The biodiversity cost of reducing management intensity in species-rich grasslands: Mowing annually vs. every third year

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Abstract

Mowing is an important management method for species-rich semi-natural grasslands in Europe. Since mowing is costly, it is important to find a balance between mowing frequency and conservation benefits. We compared vegetation data from eleven field trials situated in southern Sweden that involved two mowing regimes, annually and every third year, as well as a no-management control. After approximately 14 years, mowing every third year showed (i) a drop in species richness and Shannon and Gini-Simpson diversity indices, (ii) an increase in woody species, and (iii) increases in tall-grown species. However, there were no apparent changes in (iv) species that were indicative of poor management, nor (v) those indicating good management. For one of the trials, data after 38 years were also evaluated. Compared with annual mowing, there were strong negative changes in the number of species in the untreated control, while the results were conflicting for mowing every third year. In conclusion, the expected loss of conservation values from reduced mowing intensity was 50 to 60% of the loss after abandonment. The outcomes, however, varied among the eleven sites.

Keywords:

Cutting, Indicators, Meta-analysis, Mowing, Odds ratio, Response ratio, Semi-natural grassland, Sweden

Nomenclature

Karlsson (1998)

Introduction

Ongoing changes in agriculture mean that it is a challenge for conservation efforts to maintain the rich biodiversity that is associated with traditionally managed semi-natural grasslands in Europe (Kull and Zobel 1991, Poschlod et al. 2009, D'Aniello et al. 2011, Wilson et al. 2012, Habel et al. 2013, Babai & Molnár 2014). Grazing and mowing are the two options for exploiting the biomass of grass-dominated vegetation, and both seem relatively similar in regard to the benefit that they provide for preserving biodiversity (Tälle et al. 2015, 2016). However, the amount of available cattle for conservation-oriented grazing is decreasing (Kumm 2003), while the labour costs for annual mowing in most cases remain high (Schreiber et al. 2009, Török et al. 2011).

Although not the first choice, a reduction management intensity might allow more land to be managed. Managers can skip mowing every second year and thereby double the acreage that is mowed or reduce the stocking density (a limited number of animals spread over a larger area). Theoretically, there should be an optimal balance between the available resources (e.g., number of animals, or cost for labour), area covered and biodiversity benefits provided. Put another way, a larger area with relaxed management (intensity or type) might be preferred over a smaller area under the best management option as long as the losses in biodiversity are acceptable. Hence, it is important to estimate the “cost”, in terms of lost biodiversity, for reducing the management intensity (Marriott et al. 2004).

As the grazing intensity is notoriously difficult to estimate, experiments that vary the stocking density are less suited to address this issue than experiments assessing the effect of different mowing intensities on biodiversity. The results from studies comparing the effect of higher and lower mowing intensities vary, with some in favour of higher mowing intensities (e.g., Köhler et al. 2005, Noordjik et al. 2009, and others in favour of lower mowing intensities (e.g., Everwand et al. 2014, Kőrösi et al. 2014). However, few studies have examined the effect of mowing less than once per year (but see e.g., Bakker et al. 2002). Hence, the effect of different mowing intensities on biodiversity remains poorly understood, but could be expected to resemble secondary succession when management ceases, that is, woody species and tall-grown species increase while the diversity indices and richness decrease.

Here, we analyse data from eleven field trials in southern Sweden in which mowing every year was compared with mowing every third year as well an unmanaged control. We intended to estimate the rate of loss of conservation values from these eleven grasslands over 14 years. At one site, data were also available after 38 years. More specifically, when converting to more infrequent mowing, we expected:

(i) a decrease in the number of species and diversity indices (Shannon, Gini-Simpson, Shannon evenness)

(ii) a decrease of species indicative of good management

(iii) an increase in species indicative of poor management

(iv) an increase in woody species

(v) an increase in tall-grown species

Methods

Study sites

In the early 1970s, a long-term experiment was established at eleven experimental sites at nine locations in southern Sweden (Figure 1). Two of the locations (Ekenäs, Tagel) had two experimental sites each. The mean annual temperature in southern Sweden is approximately 6 °C, mean annual precipitation is 500-1000 mm (Alexandersson et al. 1991) and growing period is 180-220 days (Sjörs 1999). Before the start of the experiment, most of the sites were managed using grazing. However, two of the sites (Dämkärr and Österplana) had been abandoned for a few years, and one site (Gränö) was mowed. Furthermore, two of the sites (Gränö and Tagel/former field) were fertilized prior to the start of the experiment (Table 1, Hansson 1991). The sites differed in vegetation type and productivity, but the majority of the sites were of the mesic meadow type (Table 1), with medium productivity (Hansson 1991). If using the Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora (Swedish Environmental Protection Agency 2011), the sites are semi-natural dry grasslands (6210), hay meadows in submontane zones (6510), Fennoscandian wooded pastures (9070) or wet meadows (Molinion caeruleae) (6410).

