

Species turnover reveals hidden effects of decreasing Nitrogen deposition in mountain hay meadows

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ABSTRACT

Nitrogen (N) deposition is a major threat to biodiversity in many habitats. The recent introduction of cleaner technologies in Switzerland has led to a reduction in the emissions of nitrogen oxides, with a consequent decrease in N deposition. We examined different drivers of plant community change, i.e. N deposition, climate warming, and land-use change, in Swiss mountain hay meadows, using data from the Swiss biodiversity monitoring program. We compared indicator values of species that disappeared from or colonized a site (species turnover) with the indicator values of randomly chosen species from the same site. While oligotrophic plant species were more likely to colonize, compared to random expectation, we found only weak shifts in plant community composition. In particular, the average nutrient value of plant communities remained stable over time (2003-2017). However, we found that the nutrient values of colonizing species showed the largest deviations from random expectation, suggesting that N deposition or other factors that change the nutrient content of soils were important drivers of the species composition change over the last 15 years in Swiss mountain hay meadows. In addition, we observed an overall replacement of species with lower indicator values for temperature with species with higher values. Apparently, the community effects of the replacement of eutrophic species with oligotrophic species was outweighed by climate warming. Our results add to the increasing evidence that inferring species turnover will generate a far more reliable understanding of the biotic response to changing environments than solely tracking average community composition.

INTRODUCTION

Nitrogen (N) deposition is the entry of reactive nitrogen compounds into soil, water, and vegetation, input from the atmosphere to the biosphere. Since nitrogen is an essential plant nutrient and many species-rich communities are adapted to conditions of low nitrogen availability (Vitousek et al. 1997), the addition of nitrogen is likely to change these communities. Indeed, together with land-use and climate change, N deposition is one of the major threats to biodiversity (Sala et al. 2000; Bobbink et al. 2010; Murphy and Romanuk 2013). While there is strong evidence for the reduction in diversity of species-rich grasslands due to increased N deposition (Stevens et al. 2004; Duprè et al. 2010; Maskell et al. 2010; Wesche et al. 2012), mountain grasslands have received less attention (Humbert et al. 2016).

In many parts of Europe, measures to reduce atmospheric pollution have successfully reduced emissions of nitrogen oxides since the late 1980s, with an according decrease in N deposition (Tørseth et al. 2012; Fowler et al. 2007). However, there are a number of factors that may prevent the recovery of plant communities that suffered from increased N deposition. Among others, N deposition is still high at many sites, since in contrast to nitrogen oxides, there was only a minor recent decrease of ammonia emissions. Thus, even if N deposition is reduced, large areas might still be above the critical threshold above which harmful effects on plant diversity do

occur (i.e. above the critical load for N deposition; Bobbink et al. 2010; Slootweg, Posch, and Hettelingh 2015; Rihm and Achermann 2016). Furthermore, there is a possibility that communities reach an alternative stable state after decades of increased N deposition and that the respective plant species are unlikely to disappear even if N deposition is reduced (Stevens 2016). Or, if oligotrophic species disappeared from the entire landscape, dispersal limitation may prevent oligotrophic species from recolonizing sites (Dirnböck and Dullinger 2004). Therefore, it is an open question whether and how fast the reduction in N deposition rates will lead to the recovery of plant communities.

Recovery of existing plant communities after high N deposition would imply that the state of communities measured at different points in time is improving over time (i.e. improving biodiversity endpoints sensu Rowe et al. (2017)). Species richness, a biodiversity endpoint that can be assessed and communicated relatively easily, is often negatively related to nitrogen deposition (Maskell et al. 2010; Field et al. 2014; Rowe et al. 2017). Other metrics that are potentially more useful to reflect favorable changes can be derived from the traits of the species in a community. In Europe, environmental preference of plants has often been expressed using indicator values assigned to each plant species (Ellenberg et al. 1992; Landolt et al. 2010). Examples of such metrics would be the number of oligotrophic species, or the average indicator value of the species in a community (Roth et al. 2013; Rowe et al. 2017). However, the lack of a temporal trend in such biodiversity endpoints – particularly in species richness – does not necessarily mean that species composition remains unchanged. This is because immigration and extinction might be equally frequent and may cancel each other out (Hillebrand et al. 2018). Thus, a useful approach to understanding biodiversity change is through estimates of species turnover reflecting both colonization and local extinction (Hillebrand et al. 2018), especially if colonization and local extinction are compared to random expectation (Chase and Myers 2011).

In Switzerland, grassland accounts for 70% of the agricultural land. With extensive cultivation, permanent grassland has a very high biodiversity. This applies in particular to the meadows in the alpine region, where meadows with a high plant diversity are also of agronomical importance (Leiber et al. 2006). In mountain hay meadows, the spatial variation of species richness in vascular plants is negatively correlated with N deposition (Roth et al. 2013), suggesting that mountain grasslands are negatively affected by increased N deposition. However, between 1990 and 2010, NO_x emissions in Switzerland decreased by 46% and NH₃ emissions by 14% (Maas and Grennfelt 2016). Potentially, this could have led to a partial recovery of plant communities.

