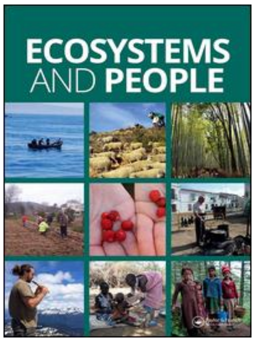

Ecosystems and People

ISSN: (Print) 2639-5916 (Online) Journal homepage: https://www.tandfonline.com/loi/tbsm22

Ceased grazing management changes the ecosystem services of semi-natural grasslands

Line Johansen, Simon Taugourdeau, Knut Anders Hovstad & Sølvi Wehn

To cite this article: Line Johansen, Simon Taugourdeau, Knut Anders Hovstad & Sølvi Wehn (2019) Ceased grazing management changes the ecosystem services of semi-natural grasslands, Ecosystems and People, 15:1, 192-203, DOI: 10.1080/26395916.2019.1644534

To link to this article: https://doi.org/10.1080/26395916.2019.1644534

Published online: 11 Aug 2019.

ECOSYSTEMS AND PEOPLE
2019, VOL. 15, NO. 1, 192–203
https://doi.org/10.1080/26395916.2019.1644534

Taylor & Francis
Taylor & Francis Group

RESEARCH

Ceased grazing management changes the ecosystem services of semi-natural grasslands

Line Johansen, Simon Taugourdeau*, Knut Anders Hovstad and Sølvi Wehn

Department of Landscape and Biodiversity, NIBIO - Norwegian Institute of Bioeconomy Research, Trondheim, Norway

ABSTRACT
Understanding how drivers of change affect ecosystem services (ES) is of great importance. Indicators of ES can be developed based on biophysical measures and be used to investigate the service flow from ecosystems to socio-ecological systems. However, the ES concept is multivariate and the use of normalized composite indicators reduces complexity and facilitates communication between science and policy. The aim of this study is to analyze how land use change affects ES and species richness and how the effects are modified by environmental factors by using composite indicators based on biophysical indicators. Using multivariate and regression analyses, we analyze the effect of grazing management abandonment in semi-natural grasslands in Norway on six ES: nutrient cycling, pollination, forage quality, aesthetics and global and regional climate regulation in addition to species richness along soil and climate gradients. Nutrient cycling, forage quality, regional climate regulation, aesthetics and species richness are larger in managed compared to abandoned grasslands. There are trade-offs among ES as different management strategies provide various ES and these trade-offs vary along environmental gradients. Management policies that aim to conserve ES need to have conservation goals that are context dependent, should recognize ES trade-offs and be adapted to local conditions.

ARTICLE HISTORY
Received 27 August 2018
Accepted 11 July 2019

EDITED BY
Graciela Rusch and
Alexander van Oudenhoven

KEYWORDS
Management strategies; abandonment; grazing; grassland; sheep; biodiversity

Introduction

Semi-natural grasslands are influenced by a long history of extensive agricultural management and are associated with high biodiversity and provides ecosystem service (ES) such as forage for livestock, pollination, biological control, climate regulation and soil conservation (Bullock et al. 2011; Holland et al. 2017; Wehn et al. 2018b). Due to farmland abandonment and other land use changes, areas of semi-natural grasslands are declining and are threatened in Europe (Bullock et al. 2011; Norderhaug and Johansen 2011) but the consequences of these land use changes for ecosystem functioning and ES delivery are not well known.

The framework of ES provides a link between biodiversity, ecosystem functioning/conditions and human society (Millennium Ecosystem Assessment 2005; Maes et al. 2018). Accordingly, drivers of change for one of these components will impact the others. For instance, land use change is a major threat to biodiversity per se (Sala et al. 2000; Pereira et al. 2012) with further consequences for the capacity of ecosystems to deliver ecosystem services and, ultimately, human wellbeing (Millennium Ecosystem Assessment 2005). Understanding the linkages and processes between drivers of land use change and the delivery of ecosystem services is therefore of high importance.

Both biodiversity and ES are influenced by the physical environment (Lavorel et al. 2011; Butterfield and Suding 2013; Trilleras et al. 2015; Wehn et al. 2017). Variation in ecosystem services along environmental gradients is not only due to the direct effect of resource variation but also due to indirect effects caused by shifts in functional composition (Díaz et al. 2007). ES delivery is therefore influenced by interacting effects between land use and environmental factors such as temperature, precipitation and soil fertility (Lamarque et al. 2014).