Figure 1. Study locations in southern Sweden. Ekenäs and Tagel had two experimental sites each.

Table 1. Description of the experimental sites established in southern Sweden for the comparison of the management methods in semi-natural grassland vegetation. X indicate that the data were used in the present study, while - indicates that existing data were not used (because of the design of data collection).

Site	Year of inventory 1973	1975	1980	1986	1987	Vegetation type	Soil category	Management at start of trial
Andersby, Dannemora	X		X	X		Moist	Humus-rich light clay	Grazing
Bråbo, Oskarshamn	X		X	X		Mesic	Rock moraine	Grazing
Bräcke, Åmål	X		X	X		Mesic	Silt	Grazing
Dämkärr, Gamleby	-			X		Mesic	Humus-rich silt	Abandoned
Ekenäs, Flen								
Moist		X		X		Moist	Highly humus-rich light clay	Grazing
Mesic		-	X	X		Dry-mesic	Humus-rich loamy	Grazing
Gränö, Värmdö	-			X		Moist	Slightly humus-rich silt	Mowing, fertilized
Sättra Ödeshög	-		X	X		Mesic	Slightly clayey sand	Grazing
Tagel, Alvesta								
Mesic	-		X	X		Mesic	Rocky sand	Grazing
Former field	-				X	Mesic	-	Grazing, fertilized
Österplana, Götene	-		X	X		Dry	Gravelly clay loam	Abandoned

Experimental design

The experiment was set up with the aim of assessing vegetation changes as a result of the introduction of several management methods (Steen 1976; Fogelfors 1982; Hansson 1991). For the present study, we used three management methods: annual mowing, mowing every third year, and an untreated control. Treatment plots (5 m × 20 m) were established in a randomized block design with two replicates. Mowing took place in late July or early August using a scythe or sickle bar mower. Because all of the experiments were surrounded by grazed grasslands, the plots were fenced.

Vegetation sampling

Fieldwork was conducted in July, before mowing took place. Initial vegetation assessments were conducted, but the protocol for data collection varied between sites (Steen 1976). Most experiments were assessed again after approximately 8 years (Fogelfors 1982), this time using a standard protocol. All of the sites were again visited in 1986-7 using the standard protocol, which involved using five fixed 1 m^2 subplots per treatment plot (at Bräcke, three 1 m^2 fixed subplots were used).

We used the presence/absence of species in the subplots to generate a frequency per treatment plot. In some of the analyses, we used data as given in previous reports (Steen 1976, Fogelfors 1982, Hansson

1991), but corrected for numerous typing errors (see Tälle et al. 2015 for details on data retrieval and transcription). In some analyses, we first created a dataset with a species list that allowed analyses of species-wise data over the eleven trials. The latter meant deleting some records of only partially identified records (e.g., deleting *Luzula* sp., since the majority of *Luzula* records were identified at the species level). It also meant creating groups of species that had been identified with different levels of ambition (e.g., “other *Agrostis* spp.” i.e., all *Agrostis* recorded excluding *A. gigantea*). Finally, we merged the woody taxa, which were always infrequent, into “all woody species”.

One trial (Sättra ängar) has been continually managed since 1973, so here we used data collected in 1973 (Steen 1976), 1980 (Fogelfors 1982), 1986 (Hansson 1991) and 2011 (unpublished). Data also exist for 2000, but because a different methodology was used when assessing the vegetation and because the survey was conducted late in the season, well after mowing (Wahlman & Milberg 2002), we decided to not consider them in the present analysis.

Classification of plant species

To enable a comparison of data collected from different sites that differ in species composition, we simplified the data in some analyses by classifying grassland species as indicators. Based on previous experience

(Milberg et al. 2014, Tälle et al. 2014, 2015), we chose a well-documented indicator system that targeted species-rich semi-natural grasslands in southern Sweden (Ekstam & Forshed 1992). In this indicator system, which is widely used by practitioners, species are classified according to the rate at which a species is lost during secondary succession from grassland to forest (four classes) and according to a nitrogen availability gradient (three classes). We classified species as (i) indicators of good management for species richness if they decrease in early succession stages (classes A and B) and grow in sites poor under nitrogen (N1); (ii) indicator species of poor management if they do not decrease until later successional stages (classes C and D) and grow in nutrient-poor sites or sites with moderate levels of nitrogen (classes N1 and N2).