In addition to N deposition, Swiss mountain ecosystems are also threatened by other drivers of global change. In Switzerland, temperatures increased from 1959 to 2008 at all altitudes, with an average warming rate of 0.35°C per decade, which is about 1.6 times the northern hemispheric warming rate (Ceppi et al. 2012). This climate warming is likely to interact with N deposition to drive plant community changes (Humbert et al. 2016). Indeed, in an earlier study we found that, at the landscape scale, plant communities responded to climate warming even within a relatively short time period (Roth, Plattner, and Amrhein 2014). Steinbauer et al. (2018) suggest that particularly the shift of plant communities at mountain summits is the result of recent climate warming, and they assume an interaction with airborne N deposition. Furthermore, traditional management regimes are currently changing, which also has major impacts on plant communities in mountainous regions of Europe (Niedrist et al. 2009; Homburger and Hofer 2012). Management regimes of easily accessible mountainous areas are often being intensified, while poorly accessible mountainous areas are abandoned (Tasser and Tappeiner 2002; Strebel and Bühler 2015). Note that fires—an important driver of biodiversity in other grassland communities (Ratajczak et al. 2014)—hardly happen in Central European mountain hay meadows.

Here, we used data from the Swiss biodiversity monitoring program (Weber, Hintermann, and Zangger 2004) to address the following questions: (1) Did biodiversity endpoints that are likely to reflect temperature, precipitation, N deposition, or land-use intensity change over the last 15 years? (2) Was species turnover correlated with the average temperature, precipitation, N deposition, or inclination (we expect steep areas to be less intensively managed)? (3) Did species that newly colonized or disappeared from (local extinction) sites differ from random expectation, according to their indicator values for temperature, soil moisture, nutrients, or light?

MATERIALS & METHODS

Monitoring data and community measures

We analyzed the presence/absence of vascular plants sampled in the Swiss Biodiversity Monitoring (BDM) program that was launched in 2001 to monitor Switzerland's biodiversity and to comply with the Convention on Biological Diversity of Rio de Janeiro (Weber, Hintermann, and Zangger 2004). The sampling sites were circles with a size of 10 m², and data collection was carried out by qualified botanists who visited each sampling site twice within the same season. During each visit, all vascular plant species detected on the plot were recorded except for young plants

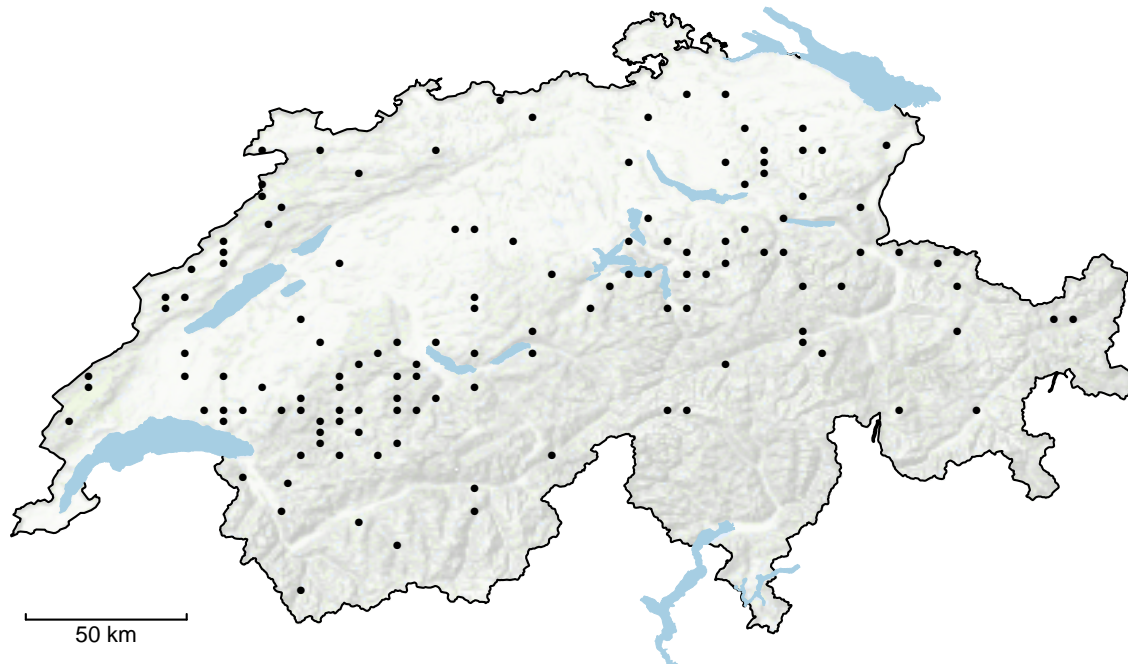


Figure 1. Distribution of the 129 study sites across Switzerland. Background data source: Swisstopo, Federal Office of Topography.

that have not yet developed at least the first pair of leaves after the cotyledons. For details on the field methods see Plattner, Birrer, and Weber (2004), Roth et al. (2013) and Roth et al. (2017).

After the sampling of the plant data, the botanists also assigned a habitat type to each sampling site according to the classification system developed for Switzerland (Delarze and Gonseth 2008). We matched the habitat types of the Swiss classification system with the categories from the EUNIS system (level-3 classification; Davies, Moss, and Hill 2004) and selected all sampling sites in mountain hay meadows (EUNIS E2.3). We analyzed the data from 2003 to 2017. During that study period, each sampling site was surveyed once per five-year period: the first period lasted from 2003 to 2007, the second from 2008 to 2012, and the third from 2013 to 2017. These selection criteria yielded 129 sites that had been sampled three times during 2003-2017. The distribution of sites across Switzerland is given in Fig. 1.