Normally, ES are assessed and evaluated based on indicators rather than direct measures of ES delivery as these are hard to achieve (Layke et al. 2012; Maes et al. 2018). There are several methods available for assessing ES, i.e. biophysical, social-cultural and monetary techniques (Harrison et al. 2018). Biophysical measures can be used to indicate the provision of a range of ES and a range of studies have assessed the relationship between a biophysical measure, the ecosystem function and the actual service flow that support the socio-ecological system (Potschin-Young et al. 2018; Häyhä et al. 2015; see references in Table 1). However, indicators appropriate for decision-makers need to have credibility, legitimacy, salience and feasibility (Van Oudenhoven et al. 2018). This means that, among other factors, the indicators

CONTACT Line Johansen line.johansen@nibio.no
*Present address: CIRAD UMR SELMET, Montpellier, France

Table 1. Ecosystem services and associated beneficiaries, indicator measures and relationships between indicators and the corresponding ES. The relationship indicates if there is a positive or negative relationship between the ecosystem service and the indicator measure. CWM: Community weighted mean.

ES type	Ecosystem service	Definition	Beneficiaries	Indicator measure	Relationship	Reference
Supporting	Nutrient cycling	Recycling of nutrients	Farmers	Abundance of legumes (%)	+	de Bello et al. 2010
				CWM of leaf dry matter content (mg g^{-1})	+	de Bello et al. 2010, Pakeman 2014
				CWM of specific leaf area (mm^2 mg^{-1})	+	de Bello et al. 2010
Provision	Forage quality	The quality of forage available for grazing (not obtained with fertilizers)	Farmers	CWM of leaf nitrogen content (mg g^{-1})	+	de Bello et al. 2010, Lavorel et al. 2011
				Cover of graminoids	+	Lavorel et al. 2011
				Cover of herbs		
Cultural	Aesthetics	The appreciation of the site and the landscape	Public	Herb grass ratio (%)	+ +	de Bello et al. 2010, Ford et al. 2012
				Number of flower colors		de Bello et al. 2010, Ford et al. 2012
				Canopy cover (%)	-	Vinge and Flø 2015, Wehn et al. 2018a
				Cover of shrub layer (%)	-	Vinge and Flø 2015, Wehn et al. 2018a
Regulation	Global climate regulation	Carbon storage in plants and soil	Environmental manager, public	Cover of shrubs and trees trunks	+	de Bello et al. 2010
				Loss-on-ignition	-	Maskell et al. 2013
	Regional climate regulation	Albedo	Environmental manager	Canopy cover (%)	-	Lutz et al. 2016
				Cover of shrub layer (%)		
	Pollination	Available food plants for wild pollinators	Environmental managers	Abundance of food plant species for butterflies (%)	+	Maskell et al. 2013
				Abundance of food plant species for Hymenoptera species (%)	+	Maskell et al. 2013

should be understandable and relevant, supported by scientific literature, and developed considering the data available.

A single indicator can simultaneously cover multiple ES and a single ES can be reflected by several indicators (de Bello et al. 2010; Harrison et al. 2014). For example, abundance of legumes, leaf dry matter content and specific leaf area are all characteristics that define the process of nutrient recycling (de Bello et al. 2010). Hector and Bagchi (2007) argue that the overall ecosystem function found in ecosystem assessments depends on the number of processes analyzed since each process is affected by a different set of species. Wehn et al. (2018b) demonstrate that various processes underlying one or several ES show different response to change. Both studies illustrate the benefit of using multiple indicators in the assessment of individual ES. To evaluate the overall impact of an ES, all the available indicators reflecting the ES have to be analyzed together (Burkhard et al. 2012). In order to perform a value proposition of an ES based on several indicators, it is possible to develop a composite indicator that combine a few or several indicators into a single numerical value.

In this study, we use biophysical indicators of several ES to develop composite ES indicators. This includes nutrient cycling, forage quality, aesthetics, global and regional climate regulation and pollination. Nutrient cycling support ES that underpins all other ES in semi-natural grasslands (Lavelle et al. 2005; Mace et al. 2012) as well as in other ecosystems. Fodder production is an essential provisioning ES in semi-natural grasslands and quality of fodder is of high importance for life stock and food production (Bullock et al. 2011). In addition, semi-natural grasslands are essential habitats for pollinators because of high diversity in both floral resources and nesting sites (IPBES 2016) and there is a positive link between abundance of pollinator food plants and the potential abundance of pollinators (Potts et al. 2003; Biesmeijer et al. 2006; Holzschuh et al. 2007). Other important regulation ES are climate regulation at both global and regional scale. Ecosystem functions that influence the concentration of greenhouse gases in the atmosphere contribute to global climate regulation (Haines-Young and Potschin 2018). An important example of this is carbon sequestration in plants and soils reducing greenhouse gases in the atmosphere and thus global warming. However, other studies have found that in situations with expanding forests the cooling effect of carbon sequestration can be much lower than the warming effect caused by the lower surface albedo of forests as compared to grasslands and other open habitats (Betts 2000; de Wit et al. 2014). The surface albedo of [the] vegetation is an important ES for regional climate regulation (Lutz et al. 2016).