Statistical analyses

Species richness and diversity

The treatment effect on the number of species recorded in the 5 (or 3) m^2 sampled per treatment were calculated as ln-transformed response ratios (RR). To avoid bias in the RR due to small sample sizes, Lajeunesse's (2011) method of calculating ln(RR) was used:

$$\ln(RR) = \ln((X_T/X_C) + 1/2[SD_T^2/N_T X_T^2 - SD_C^2/N_C X_C^2]) \qquad \text{(eqn. 1)}$$

where X is the mean, T is mowing every third year, C is annual mowing, SD is the standard deviation and N is the sample size. Corresponding ln(RR) values were calculated for two diversity indices (Shannon, Gini-Simpson) and for evenness (Shannon). Using a meta-analysis tool (Comprehensive Meta-analysis version 2; Biostat, Inc. 2006; www.meta-analysis.com), we then calculated the weighted average, based on the random effects model, for the eleven trials. A positive value of ln(RR) signifies a more positive effect of mowing every third year.

To estimate the rate of change, we used trial-wise estimates of species numbers and diversity, where comparable, to follow the development of ln(RR) over time. Three trials had data from three time points, five from two (the remaining three had only been surveyed once with the standard protocol). The slopes of the trial-wise regressions for these eight trials were then used to calculate the average and weighted average (giving the three sites with the best data more weight).

Species indicating good or poor management

We analysed the data using an odds ratio as a way of overcoming differences in the vegetation composition and the differing number of subplots. This analysis method compared the odds of a species recorded being an indicator, contrasting two different treatments. For each type of indicator, we first calculated the total number of indicators and non-indicators (i.e., all other species) per subplot or per treatment plot if subplots were not used. We then summed the number of indicators and non-indicators per treatment plot (if subplots were surveyed). This measure reflects the frequency of the indicator species and non-indicator species in each treatment plot.

To compare annual mowing with mowing every third year, or the untreated control, a random effects meta-analysis was performed using Comprehensive Meta-analysis.

The effect sizes were measured as log odds ratio (logOR):

$$\text{logOR} = \log [(A \times D/(B \times C)] \quad (1)$$

where A = the frequency of indicators in the annual mowing treatment; B = the frequency of non-indicators in the annual mowing treatment; C = the frequency of indicators in the alternative treatment; and D = the frequency of non-indicators in the alternative treatment. A positive logOR signifies higher odds of finding indicators in plots with annual mowing. Since each trial contained two blocks, each trial contributed two entries to the meta-analysis.

Woody species and tall-grown species

Since the majority of woody species were observed only once or twice, we based these calculations on the occurrence of woody species in subplots, irrespective of species identity. Because these data contained several zeros, ln(RR) was deemed to be less suited for the analysis, and we instead calculated the difference in occurrence after 11-14 years over the eleven trials between mowing every third year and annual mowing. For comparison, differences between the untreated control and annual mowing were also calculated.

In a similar manner, we calculated species-wise differences between mowing every third year, or the untreated control, and annual mowing. Only species occurring in at least seven of the 22 experimental blocks were considered (fewer occurrences were not considered to appropriately reflect between-site variation), which meant 56 species in total. For these species, we extracted information on plant height from the LEDA traitbase (Kleyer et al. 2008; http://www.uni-oldenburg.de/en/biology/landeco/research/projects/leda/, accessed August 2016), using averages when there was more than one entry for a species. Plant height was then used in a regression analysis to predict the treatment differences.

Results

Effect on species richness and diversity

As expected, there was an overall drop in ln(RR) for the number of species when mowing every third year was compared with annual mowing, but with considerable variation between sites (Figure 2a). In line with this, the two diversity indices also decreased, albeit the Gini-Simpson decreased only very modestly (Figures 2b, 2c), while there were no apparent change in evenness (Figure 2d). Generally, after 11-14 years, the losses were approximately twice the magnitude when comparing the untreated control with mowing every third year (estimates of Overall in Figure 2).