Before the analyses we removed all records that were not identified on species level. For each survey (that consisted of two visits per season) we then calculated the following biodiversity endpoints (Rowe et al. 2017): We used the number of recorded species (species richness) that can be easily related to many conservation targets (Rowe et al. 2017). Additionally, we calculated the community mean of the Landolt indicator values of recorded species. Similar to the Ellenberg's indicator values (Ellenberg 1974), the Landolt values are ordinal numbers that express the realized ecological optima of plants species for different climatic, soil or land-use variables. The Landolt indicator values were developed for the specific situation in Switzerland, published the first time in Landolt (1977) and recalibrated in Landolt et al. (2010). Their predictive power was tested in different studies (e.g. Scherrer and Körner 2011). We analysed the indicator values for temperature (1: high elevation species; 5: low elevation species), soil moisture (1: species that grow in soils with low moisture; 5: species that grow in water-saturated soils), nutrients (in particular nitrogen, but also phosphorus; 1: species that grow in nutrient-poor soils; 5: species that grow under nutrient-rich conditions) and light (1: species that grow in shade; 5: species that predominantly occur in bright places).

In addition to the five biodiversity endpoints that describe the state of plant communities for each site at a given time point, we also estimated the temporal turnover (i.e. species exchange ratio sensu Hillebrand et al. (2018)) as the

proportion of species that differ between two time points, to describe the community change between two points in time.

To test data quality, independent replicate surveys are routinely performed within the BDM program by botanists who were not involved in the regular BDM surveys. The regular surveyors did not know if and which sites were replicated (Plattner, Birrer, and Weber 2004). We used the data from 14 such replicated surveys that were available to calculate the pseudo-turnover, which is the proportion of species that differed between two surveys that were conducted by two different surveyors during the same year on the same site.

Environmental gradients

We expected different drivers of global change to cause temporal change in mountain hay communities. To better disentangle the importance of these mechanisms, we ordered the sites along four main environmental gradients. First, we expected communities to respond to climate warming (Roth, Plattner, and Amrhein 2014). To describe the temperature gradient, we used the mean annual temperature per site from the WordClim database (Fick and Hijmans 2017). The average \pm SD mean annual temperature at our sites was 5.85 ± 2.16 °C. The monthly-mean surface air temperature for Switzerland shows a linear increase of 1.29 °C per 100 years between 1864 and 2016 with the warmest three years of the entire period measured in 2011, 2014 and 2015 (Begert and Frei 2018). Also from the WordClim database we used the annual precipitation per site, another key driver for plants that is likely to be affected by climate change (Beier et al. 2012). The average annual precipitation at our sites was 1284.71 ± 196.53 mm. Further, we estimated atmospheric N deposition for each site using a pragmatic approach that combined monitoring data, spatial interpolation methods, emission inventories, statistical dispersion models and inferential deposition models (Rihm and Achermann 2016). The average nitrogen deposition at our sites was 17.54 ± 6.47 kg ha⁻¹ yr⁻¹ in 2000 and 14.84 ± 6.12 kg ha⁻¹ yr⁻¹ in 2015. We assume that N deposition is a surrogate for N availability in the soil because we found that the spatial variation in oligotrophic species richness is clearly linked to N deposition (Roth et al. 2013; Roth et al. 2017). Unfortunately, we do not have any soil measurements to support this assumption. It seems that the total N as well as the soil carbon content down to 20 cm depth in the extensively used grassland sites of the Swiss soil monitoring network (NABO) were mostly stable over the last 20 years (R. Meuli, personal communication). Finally, we used inclination as a proxy for land-use intensity, because we assumed that steeper sites are likely to be less intensively managed (Strebel and Bühler 2015). The average inclination at our sites was 15.87 ± 9.66 °.

Statistical analyses

To estimate the linear trend over time for each of the five biodiversity endpoints, we applied linear mixed models (LMM) with normal distribution except for species-richness with Poisson distribution. We specified site-specific trends with the assumption that the between-site differences in intercepts and slopes can be described with normal distributions (i.e. a random intercept random slope model, Gelman and Hill 2006). Model parameters were estimated in a Bayesian framework using the R-Package *rstanarm* (Stan Development Team 2016, Muth, Oravecz, and Gabry (2018)).

To infer whether species turnover was changing along the gradient, we used a binomial generalized linear mixed model (GLMM) with the proportion of species that differed between two surveys as dependent variable and the site gradients and period (first/second vs. second/third surveys) as predictors, and with site-ID as random effect. Model parameters were estimated in a Bayesian framework using the R-Package *rstanarm* (Stan Development Team 2016, Muth, Oravecz, and Gabry (2018)).