Cover of forest and shrubs, in addition to flower diversity, are also related to the aesthetical appreciation of the landscape and contribute to human well-being (de Bello et al. 2010; Ford et al. 2012; Vinge and Flø 2015). The public considers grasslands with high diversity of herbs and flower colors to be more aesthetically appealing (de Bello et al. 2010; Ford et al. 2012). In several areas of Europe, there is widespread encroachment of the cultural landscape due to the abandonment of agricultural farmland (Emanuelsson 2009; Wehn 2009) and some therefore find abandonment to reduce the scenic beauty (Schirpke et al. 2013). In Norway, the open landscape is appreciated by the population as part of their cultural heritage (Vinge and Flø 2015; Wehn et al. 2018a). In this study, we also include species richness as conservation of biodiversity is an important objective for nature management and land use policy and goal in semi-natural grasslands in addition to ES (Bullock et al. 2011; Wehn et al. 2018a).

The overall aim of this study is to analyze how the abandonment of grazing management in semi-natural grasslands effects the ecosystem. We use the ES framework because land use change is known to affect ecosystems condition and services (Millennium Ecosystem Assessment 2005). In addition, the ES framework is effective for communicating the multiple effects of land use change to policy makers and managers. We use composite indicators based on biophysical measures and analyze how land use change affects ES and species richness and how land use interacts with environmental factors, i.e. climate and soil. We compare ES provision and species richness in managed and abandoned semi-natural grasslands and provide empirical-based knowledge to inform agri-environmental policies. We analyze the effect of abandonment of grazing management in semi-natural grasslands in Norway on species richness and six different ES: nutrient cycling, pollination, forage quality, aesthetics, and global and regional climate regulation along soil and climate gradients.

We use composite indicators to perform a value proposition of each of the ES based on the biophysical measures: abundance of legumes, leaf dry matter content, specific leaf area, leaf nitrogen content, cover of graminoids, herb/grass ratio, number of flower colors, canopy cover, cover of shrub layer, cover of shrubs and tree trunks, and abundance of food plants for butterflies and Hymenoptera species.

The ES assessed are relevant for semi-natural grassland ecosystems (Bullock et al. 2011) and benefit environmental managers, farmers, and the society in general (see Table 1). The biophysical measures used are well documented in scientific literature as good indicators of one or several ES (see Table 1). They are intuitive and relevant for beneficiaries, and easily accessible. Hence, the indicators are appropriate to be used (following the suggestions in Van Oudenhoven et al. 2018) to develop composite indicators that will inform decision-makers about trade-offs between ES supply due to different management strategies.

Linked to the case study we ask the specific research questions: 1) How does the amount of ES provision and species richness differ between managed and abandoned semi-natural grasslands? 2) Are effects of land use on the amount of ES provision and species richness influenced by climate and soil?

Material and methods

Study areas and design

The case study comprises agricultural landscapes in Norway that include a mosaic of forests, arable lands, pastures and meadows ranging from monocultures to species-rich semi-natural grasslands. In Norway, semi-natural grasslands are red-listed and are ecosystems of the highest conservation value (Norderhaug and Johansen 2011). Semi-natural grasslands have high biodiversity at several trophic levels but are red-listed mainly because they are decreasing due to abandonment of management. Environmental schemes have been implemented to improve their ecological quality (Direktoratet for Naturforvaltning 2009) and these schemes operate at a local scale and target each grassland separately. In this project, we have selected semi-natural grasslands with high biodiversity that are managed adjacent to abandoned area.

The study areas are situated in west (5°33′50″ – 7° 21′18″ E; 61°11′51″ – 61°33′50″ N) and central Norway (10°16′35″ – 11°16′52″ E; 63°09′24″ – 63°32′32″ N) within the boreal zone (Figure 1). Both study areas include seven study sites (n_{sites} = 14) established in 2012 (west) and 2013 (central). The study was carried out in the year of establishment and not repeated. A ‘space for time’ design approach is used; all sites include one plot in a managed semi-natural grassland and one plot in an abandoned semi-natural grassland (Figure 1). Managed semi-natural grasslands (1–10.5 ha) were all fenced pastures used for sheep grazing in spring and/or autumn. Semi-natural grasslands were not grazed by other domestic animals.