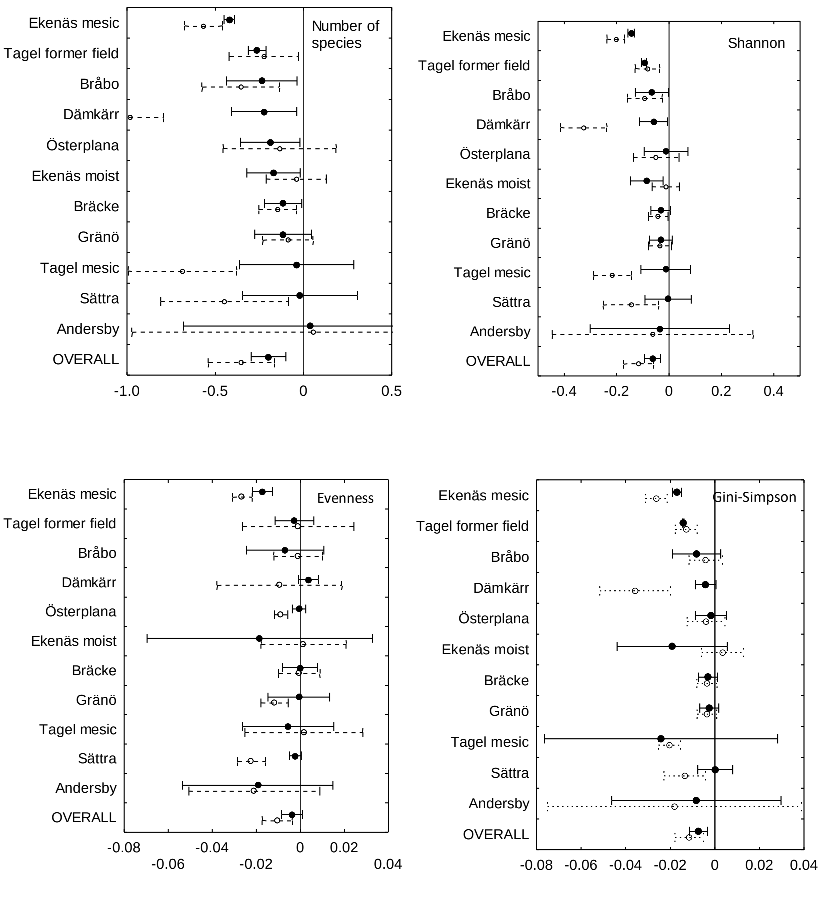
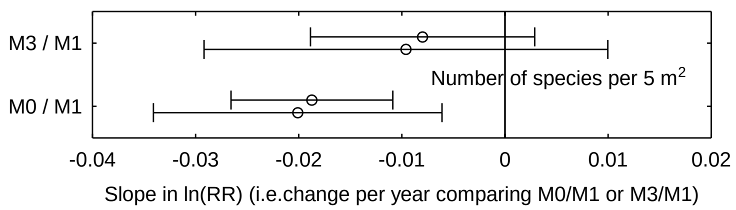


Figure 2. ln(RR) for the 2a) number of species, 2b, 2d) two diversity indices (Shannon and Gini-Simpson) and 2c) index on evenness (Shannon). A negative value means smaller diversity or evenness in mowing every third year (filled symbol with solid bars) or in the untreated control (unfilled symbol with broken bars), compared with annual mowing. The trials had been running for 13 years, eleven years (Ekenäs moist, Ekenäs mesic) or 14 years (Tagel former field). Note the differences in scale on the x-axes.

The rate of change

The estimates of change in ln(RR) for the species number and Shannon diversity index suggested that the loss was slower, approximately half the rate, when mowing every third year compared with the untreated control (Figure 3a, 3b). For the Gini-Simpson and evenness indexes, there were much smaller, but still clear losses, and if anything, the untreated control performed better than mowing every third year (Figure 3c, 3d).

Trial-wise trajectories in ln(RR) for the species number varied greatly among sites: in some trials, there were increases when mowing every third year compared with annual mowing (Figure 3e), and in some trials, there were increases in ln(RR) in the untreated control (Figure 3f).