To infer whether species that colonized or disappeared from a site had particular indicator values that differed from the other species at that site, we produced for each site a list with all species that were recorded during the three surveys (total community). We then calculated the community mean (CM) of the indicator value for all species that colonized the site during the three surveys (i.e. not recorded during the first survey and recorded during the second, or not recorded during the second and recorded during the third survey). We then randomly selected the same number of species from the total community and also calculated the community mean of the value for these species (random-CM). We repeated the random selection of species 1000 times. We then calculated the differences of the CM minus the average of the random-CMs to obtain a standardized difference (standardized-CM) of how different the colonizing species were from random expectation. For example, a difference < 0 would suggest that the indicator values of colonizing species were lower than might be expected from random colonization from the species-pool for this site. We applied this method for both colonizing and disappearing species and for the indicator values for temperature, soil moisture, nutrients, and light (see Appendix A). We then tested whether this standardized difference was changing along the corresponding gradient (inferred from independent datasets, see Environmental gradients

section) using linear models. Model parameters were estimated in a Bayesian framework using the R-Package *rstanarm* (Stan Development Team 2016, Muth, Oravecz, and Gabry (2018)).

We used logistic GLMM (i.e. a generalized linear mixed model with binomial distribution) to analyze whether the colonization probability or local survival probability was changing along the Nitrogen deposition gradient and whether this change depended on the species indicator value for nutrients. To analyze the colonization probability of species, we selected all species that were not observed during the first survey. For each of these species we asked if they were observed ($\text{Occ} = 1$) or not observed ($\text{Occ} = 0$) during the second survey. The same was also done for all species that were not observed during the second survey whether they were observed or not observed during the third survey. The variable ‘occ’ was then used as dependent variable in the logistic GLMM. As predictor variables we used the N deposition of the site, the average indicator value for nutrients, and the interaction of these two variables. Additionally, species-ID and site-ID were included as random effects. The same logistic GLMM was also used to investigate local survival probabilities. In that case, however, we selected all species that were recorded during the first or second survey, and the variable ‘occ’ then indicated whether or not the species was also observed during the next survey. Model parameters were estimated using an approximate Bayesian approach using the R-Package *arm* (Gelman and Su 2018).

To estimate the effect of N deposition on total species richness at a given point in time, we described the plant species richness at the sites using a generalized linear model (GLM) with Poisson distribution and the logarithm as link function. As predictors we used the four environmental gradients as described above. Model parameters were estimated using an approximate Bayesian approach using the R-Package *arm* (Gelman and Su 2018).

Data accessibility and reproducibility of results

Data and R Markdown documents (Manuscript.Rmd and Appendix_A.Rmd) to fully reproduce this manuscript including figures and tables are provided at <https://github.com/TobiasRoth/NDep-Trend>. An R Markdown document is written in markdown (plain text format) and contains chunks of embedded R code to produce the figures and tables (Xie, Allaire, and Golemund 2018). Raw data for analyses are provided in the folder ‘RData’ and the folder ‘R’ contains the R-Script that was used to export the data from the BDM database. The folder “Settings” contains a list of all the R packages (including version number) that were in the workspace when the manuscript was rendered. Note, however, that the files on the github repository might have slightly changed since submitting the manuscript. The version of all files will be archived on Zenodo upon acceptance of the manuscript.

RESULTS

Plant communities

In total, 623 plant species were recorded on the 129 plots. Including the data of all three visits, 45.83 ± 11.54 (average \pm SD) species were observed per plot. The lowest number of species recorded during a survey was 19 species and the highest number was 81 species. In average $7.70 \pm 8.20\%$ of the recorded species were annual species with one plot reaching up to 7.70% annual species. The average indicator value for temperature across all surveys was 3.12 ± 0.37 ranging from 1.59 to 3.66. The average indicator value for soil moisture was 2.99 ± 0.20 ranging from 2.46 to 3.59. The average indicator value for nutrients was 3.20 ± 0.35 ranging from 2.26 to 4.00. And the average indicator value for light was 3.55 ± 0.19 ranging from 2.83 to 4.16.

Temporal change in community structures

Species richness and the four measures of plant community structure according to Landolt indicator values (i.e. biodiversity endpoints) suggested that plant communities in mountain hay meadows were rather stable between 2003 and 2017 and did not show a clear increase or decrease over time (Table 1): for each of the three 5-year survey periods, the averages of species richness and the average indicator values for temperature, soil moisture, nutrients and light did not vary much among the three sampling periods, and the estimated trends were rather small. The results from the linear mixed models suggest that a linear temporal change was most likely for the community mean of the indicator value for temperature (probability of increase: 0.97), followed by the community mean of the indicator value for light (probability of decrease: 0.93). A linear temporal change was least likely for the species richness (probability of increase: 0.53). The chance that the community mean of the nutrient value decreased between 2003 and 2017 was 0.67 (0.33 gives the probability for an increase).

Species turnover

The average \pm SD percentage of species that differed between replicated surveys (i.e. different botanists surveyed the sites) was $\text{NaN} \pm \text{NA}\%$. This turnover between replicated surveys was lower than the observed temporal turnover:

Table 1. Average measures of the biodiversity endpoints for the three sampling periods (period 1: 2003-2007; period 2: 2008-2012; period: 2013-2017). The temporal trends are given as change per 10 years and were estimated from linear mixed models with normal distribution (except for species richness with Poisson distribution and a log-link function). The measure of precision for the temporal trend is given as the 5% and 95% quantiles of the marginal posterior distribution of the linear trend (90% credible intervals). The column 'Prob. for trend' gives the probability that the linear trend is > 0 . Indicator values according to Landolt et al. (2010).