At each site, two plots of equal size were located on either side of a fence and the locations were approximately perpendicular to the contour lines of the terrain. The maximum parallel length of the plots along the fence was 100 m and each of the plots extended a maximum of 50 m perpendicular from the fence. The size of the plots varied due to varying sizes of the extensively managed semi-natural grasslands and ranged from 0.08 to 0.5 ha (mean size = 0.26 ha). Each plot (n_{plot} = 28) comprises four subplots ($n_{subplot}$ = 112, size = 4 m^2). Half of the subplots were randomly located in the lower part of the plot and the other half in the upper

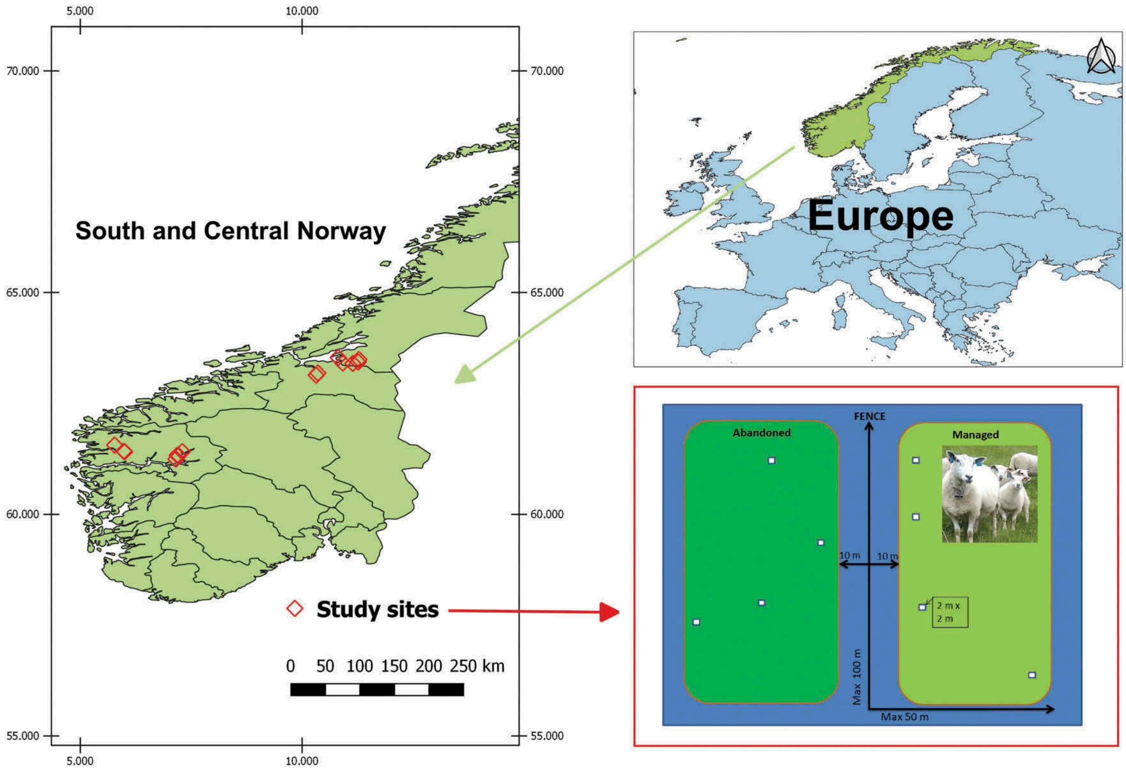

Figure 1. The study areas in west and central Norway includes 14 study sites in total. Each study site includes four subplots in managed and abandoned semi-natural grassland separated by a fence. Source: Natural Earth QGIS 3.0.3. Coordinate system: WGS84. Natural Earth QGIS 3.0.3. Coordinate system: WGS84.

part (Figure 1). In the managed semi-natural grasslands, subplots were located more than 10 m from the fence to avoid edge effects.

The grazing management was not experimental but a result of the farmers practice. Information about the grazing management was gained through semi-structured interviews with 12 farmers. The interviews revealed that all farms had the same farming system, which is typical for Norwegian sheep husbandry, with extensive grazing in outfields (alpine and forest) in the summer and indoor feeding during winter. In spring (May), sheep and lambs graze for 2–6 weeks in enclosures near the farm before being herded to the outfields after snowmelt. In autumn (September), the animals are herded back to the enclosures where they again graze for 2–6 weeks before winter. Management of all semi-natural grasslands in the study system included relatively low-intensity land use practices; no ploughing, reseeding, or artificial fertilizing during the last decade(s). In some of the managed semi-natural grasslands, farmers increased pasture quality by clearing shrubs and trees. The grazing pressure varied each year within the enclosures due to differences in timing of lambing, snow cover, availability of forage in the outfield and the need to use the enclosures as preventives for carnivore loss. Hence, information on exact grazing pressure was not available. All semi-natural grasslands had an even field layer, no erosion or open soils, were regularly grazed and not overgrazed. We therefore assumed that grazing pressure during the last decade was comparable among sites. The abandoned areas were not grazed and were at different successional stages toward forest. Even though the time since abandonment ranged from five to 70 years, time since abandonment had no effect on species richness and minimal effect on ES indicators (Wehn et al. 2018b).