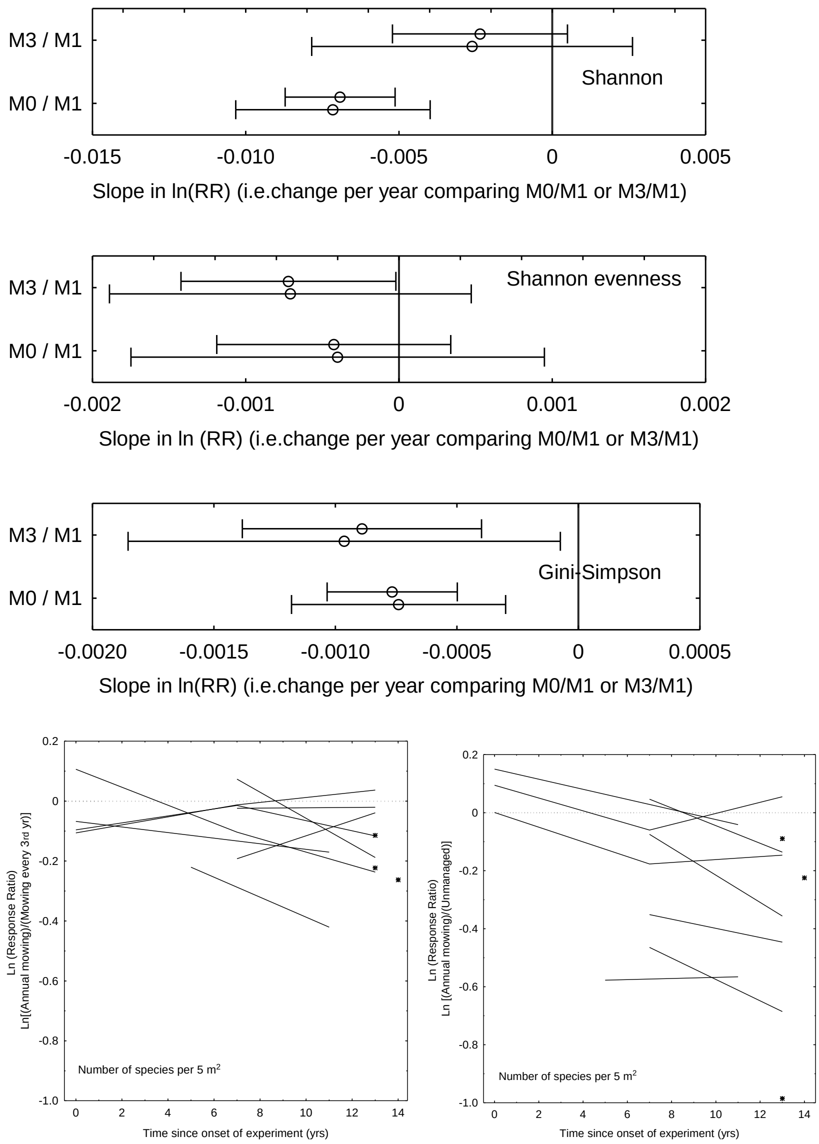


Figure 3. Development of ln(RR) over time. 3a-3d) Averages for the eight trials with comparable data. In the pairs, the bottom value is an average and the top value is a weighted average (weighted based on whether there were two or three time points). Note the differences in scale on the x-axes 3e) ln(RR) for the species number when mowing every third year compared with annually and 3f) untreated control compared with annual mowing. Stars in 3e and 3f indicate trials that only had one time point

using the standard protocol for vegetation sampling. A negative value means that the number of species, biodiversity or evenness is higher in annual mowing than in the alternative treatment.

The site at Sättra, which had been followed for 38 years, showed a clear negative trend over time when mowed every third year in one of the two blocks. In the other block, however, there was no negative trend (Figure 4). In both blocks, the untreated control had lost species over time (Figure 4; no assessment was performed after 38 years because the dense tree canopy that had developed in the plots had recently been thinned to eliminate the shading of adjacent plots).

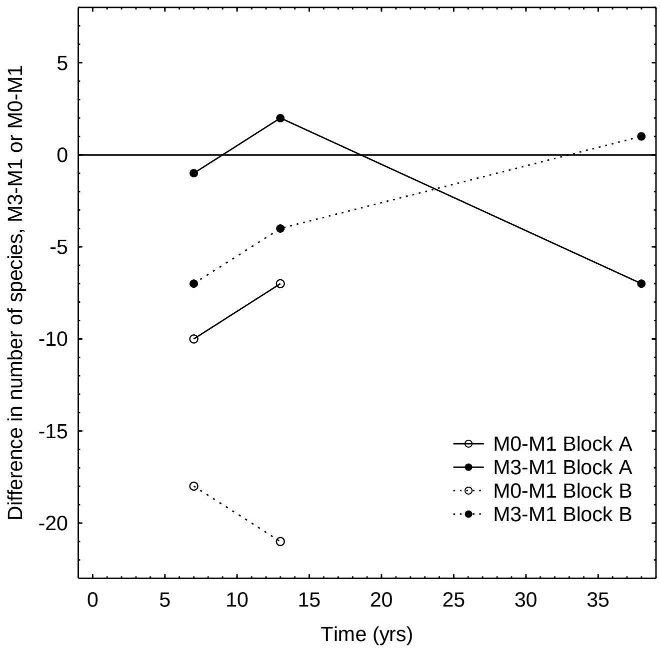


Figure 4. The development of the species number recorded in the long-term trial at Sättra and expressed as the difference between M1 (annual mowing) and M3 (mowing every third year) or M0 (untreated control).