Measures	Period 1	Period 2	Period 3	Trend	5%	95%	Prob. for trend
Species richness	45.72	46.02	45.74	0.00	-0.03	0.03	0.53
Temperature value	3.11	3.13	3.13	0.01	0.00	0.03	0.97
Humidity value	2.99	2.98	2.99	0.01	-0.01	0.02	0.80
Nutrients value	3.20	3.20	3.20	0.00	-0.02	0.01	0.33
Light value	3.56	3.55	3.55	-0.01	-0.02	0.00	0.07

Table 2. Change of species turnover along the four gradients when differences between the two periods (first period: turnover between first and second survey; second period: change in turnover between second and third surveys) and species richness effects are accounted for. Estimates for the period effect (change in turnover from first to second survey), the species richness effect (change in turnover per 10 species) and along the four gradients (slopes) with the corresponding 5% and 95% quantiles of the marginal posterior distribution were obtained from a binomial GLMM.

Predictors	Estimate	5%	95%
Period	-0.09	-0.15	-0.03
Number of species	0.14	0.09	0.18
Mean annual temperature	0.04	-0.02	0.10
Mean annual precipitation	-0.06	-0.15	0.03
Nitrogen deposition	-0.07	-0.23	0.08
Inclination	-0.07	-0.14	0.00

Table 3. Difference in the average indicator value of species that (a) disappeared from a site or (b) newly colonized a site compared to the same number of species that were randomly selected from all species recorded at a site. Shown are the results from linear models, with the difference between disappeared/colonized species and random species as dependent variable and the site-measure (gradient) as predictor variable.

Indicator value	Gradient	Difference from random			Change along gradient		
		Estimate	5%	90%	Estimate	5%	90%
<i>(a) Plants that disappeared from a site</i>							
Temperature	Annual mean temperature	-0.012	-0.033	0.009	0.007	-0.002	0.016
Soil moisture	Annual mean precipitation	-0.004	-0.042	0.033	0.008	-0.013	0.030
Nutrients	Nitrogen deposition	-0.021	-0.055	0.013	0.013	-0.030	0.053
Light	Inclination	-0.023	-0.049	0.004	-0.002	-0.025	0.020
<i>(b) Plants that newly colonized a site</i>							
Temperature	Annual mean temperature	0.016	0.000	0.032	-0.001	-0.008	0.005
Soil moisture	Annual mean precipitation	0.019	-0.013	0.050	-0.001	-0.019	0.017
Nutrients	Nitrogen deposition	-0.078	-0.110	-0.047	0.060	0.023	0.097
Light	Inclination	-0.039	-0.061	-0.017	0.010	-0.010	0.028

the average percentage of species that differed between the first and second survey at a site was $37.65 \pm 10.43\%$, and the percentage of species that differed between the second and third survey was $35.66 \pm 10.36\%$. Thus, it seemed that the turnover from the first/second survey to the turnover of the second/third survey moderately decreased (period effect in Table 2). Species richness was a good predictor of species turnover: species rich sites were subject to higher turnover than sites with lower species richness (Table 2). The four gradients (temperature, precipitation, Nitrogen deposition and inclination) were less conclusive in explaining the variation in species turnover among sites.

High species turnover at a site is the result of species that disappeared from the site and/or of species that newly colonized the site. To better understand the factors that drive these changes we were particularly interested whether the species that disappeared or colonized the sites differed in indicator values compared to what would be expected if the same number of species randomly disappeared or colonized the sites (i.e. random disappearance and random colonization), and whether there is spatial variation along the environmental gradients. It seems that the indicator values of newly colonizing species differed more from random colonization than the indicator values of disappearing species (Table 3). For colonizing species, we found the largest differences from random colonization in the indicator value for nutrients: at sites with relatively low N deposition of $10 \text{ kg ha}^{-1} \text{ yr}^{-1}$, the newly colonizing species had on average a lower indicator value for nutrients than species under random colonization (column “Difference from random” in Table 3), but the differences between colonizing species and random colonization decreased with increasing N deposition (column “Change along gradient” in Table 3). Thus, at high N deposition, colonizing species did not differ from random species (see Figure 3b in Appendix A).

While colonizing species had higher temperature values compared to what we would expect under random colonization, the effect size (i.e. the absolute value of the estimate for the difference from random expectation) for temperature was about four times smaller than the effect size for nutrients. Nevertheless, the variation in the indicator value for temperature seems to be important for explaining the total species turnover. This is because disappearing species tend to have lower temperature values than random species, and colonizing species tend to have higher temperature values than random species; both processes lead to an overall replacement of species with lower temperature value by species with higher temperature values. This was not the case for the indicator value for nutrients: species with lower nutrients values tended to be more likely to disappear from, and to colonize sites compared to random species (Table 3). See also Appendix A, where we present detailed results for the comparison between colonizing or disappearing species with randomly selected species.

Potential effects of reduction in Nitrogen emissions

Nitrogen deposition decreased with increasing elevation (Fig. 2a). In 2000, only 11.63% of sites had a N deposition rate of less than $10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, all situated above 1000 m. Between 2000 and 2015, the N deposition decreased on average by $-2.70 \pm 1.74 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, with slightly higher net decreases at lower elevation (Fig. 2b).