Environmental data

For each site, values of bioclimatic variables were extracted from the WorldClim database (WorldClim 2015): mean annual temperature, annual precipitation, mean temperature of warmest quarter, precipitation of warmest quarter and elevation. Per subplot, we randomly collected five soil samples of 500 ml (0–10 cm below the litter layer) with an auger (22 mm diameter). These were mixed to give one bulk soil sample per subplot. The soil samples were analyzed by Eurofins Environment Testing Norway AS. The measured variables were pH, available phosphorus, potassium, magnesium, calcium and sodium in addition to bulk density and loss-on-ignition (a measure of organic matter). Available phosphorus, potassium, magnesium and sodium and organic matter are measures of soil fertility and available calcium and pH indicate whether the soil is acidic or calcareous. Values below the detection threshold were assigned a zero value.

To reduce the number of bioclimatic and soil variables in the analysis, and to deal with collinearity, principal component analyses (PCA) (Janžekovič and Novak 2012) were performed on climatic and soil variables separately. By performing this multivariate analysis, we combined i) the two bioclimatic variables annual precipitation and precipitation of warmest quarter in the climate principal component 1 (Climate PC1), ii) the three bioclimatic variables mean annual temperature, annual mean temperature of warmest quarter, and elevation in the climate principal component 2 (Climate PC2), iii) the six soil variables available phosphorus, potassium, magnesium, sodium, organic matter and bulk density in the soil principal component 1 (Soil PC1), and iv) the two soil variables pH and calcium in the soil principal component 2 (Soil PC2). By such, we could use four environmental variables in the following analyses. PCA was also used to rule out correlation between the two first principal components (PCs) of both climatic and soil analyses. The four PCs were therefore used as descriptors of climate and soil conditions in the following analyses. PCAs were performed using the FactoMine R package (Lê et al. 2008). Climate PC1 was positively related with levels of precipitation and Climate PC2 negatively with temperature. Soil PC1 and PC2 were positively related with soil fertility and pH, respectively.

Construction of composite indicators

We selected six ES delivered from semi-natural grasslands relevant for farmers, the public and environmental managers and identified 14 indicators that relate to these ES based on literature (Table 1). All indicators and ES are not necessary relevant for all stakeholders (Plantureux et al. 2016). ES related to food production is most relevant (nutrient cycling, forage quality) for farmers (Bullock et al. 2011) and landscape aesthetics for the public appreciating the landscape (Ford et al. 2012; Vinge and Flø 2015; Wehn et al. 2018a). ES climate regulation and pollination of wild species are more relevant for environmental management in addition to species richness. The ES were assessed at grassland scale and it was therefore appropriate to use mainly botanical measures (vegetation structure, plant species composition, functional diversity) as indicators (Plantureux et al. 2016). Table 1 gives an overview off all indicator measures for each ES, the relationship between each indicator and ES, and the related literature.

To obtain botanical data for these indicator measures we registered all vascular plant species in each subplot. We estimated cover of all species and trunks of shrub and trees in eight categories (0, >0–1/64, 1/64–1/32, 1/32–1/16, 1/16–1/8, 1/8–1/4, 1/4–1/2, 1/2–1). In addition, percent cover (0–100%) of canopy and shrub layer was registered in each subplot.

The botanical survey data were linked to functional traits for the majority of ES indicators (Table 1). For all species, data on leaf dry matter content, specific leaf area and growth form (legumes, graminoids, herbs) were extracted from the LEDA database (Kleyer et al. 2008). Leaf nitrogen content and the growth forms shrubs and tree for all species was extracted from the TRY database (Kattge et al. 2011) and flower colors from the Norwegian Flora (Lid and Lid 2005). Data on food plants for pollinators were extracted from the Biological Records Centre's database of insects and their food plants (BRC 2015).

For a few of the species registered in the survey, trait data were missing in the LEDA database. The percentages of missing data of the functional traits leaf dry matter content, specific leaf area and leaf nitrogen content were 8%, 3% and 27%, respectively. We replaced these missing data with values estimated based on the other traits with values present for that particular species using imputation methods as described by Taugourdeau et al. (2014). Then, we calculated measures based on 'community-aggregated trait values' as suggested by Garnier et al. (2004), which describe the functional identities of a community (community weighted means (CWMs)) using the FD package in R (Laliberté and Legendre 2010).

The measures of all ES indicators based on cover of the characteristics (legumes, graminoids, shrub and tree trunks, food plants for butterflies, food plants for Hymenoptera and herbs; see Table 1) were calculated as the sum of the centers of the eight registered cover categories for each species containing the characteristic in the subplots.