Change in species indicating good or poor management

The odds of finding indicators of good management showed no apparent change over time, nor did the odds of finding indicators or poor management (Figure 5).

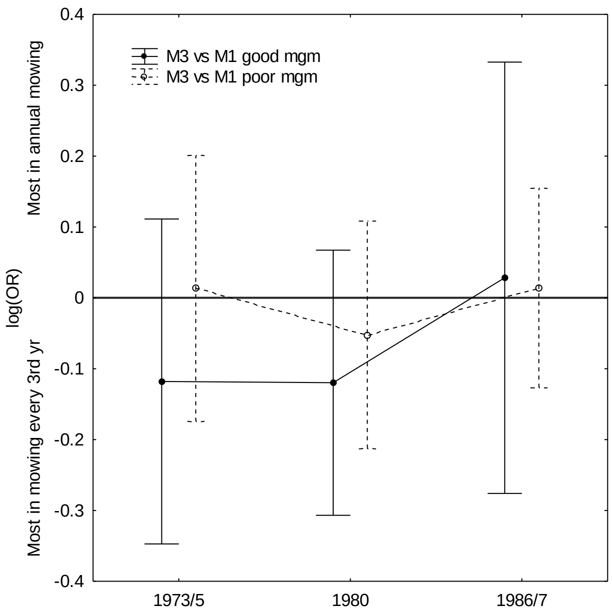


Figure 5. ln(OR) contrasting mowing yearly with every third year for indicators of good management and/or poor management.

Effects on woody and tall-grown species

Species-wise analyses showed that a majority of species occurred less frequently when mowing was performed every third year, and for many of these species, the negative trend was stronger in the untreated control treatment (Figure 6). While there were 15 species that showed a significant decrease with relaxed management, there was only one species group with a significant increase, woody species (Figure 6), hence confirming our hypothesis.

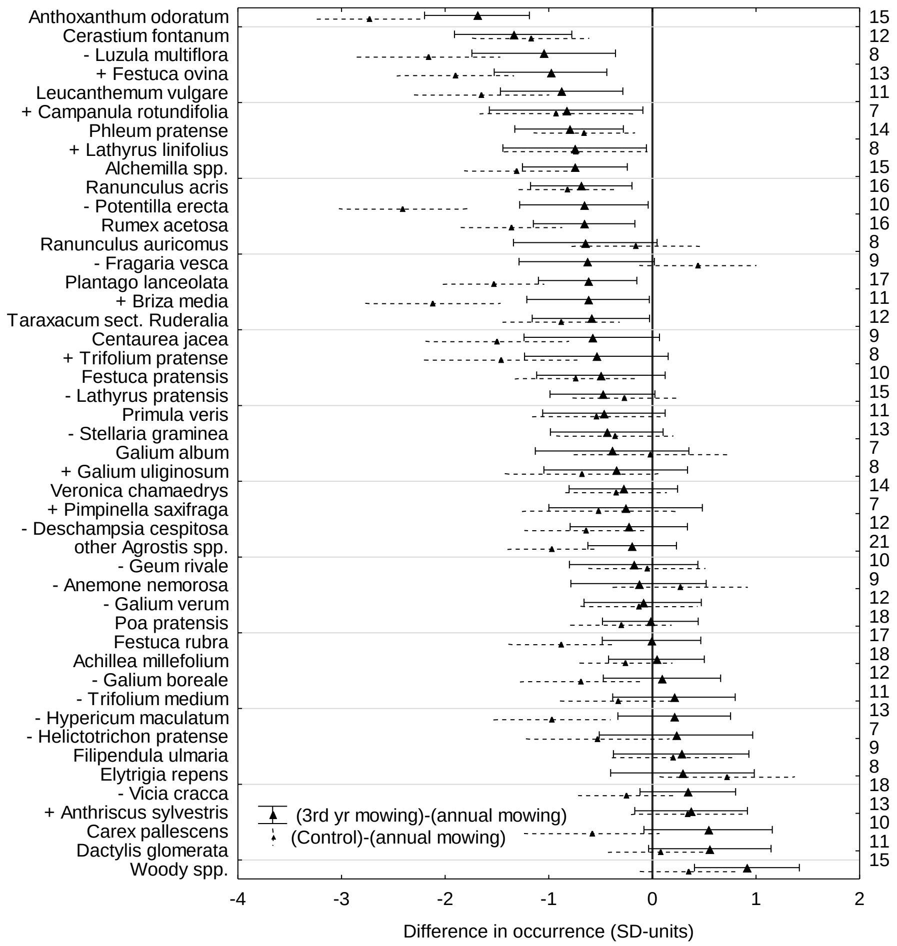


Figure 6. Species-wise responses after 11, 13 or 14 years of differing treatments. A negative value means that a species is decreasing when the mowing frequency decreases compared with annual mowing. The indicator status of species is shown with a + or -, for good and poor management indicators, respectively (based on Ekstam & Forshed 1992). Numbers to the right are the number of blocks that had an occurrence (max 22); species with <7 are not shown.