In Fig. 3 we compare the colonization and local survival probability of oligotrophic species (indicator value of nutrients < 3) and eutrophic species (indicator value of nutrients > 3) along the N deposition gradient. Local

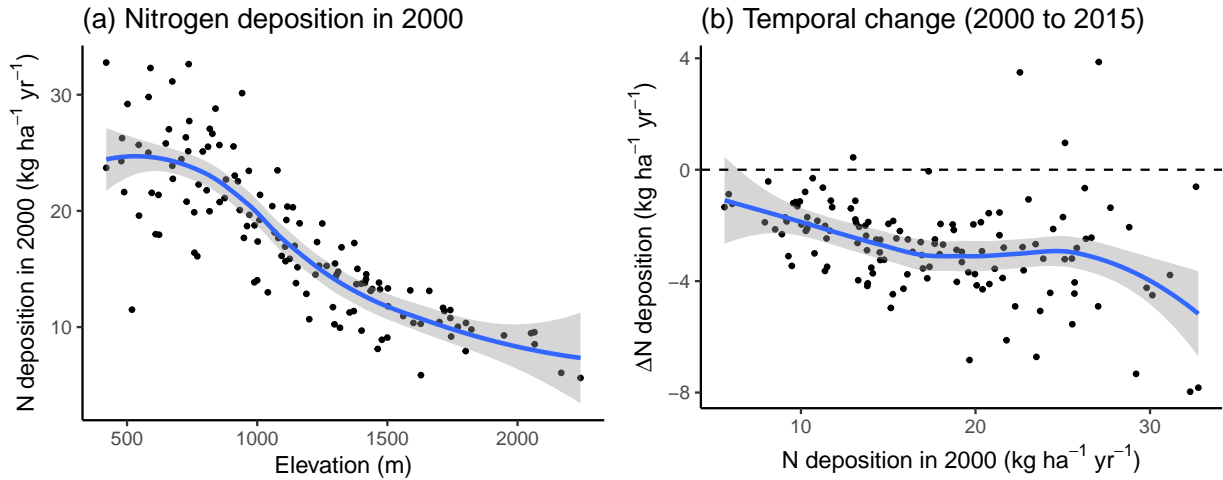


Figure 2. (a) The nitrogen (N) deposition in 2000 and (b) the change in N deposition between 2000 and 2015, along the N deposition gradient of the study sites used in 2000.

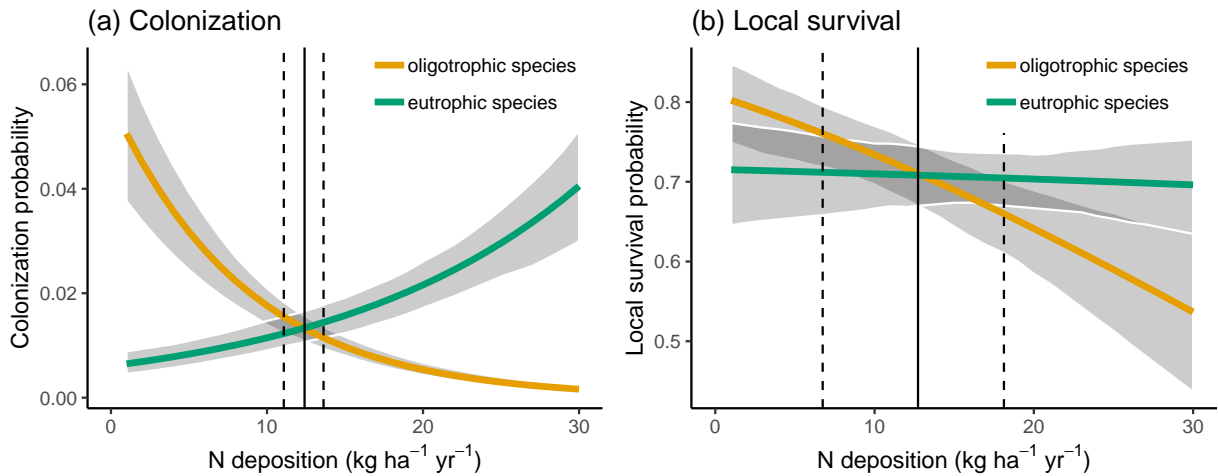


Figure 3. Colonization (a) and local survival (b) of oligotrophic species (indicator value for nutrients < 3 ; red line) and of eutrophic species (indicator value for nutrients > 3) species along the N deposition gradient. Given are means and 95%-credible intervals from logistic linear mixed models. The vertical lines indicate the deposition rate, with equal colonization or survival probabilities for oligotrophic and eutrophic species, the solid line indicating the median, and the dashed lines the 5% and 95% quantiles of the marginal posterior distribution.

