offers results of a survey on limited sites, but the situation is typical for organic farms not only in central Europe, as P fertilization has so far been widely neglected. Legumes, which play an important role in the qualitative and quantitative improvement of forage (Phelan et al. 2014), have high P requirements, so that P fertilization can increase their N-fixating capacity. The
increased N-input in sward in turn has a positive effect on biomass production. P itself also promotes biomass production, while it has a negative effect on phytodiversity. Increased biomass production as a result of N-input can also lead to reduced phytodiversity. In conclusion, the ecosystem service 'forage production', is hardly incompatible with the ecosystem service 'biodiversity' (Fort et al. 2015). Hopkins and Holz (2006) also noted that agronomic management for quantitatively and/or qualitatively improved herbage production is not in line with management for biodiversity.

There is no single solution to reconcile economically sustainable organic grassland farming with the preservation and promotion of grassland phytodiversity, but there are options in different areas to escape from the described dilemma: agronomy, breeding and site-use strategy. The first option is the choice of an appropriate legume species that is adapted to less P supply than *T. repens*, which has comparatively large P requirements (Hart and Jessop 1984). In our study, in plots with large soil P concentrations, we only found *T. repens* and sometimes *L. pratensis*. In plots with moderate or small soil P concentrations, we detected more legume species. Although legumes in general have large P requirements, the needed amounts differ among legume species with various strategies to adapt to sub-optimal growth conditions (Fort et al. 2015).

With greater biomass production of *L. pedunculatus* under low soil P conditions in both pot and pasture field experiments in New Zealand, its potential as an alternative to *Trifolium* spp. has been shown, especially for acidic soils (Lowther 1991; Moir et al. 2012). This has been explained by a different root morphology and a better P absorption efficiency (PAE) of this species (Haynes and Ludecke 1981; Hart and Jessop 1984). In contrast, Davis (1991) detected comparable dry matter yields of *L. pedunculatus* and *T. repens* at small soil P concentrations in a field experiment in the same region testing seven temperate perennial pasture legumes. There, *L. corniculatus*, *T. pratense* and *T. hybridum* consistently showed significantly larger biomass production than *T. repens* with P application rates from 50 up to 800 kg P ha$^{-1}$. However, the practical feasibility of
such findings is limited to leys because of the poor persistence of *T. pratense* and *T. hybridum*.

In a pot experiment in Chile, Castillo et al. (2013) determined differences among *Lotus* species according to their PAE and P use efficiency (PUE). With a small P supply, *L. corniculatus* showed the largest PUE, meaning that the species could produce more dry matter per unit absorbed P, whereas *L. pedunculatus* had the largest PAE. P content, soil type and water availability interact with PAE, PUE, biomass production and colonization with arbuscular mycorrhizae (Castillo et al. 2013). Even cultivars and populations of one single legume species can show differing PUE and PAE and thus different potentials for application in practice (Caradus 1994; Acuña and Inostroza 2013). This indicates also the potential for plant breeding to help lessen the conflict: selection of deep-rooting genotypes and application of modern breeding methods can lead to new varieties better adapted to both drought stress and low P conditions (Marshall et al. 2016). Implementation of such strategies into practice requires also innovations in reseeding techniques to increase reseeding success while maintaining diverse permanent grassland swards.

With the choice of a suitable legume species for grasslands low in soil P like the ones we investigated, not only the total biomass production, but also the P content of the product can be enhanced (Käding and Werner 2003; Castillo et al. 2013). This also affects its quality and suitability for high-performance livestock. Legumes and plant species richness in general contribute to an enhancement of both P concentrations in the aboveground biomass as well as of P exploitation of the sward (Oelmann et al. 2011). This, of course, leads to a progressive mining of soil P over time and compensation by an appropriate application of P fertilizer will be necessary at some stage to conserve optimum nutrient ratios. *Lotus* species could be an alternative, but their particular requirements in terms of seeding and use intensity should be taken into account. For a widely successful establishment of alternative legume species in our region, breeding initiatives on traits such as frost tolerance, tolerance to pronounced alternating soil water contents and persistence are necessary.