All indicators that reflect an ES were combined into a composite indicator by using a multi-criteria assessment (TATALE) approach that enables normalization and aggregation of indicators (Taugourdeau and Messad 2017). First, the measures for all selected ES indicators and species richness were normalized by a linear transformation to gain values between zero and one. The relationship between each indicator and the corresponding ES (Table 1) was decided based on published literature (de Bello et al. 2010; Lavorel et al. 2011; Duru et al. 2012; Ford et al. 2012; Schirpke et al. 2013; Harrison et al. 2014; Pakeman 2014; Vinge and Flø 2015). Negative transformations were used for loss-on-ignition (soil carbon storage) and canopy and shrub layer cover (both when indicating aesthetics and regional climate regulation), while positive transformations were used for the remaining indicator data. All indicators relating to each ES (see Table 1) were then aggregated to a composite indicator for the seven ES using arithmetic mean values of the notations except for pollination where weighted means were used. In Norway, Hymenoptera species are thought to contribute more to pollination than butterflies (Totland et al. 2013) and we therefore weighted the abundance of food plant species for Hymenoptera species to account for 60% of the composite indicator value and abundance of food plant species for butterflies to account for 40%.

Model application and statistical analyses

Linear mixed models (LMMs) were used to estimate the influence of land use (sheep grazing management or abandonment) on the delivery of multiple ES and species richness using the lmer4 package (Bates et al. 2015). In general, the plot size did not influence the difference in ES amount in abandoned and managed plots. This was evaluated by visual interpretation of linear regression lines of the difference in ES amount. Hence, the plot size was not included in the models. The seven composite indicators were included as dependent variables, site as a random factor and management and the four principal components (PCs; related to temperature, precipitation, soil fertility and soil pH) as fixed explanatory factors in the LMMs. First, the effect of land use on each composite indicator was investigated and then, interactions between land use and climate or soil were tested (represented by the four PCs). Models were made sequentially starting with a null model. Model fit was assessed by a chi-square test on the log likelihood values (α = 0.05). All data analyses were carried out using the R 3.1.1 software (R Core Team 2015).

Results

Land use had a significant effect on all ES except pollination (Table 2). Nutrient cycling, forage quality, regional climate regulation, aesthetics and species richness all indicated higher provision in managed semi-natural grasslands as compared to abandoned grasslands (Figure 2). Global climate regulation, on the other hand, was lower in managed semi-natural grasslands than in abandoned grasslands. Significance for bold values are 0.05.

The effects of land use on composite indicators varied along the climatic and soil gradients (Table 2; Figure 3). Interactions between land use change and climatic gradients were significant for several of the ES (nutrient cycling, regional climate regulation and aesthetics) and species richness. Interactions with soil gradients were only noted for the global climate regulation ES and species richness (Figure 3). The effect of land use on species richness was most pronounced in base-rich soils (the right end of the soil PC2) and soil fertility (represented by the soil PC1) caused varying responses of changed land use on global climate regulation. The effect of land use change on species richness, regional climate regulation, as well as aesthetics, varied along the temperature gradient (represented by the climate PC2). In warmer climates (the left end of PC2), managed semi-natural grasslands provided more species, regional climate regulation and aesthetics as compared to abandoned semi-natural grasslands. In colder climates (the right end of PC2), the effects of land use change were less clear. Precipitation (represented by the climate PC1) modified the effect of land use on nutrient cycling. In abandoned plots, there was a positive relationship with climate PC1 (precipitation) while there was a negative relationship in managed plots.

Discussion

Based on a case study where soil and botanical data were extracted from the model semi-natural grassland ecosystem, we have shown that abandonment of semi-natural grasslands can alter the delivery of multiple ecosystem services (ES) in addition to species richness. Provision of several of the assessed ES, was higher in semi-natural grasslands with management as compared to abandoned grasslands. Hence, the case study shows that land abandonment and encroachment with shrubs and forest, as reported in many regions across Europe (Pereira et al. 2005; Ford et al. 2012; Fontana et al. 2014), demote ES delivery from open semi-natural grasslands. If the practice of exploiting grazing resources in semi-natural grasslands is abandoned, the landscapes reduce in quality for agricultural production as well as in biological

Table 2. Effects of land use and soil PCI (fertility), soil PC2 (pH), climate PC1 (precipitation) and climate PC2 (temperature) on the delivery of ecosystem services and species richness. Test statistics from log-likelihood chi-square tests of linear mixed models. Land use: low-intensity sheep grazing management or abandonment. x: interaction effect.

	Land use		Soil PC1 (fertility) x Land use		Soil PC2 (pH) x Land use		Climate PC1 (precipitation) x Land use		Climate PC2 (temperature) x Land use	
Response	χ2	p	χ2	p	χ2	p	χ2	p	χ2	p
Species richness	25.851	**<0.001**	1.133	0.567	27.081	**< 0.001**	0.041	0.979	17.693	**< 0.001**
Nutrient cycling	5.902	**0.015**	1.349	0.509	1.347	0.510	7.877	**0.019**	3.798	0.150
Forage quality	9.65	**0.001**	3.866	0.144	2.134	0.344	0.848	0.654	4.230	0.121
Aesthetics	56.4	**<0.001**	0.596	0.742	1.892	0.388	0.476	0.788	20.127	**<0.001**
Global climate regulation	4.39	**0.036**	88.657	**<0.001**	1.406	0.495	2.578	0.276	2.400	0.301
Regional climate regulation	55.59	**<0.001**	1.124	0.570	0.189	0.910	0.0316	0.984	10.086	**<0.001**
Pollination	1.075	0.300	5.552	0.06	1.033	0.597	2.089	0.352	4.295	0.117