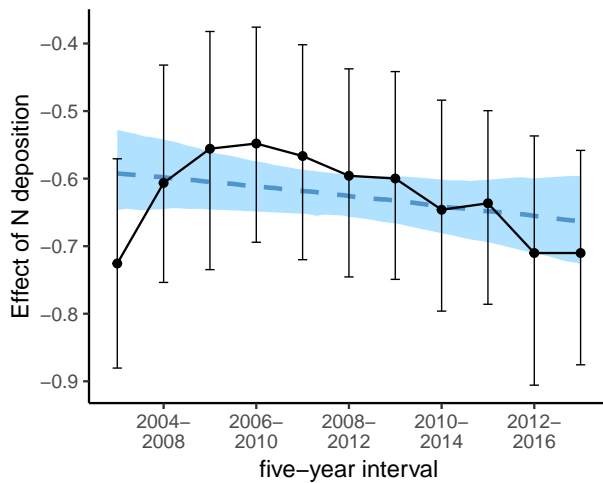


Figure 4. Effect size of N deposition on oligotrophic species richness estimated from applying the Poisson GLM, with species richness as dependent variable and N deposition plus other site covariates (elevation, precipitation, inclination, mean indicator values for soil moisture and light) as predictors, using only the surveys from one five-year interval. Note that within every five-year interval, all plots were sampled once. Effect sizes are given as averages (points) and 5% and 95% quantiles (dashed lines) of the marginal posterior distribution. The dashed blue lines gives the linear regression with the effect size as dependent variable and the five-year interval as predictor variable. The blue area gives the 5% and 95% quantiles of the marginal posterior distribution of the regression line.

survival probability was the same for oligotrophic and eutrophic species at a deposition rate of $13.07 \text{ kg N ha}^{-1} \text{ yr}^{-1}$; colonization probability was the same for oligotrophic and eutrophic species at a deposition rate of $12.41 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. In 35.66% of the sites, the deposition rate was below $12.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ at which the replacement of eutrophic with oligotrophic species is likely according to Fig. 3.

While we could not detect a consistent decrease in the average indicator values for nutrients (Table 1), the nutrient value of colonizing species was below average (i.e. a strongly negative effect size) at low deposition rate and the difference to random expectation became smaller along the N deposition gradient (second last line in Table 3). This higher colonization rate of species with low nutrient value at sites with low N deposition rate might have affected the spatial pattern of oligotrophic species richness: sites with low N deposition were likely to become more species-rich over time. This likely resulted in a steeper slope of the negative relationship between N deposition and oligotrophic species richness, when comparing this relationship with a spatial approach as in Roth et al. (2013). Indeed, if we apply such an approach at different points in time to infer the effects of N deposition on the spatial variation of oligotrophic species richness, the resulting effect size (i.e., the slope) became more negative over time (Fig. 4).

DISCUSSION

Although N deposition declined between 2000 and 2015 (Fig. 2), we could detect only weak shifts in plant community structure (i.e. biodiversity endpoints *sensu* Rowe et al. (2017)) during the same time period (Table 1). While the slight increase in average temperature indicator values suggests that plant communities adopted to increasing temperatures, the constant average nutrient value suggests that the decrease in N deposition did not yet affect plant communities. However, this apparent stability in community composition was accompanied by a marked temporal turnover in species identities. It seems unlikely that this temporal turnover can entirely be explained by methodological issues such as overlooked species. First, pseudo-turnover of species entities in independent surveys of the same site during the same season was smaller than the observed temporal turnover between two surveys from different years. Second, spatial variation of turnover showed patterns that can hardly be explained by methodological issues. For instance, species turnover varied along the N deposition gradient, with highest species turnover at sites with low N deposition (Table 2). Taken together, our results add to the increasing evidence that contemporary communities are often relatively stable at community level, but this apparent stability is often accompanied by marked turnover of species (Vellend et al. 2013; Dornelas et al. 2014; Hillebrand et al. 2018).

Species communities are shaped by a range of factors, including deterministic processes such as environmental

filtering or competitive interactions (Götzenberger et al. 2012; Janeček et al. 2013). Such factors select for species with specific characteristics. Community assembly theories thus suggest that the factors that drive the composition of species in a community can be inferred from comparing the characteristics of the species in the community with random expectation (Chase and Myers 2011). We adopted this idea and compared the indicator values of species that disappeared or colonized a site with the indicator values from randomly chosen species from the same site (Appendix A). We found that the nutrient values of colonizing species showed the largest deviations from random expectation (Table 3), suggesting that N deposition or other factors that change the nutrient content of soils were drivers of the change in species composition over the last 15 years in our sample.

In Swiss mountain hay meadows the average N deposition was still rather high with an average of 14.84 kg ha⁻¹ yr⁻¹, which is at the upper limit of the suggested critical load (Roth et al. 2017). Furthermore, Nitrogen deposition only weakly decreased by about -2.70 kg ha⁻¹ yr⁻¹ between 2000 and 2015. This is only about one tenth of the decrease in England, where N deposition decreased by 24 kg ha⁻¹ yr⁻¹ from 1996 to 2011 (Storkey et al. 2015). The still comparatively high N deposition rate and the rather low decrease in N deposition combined with the fact that most of the species are perennials likely explain why we observed no change in average nutrient value of communities. Additionally, other anthropogenic pressures such as climate change might have overruled effects of N deposition on community composition. In particular, we found that the species disappearing from the sites tended to have below average indicator values for temperature, while species that newly colonize sites had above average indicator values for temperature (Table 3). Thus, the effect of disappearing and the effect of colonizing species on the community mean for temperature is additive resulting in increasing average temperature values (Tab. 1). This is in contrast to how N deposition is affecting disappearance and colonization of species: It seems that both species disappearing from the sites as well as species colonizing the sites tend to have below average indicator values for nutrients (Table 3). Thus, the effects of disappearing and colonizing species on the average community value for nutrients partially cancel each other out. Furthermore, eutrophic species had rather high local survival across the entire deposition gradient, while oligotrophic species have much reduced local survival at higher N deposition rates. This suggests that mountain hay meadow communities can reach alternative stable states, with eutrophic species that are unlikely to disappear even if N deposition is reduced (Stevens 2016). Taken together, these reasons might explain why the composition of mountain hay meadow communities responded stronger to climate warming than to nitrogen reduction, although the reduction in nitrogen resulted in above average colonization of oligotrophic species.