Comparing the height of species with the values shown in Figure 6, there was a strong positive relationship ($F_{(1,54)}$=16.759; R^2= 0.2368; P=0.000143), i.e., short-grown species mainly became less frequent with relaxed mowing (Figure 7).

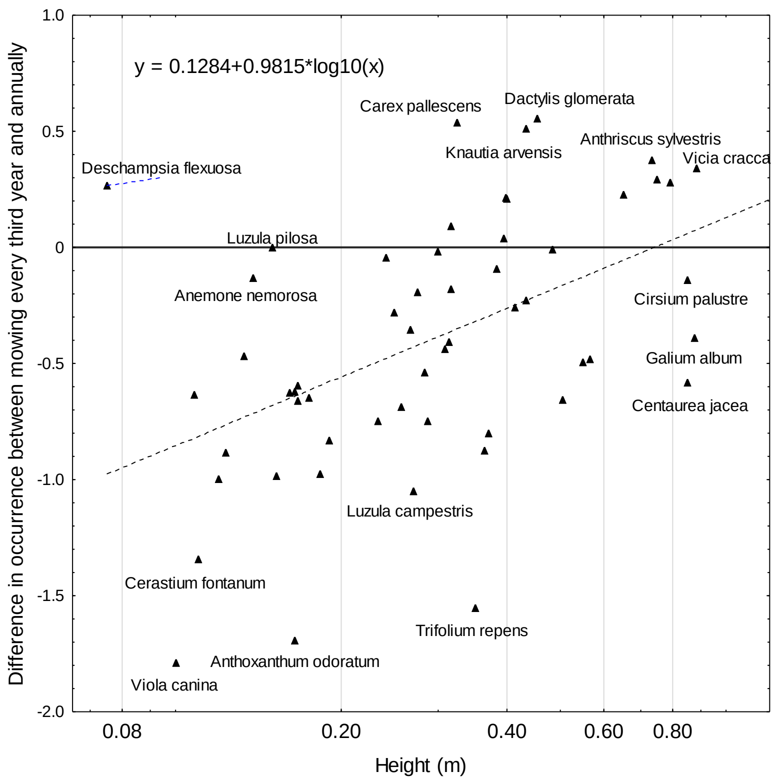


Figure 7. Differences between the two mowing treatments (annual and every third year) per species (N=57) as a function of the species' plant height. The latter was according to the LEDA traitbase, while differences were according to eleven field trials after 11-14 years (see Figure 6). Species names are shown for a selection of the species.

Discussion

By using data from eleven replicated field trials, we showed that relaxed management, in this case mowing every third year, means losing biodiversity attributes in vegetation after 11-14 years compared to annual mowing. This was most clearly seen in data on the species number (Figure 2a) as well as for some of the species (Figure 6) but also in one diversity index (Figure 2b). However, it is important to note that the effect sizes were quite small (an ln(RR) of -0,1 represents e.g. comparing species richness of ten and eleven species), despite the management trials running between 10 and 15 years. That the evenness index (Figure 2d) seemed unaffected suggests that the balance among the species resilient to relaxed management does not change much. More surprising was the lack of treatment effects in the odds of finding indicator species (Figure 5). This particular system of indicator species has previously been shown to be relatively sensitive to management effects in grasslands in the area in which these trials were conducted (Milberg et al. 2014, Tälle et al. 2015). Using indicator species has sometimes been subject to criticism (Siddig et al. 2016), and their usefulness is known to vary in reliability along environmental gradients (e.g., Jansson et al. 2009, Zettler et al. 2013). Hence, some caution is always needed, and validation for a particular scenario is important.