Note. Significance for bold values are 0.05

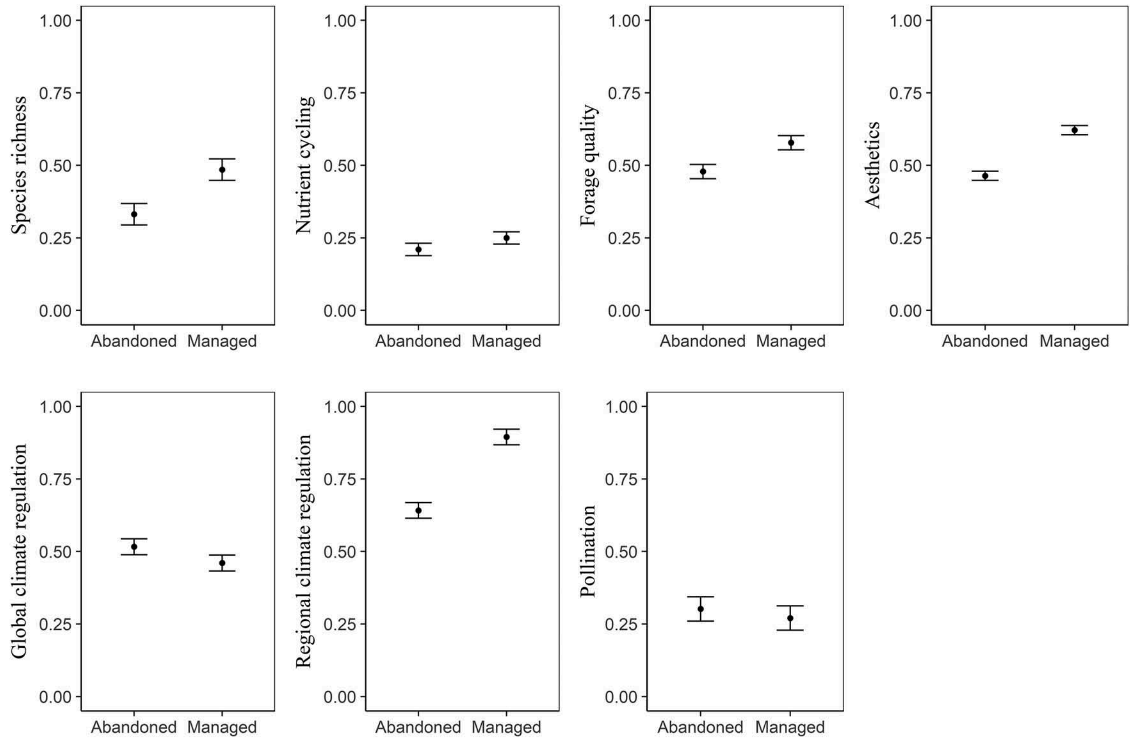

Figure 2. Estimated effects and standard errors from linear mixed models (LMM) for ecosystem services (ES) and species richness in managed and abandoned semi-natural grasslands. The ES values are normalized notations between 0 and 1 and based on composite indicators.