Our results conform to the results of recent reviews on biodiversity change suggesting that local-scale species communities are often undergoing profound changes but not systematic loss (Dornelas et al. 2014). However, our comparison with replicated surveys from the same year warn that an important portion of the observed turnover of species might be due to pseudo-turnover (i.e. species difference between two surveys that were conducted by two surveyors during the same year on the same site). Given that the BDM program had invested significantly in developing reproducible methods and has continuously invested in quality control (Plattner, Birrer, and Weber 2004), the recorded pseudo-turnover was quite high. A potential explanation is that the replicated surveys are not conducted during the same days and in few cases the situation might have changed profoundly for example because the meadow was cut between the surveys. Furthermore, one of the botanist may not have identified difficult to identify species on species-level (i.e. differences in botanical skills), and—because we analysed only records that were identified on species level—it increased pseudo-turnover even though both botanists detected the species. This must be taken into account when evaluating the presented results. For example, we find that sites with high species richness have higher turnover than sites with a low species richness. This seems biologically plausible, since the average species coverage in species-rich sites must be lower than in species-poor sites and species with low coverage probably have a higher turnover. At the same time, however, the result could also be simply due to pseudo-turnover, since species identification in species-rich sites is probably more difficult than in species-poor sites. However, for the presented results that suggest differences in colonization or local survival in relation to the species indicator values we can hardly imagine how this could be caused by methodological issues. We are thus confident that most of the results presented in this study are largely unbiased.

Observational studies along a gradient of N deposition often conclude how the spatial variation in species richness is related to N deposition (Stevens, Thompson, et al. 2010; Roth et al. 2013; chapter 4 in Vries, Hettelingh, and Posch 2015). They assume that the spatial variation in species richness (or other metrics of community composition) arose because of unequal species loss of different areas over time, resulting from elevated N deposition chronically experienced by some areas. Although there is evidence supporting the pertinency of such a ‘space for time substitution’ for detecting the effects of N deposition on plant diversity (Stevens, Duprè, et al. 2010), this approach cannot replace studies that relate temporal patterns in species composition with N deposition (De Schrijver et al. 2011). There are only a limited number of studies directly relating temporal trends of plant species diversity to

varying amounts of N deposition in existing communities (Clark and Tilman 2008; Storkey et al. 2015; Stevens 2016). In a recent study, we used the mountain hay meadow data from a single survey only and estimated the empirical critical load along the N deposition gradient at which species richness of oligotrophic species richness starts to decrease with increasing N deposition (Roth et al. 2017). Using this spatial variation in species richness and N deposition, Roth et al. (2017) estimated a critical load for mountain hay meadows of $13.1 \text{ kg ha}^{-1} \text{ yr}^{-1}$. In the current study, we estimated the rate of N deposition at which local survival probability or colonization probability was equal for oligotrophic and eutrophic species. Using a temporal approach in the present study we obtained very similar results as Roth et al. (2017) using a the spatial approach. These results are thus validating the space for time approach, at least for Swiss mountain hay meadows.

However, the results of Fig. 4 also show that the results of spatial comparisons must be interpreted carefully. When we applied inferred the spatial variation in oligotrophic species richness with the same covariates as in Roth et al. (2017) for different study periods, to infer how oligotrophic species richness is decreasing along the N deposition gradient, the relationship appeared to vary between study periods. The decrease gradually became steeper (more negative) over time, except for the first two study periods. Our first interpretation was that the N deposition effect became stronger over time. This was against our prediction that the effect of N deposition should become weaker over time, since N deposition is decreasing during the study period. Then we realized that species turnover was highest at low N deposition sites (Tab. 2). At low N deposition rates, colonizing species have below average indicator values for nutrients. It seems that the decrease in N deposition resulted in oligotrophic species replacing eutrophic species particularly at sites with low N deposition. This seems to explain why the decline in oligotrophic species richness inferred from spatial patterns of species richness and N deposition is becoming steeper over time. And this may be interpreted as evidence that plant communities are recovering at least at low deposition sites, and that the negative N deposition effects have not become stronger over time.

CONCLUSIONS

Comparing the indicator values of colonizing and disappearing species with random expectation, we found that oligotrophic species are currently more likely to colonize mountain hay meadows than eutrophic species, which might be the result of the recently observed decrease in atmospheric N deposition. However, our results also indicate that the recovering of mountain hay meadows after high N deposition might take much longer than transferring species-rich mountain hay meadows to species-poor communities with a large proportion of eutrophic species. This is because eutrophic species have high local survival probabilities, even after N deposition decreases again. Our study adds to the understanding of contemporary biodiversity change (Magurran et al. 2018), and it supports the notion of Hillebrand et al. (2018) that considering species turnover will generate a far more reliable view of the biotic response to changing environments than solely tracking community composition.

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