As hypothesized, short-grown species were most likely to decrease when the mowing frequency was relaxed, confirming previous findings (Hejcman et al. 2007, but see also Milberg et al. in prep.) and suggesting one or several possible mechanisms: impaired seed regeneration due to a denser canopy or litter accumulation (e.g., Laborde & Thompson 2013, Loydi et al. 2013, Kladivová & Münzbergová 2016), impaired growth due to litter accumulation (Kelemen et al. 2013), or species being outcompeted by more tall-grown species through shading or root interactions (Lamb et al. 2009, Kiær et al. 2013). From the point of view of management, our finding might suggest the allocation of parts of a grassland – preferably with low grass sward – to an annual mowing regime, while performing more extensive management of more tall-grown areas (e.g., every second year).

As hypothesized, the occurrence of woody species increased with relaxed management. Even if these woody perennials are cut every third year, they seem to have secured enough resources to re-sprout – or have more chances for successful seed establishment – despite the more dense and tall-grown vegetation with relaxed management. Whether these small woody plants are benign or will create an increasing obstacle for mowing remains to be seen. Shoots might be larger when grown from older stumps than from younger stumps or from seed, hence requiring more force to be cut by a scythe or being more likely to remain after the passage of a sickle-bar mower.

By comparing the eleven trials, it is apparent that changes can occur at different rates (Figure 3) and even differ in direction (Figure 3e, Figure 4). Hence, basing conclusions on a single field trial involves a degree of risk of reaching false conclusions (cf Ioannidis 2005). The false discovery rate in science seems to generally be underrated (Button et al. 2013, Nosek et al. 2015), particularly so within ecology, where studies often have low power and there is a reluctance to repeat studies (Parker et al. 2016). Within conservation biology, the total research budget is limited, leading to a data deficiency (Milberg 2014). It was particularly noteworthy that the outcome after 38 years – when a potential treatment effect should have had plenty of time to manifest itself – differed between the two blocks (Figure 4). Therefore, apart from the experimental error that manifested in between-site differences, there was also substantial variation within sites, despite sharing identical land use histories and edafic factors.

The majority of studies on effects of different mowing intensities focus on comparing annual mowing with mowing at a higher intensity (e.g. mowing two times a year). Studies comparing annual mowing with mowing at a lower intensity are more rare, and results are often varying.

While some studies reveal similar results as in the present study, with a more positive effect of the higher mowing frequency (e.g. Köhler et al. 2005), others reveal results in favour of mowing every second year (Gosteli 1996, Bakker et al. 2002, Ameloot et al. 2006). A study by Ryser et al. (1995) revealed similar effect of annual mowing and mowing every second year but a negative effect of mowing every fifth year. Consequently, this might suggest that from a conservation perspective it is sufficient with mowing every second year, while mowing more seldom can have a more negative effect. However, we expect that the most suitable mowing intensity also depends on specific site conditions, e.g. more productive grasslands requiring a higher management intensity to facilitate biomass and nutrient removal (Al-Mufti *et al.* 1977; Oelmann *et al.* 2009)

Rate of change

There is no apparent unit by which to assess biodiversity changes (Milberg 2014), but indices might be more universal than the number of species and so on. In the present study, we chose a palette of different outcomes, several expressed as a response ratio (Figure 3) or odds ratio (Figure 5). For others, the nature of the data forced us to use the difference between the treatment (mowing every third year) and control (annual mowing) (Figure 4, Figure 6). The odds and response ratios are transferable and comparable with other such estimates (e.g., Tälle et al.

2016), while raw treatment differences are context-specific. Furthermore, to aid interpretation, we added corresponding data contrasting the untreated control with annual mowing because this approach hopefully allows more comparisons with other studies.

The loss of biodiversity when turning to mowing every third year was approximately half of that seen in abandonment for the two clearly affected response ratios (number of species, Shannon). The median for the species-wise reductions was 62% of that seen after abandonment, but with large differences among species, confirming previous reports from two of the experimental sites (Hansson and Fogelfors 2000, Wahlman and Milberg 2002), where the outcome of mowing every third year was intermediate between annual mowing and abandonment.

Implications

Mowing every third year resulted in biodiversity losses, and a more realistic frequency might therefore be mowing every second year (Bakker et al. 2002; Köhler et al. 2005) or skipping one year in three. Such relaxed management might not be perfect for grassland plants, but could be be beneficial for many insects, most notably pollinators and butterflies (e.g., Littlewood et al., 2012; van Klink et al., 2015, Milberg et al. 2016).

There are subsidies available for farmers in the EU, e.g., to support the management of semi-natural grasslands. Because these subsidies represent a substantial societal investment, it is important that funds be used in the best possible way. They should function as an incentive for farmers to maintain management that might otherwise not be justified economically. If adjusting subsidies to the actual cost of maintaining mowing, one method is to halve the subsidy if converting to mowing every second year, but such a move is likely to decrease farmers' incentive for maintaining management.

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