Even if phytodiversity can increase productivity, farms that aim for high biodiversity are often less economically profitable than conventional farms. In addition to biodiversity as a self-objective, other effects of high diversity such as nutrient and water retention and forage quality can play a further role in convincing farmers to manage their grassland sustainably (Wrage et al. 2011).

Proper integration of HNV grasslands into organic farm structures at the regional scale seems to be a further, often neglected, planning option to reconcile profitable organic grassland use with nature conservation goals. Economically vital extensive grassland systems are required (Bernués et al. 2011). More extensive grassland sites like the ones in this study usually produce low-energy roughages (Gottwald and Stein-Bachinger 2018). These can be used for instance for feeding young stock and dry cows, which do not need fodder as high in energy as higher producing dairy cows, or for feeding horses and goats. The amount of low-energy roughages usable on farm in such a way should be used as a benchmark to estimate the maximum HNV-area that can be integrated at the farm level. Otherwise, the increasing demand for forage quality also in the organic livestock sector cannot be satisfied without additional concentrate (Gottwald and Stein-Bachinger 2018). Additional substitute feeding in turn makes feeding more expensive, erodes the efforts of nutrient efficiency and, therefore, should be strongly avoided in organic farming. Finally, as long as the share of these HNV grassland biomass harvests can be utilized as roughage at farm level, there is no danger of intensification of use and, coupled to this, of loss of phytodiversity. At the landscape level, this means that the proportion of P-poor and species-rich grassland requires animal populations that have corresponding requirements for energetically inferior roughage. Imbalances at the farm level, which often occur during a sustainable intensification of organic dairy farming, can be compensated at least in part by selling the late cut of hay and purchasing additional feed.

If the amount of N urgently needed for primary plant production in organic farming cannot be achieved via legumes’ N fixation on parts of the grassland due to P deficiency, the importance of alternative N sources at the farm level increases. Hence, a further option to avoid soil P eutrophication might be to drop the need for symbiotic N₂Fixation by controlled use of further natural N resources. In many regions, peat soils without rewetting potential can be used as organically managed grasslands. In many places in Germany, this is common practice. Despite small soil P concentrations, the P availability in organic soils is fairly good. A concurrent and smart control of the water regime can be used to control N mineralisation (Renger et al. 2002).

In north-east Germany, where low peat soils make up most of the grassland, large quantities of N are annually mineralised. The efficiency of its use can be markedly enhanced by a shift from extensive grazing to cutting, with a notable export of N that can be removed from the area and fed into the internal nutrient cycle of the farm. The quantities of N and P in the manure remaining after feed conversion are then available at the farm level for targeted
fertilization. This strategy can lead to equally good nutrient utilization efficiencies as the biological N-fixation by legumes preceding crops (Kayser et al. 2010).

Conclusion
To turn the proposed options concerning agronomy, breeding and site-use strategy into specific measures on farm is a future challenge for organic farmers, advisors, ecologists and consumers alike. Here, our study shows that the latter have to realize that it is unrealistic to expect increasingly cheaper, high-quality meat and dairy products from organic grasslands simultaneously with the provision of sophisticated ecosystem services like the development of HNV grassland free of charge. This insight should justify a policy that provides targeted subsidies for the development of phytodiversity of grassland areas on organic farms, even if basic subsidies have already been granted for the development of organic farming. The recommendations should be considered as a thought-provoking impulse by agricultural consultants and an exchange of experience among farmers should be encouraged.

The relationships shown can serve as a case study for the various and interacting effects of an ecosystem services cascade according to Potschin and Haines-Young (2011). While ‘supporting services’ based on the exploitation of environmental factors (here: the soil and the natural legumes of permanent grassland) can be transferred more efficiently into a ‘final service’ (here: usable fodder) by a regulatory intervention (here: P-fertilization), beyond the ‘production boundary’ a ‘benefit’ (here: healthy food) can be discerned on the socio-economic scenery, which, however, is not, as often assumed, associated with an ‘additional value’ (here: biodiversity).

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