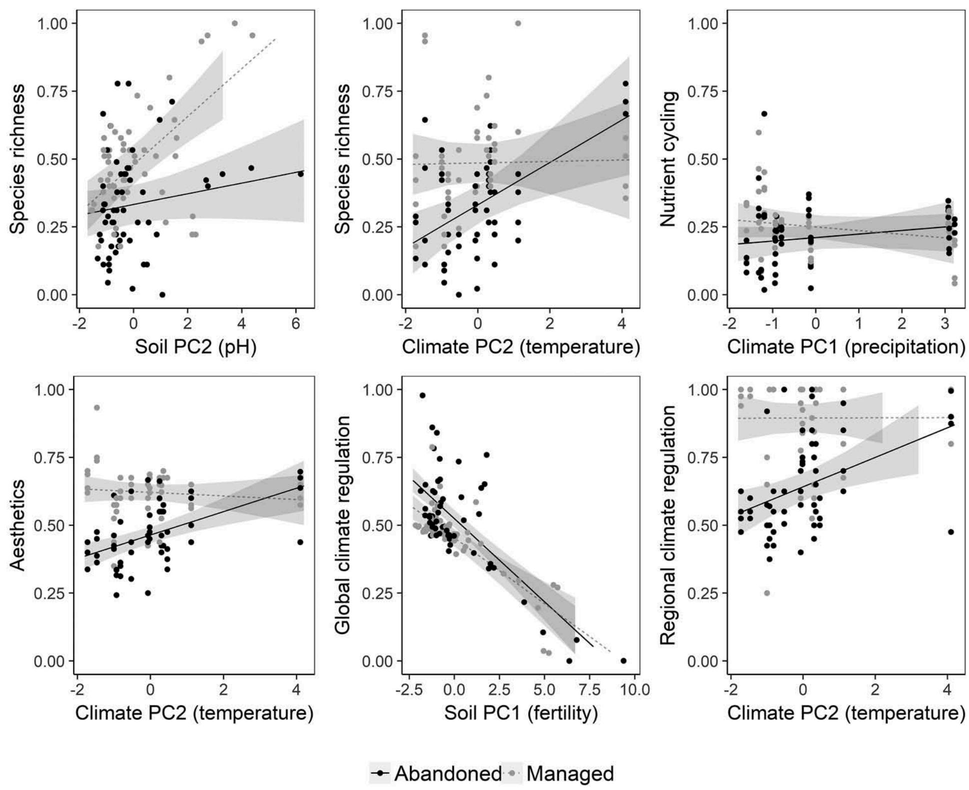

Figure 3. Observed (points) and estimated effects (lines) and 95% confidence intervals for ecosystem services (ES) and species richness in managed and abandoned semi-natural grasslands along environmental gradients (only those shown to have impact on the response of land use; see Table 2). The ES values are normalized composite indicators between 0 and 1. The environmental data is based on the first two principal component (PC) axes from principal component analyses. The x-axes are positively related to precipitation (Climate PC1), soil fertility (Soil PC1) and soil pH (Soil PC2) and negatively related to temperature (Climate PC2).

and ES conservation values. In addition to reducing ES delivery, grazing abandonment can also reduce functional diversity and hence decrease ecological processes (Peco et al. 2012).

The European Union aims to halt the loss of both biodiversity and ecosystem services (Council of the European Union 2010). In our study, species richness was much higher in managed semi-natural grasslands compared to abandoned grasslands and this result corresponds with the positive association between ES amount in semi-natural grasslands and biodiversity reported by others (Bullock et al. 2011). The importance of semi-natural grasslands for species richness and diversity in agricultural landscapes (Billeter et al. 2008) together with the positive relationship between species richness and several ES justifies the strong focus on semi-natural grassland conservation and implementation of agri-environmental schemes aimed at preserving biodiversity in farmland throughout Europe (European Commission Directorate General for Agriculture and Rural Development 2005). However, agri-environmental policy makers and environmental and agricultural agencies have to deal with the duality of monofunctionality in productive areas of high agricultural production and/or multifunctionality in traditionally managed agricultural landscapes of high biodiversity (Tscharntke et al. 2012). By using composite indicators of ES delivery, we show that it is possible to meet the multiple goals of delivery of agricultural production (forage quality) and biological conservation (species richness) via extensive and multifunctional agricultural practices. In addition, several other ecosystem services (such as nutrient cycling, temperature regulation and aesthetics) are provided. Norwegian sheep husbandry that uses semi-natural grasslands for grazing in spring and autumn is a good example of how agricultural production can be combined with conserving biodiversity and ES delivery. To effectively target the optimal delivery of both ES and biodiversity conservation in management plans, fine-scale assessments of biodiversity and ES are needed as well as of potential environmental impacts within the targeted areas. Generally, ES are assessed at relatively large geographic scales such as units ranging from municipalities to regions (Queiroz et al. 2015) or grid sizes of, for example, 1 km^2 (Crouzat et al. 2015). Most actions implemented by agri-environmental schemes are, however, targeted locally and within fine-scaled vegetation units such as the Natura 2000 habitats (European Commission 2016), the selected nature types for Norway (Bugge 2011), or the Swiss ecological compensation areas (ECA) (Schmid et al. 2000).

In our case study, most of the assessed ES were higher in managed compared to abandoned semi-natural grasslands. It should, however, also be recognized that ES delivered from abandoned grasslands, such as global climate regulation (as shown in the case study), have some value. Further, the high-profile pollination ES is important for global food production and is assumed to be high in open semi-natural habitats (IPBES 2016). Our case study indicates, however, that abandonment of grazing does not reduce the quantity of food plants for pollinators at a local scale. A study from Wales shows the same response (Ford et al. 2012). Semi-natural grasslands are typically rich in flowers and therefore important habitats for pollinators but these insects also use habitats associated with abandoned grasslands such as forest edges. For instance, pollen-producing tree species (e.g. *Salix caprea*), which establish in abandoned areas, serve as important food sources for pollinators (Totland et al. 2013). It is important, to recognize that the results shown in this case study, might be due to a time-lagged species turnover after abandonment which can allow plants that are attractive to the pollinators to exist for decades in the abandoned areas (Johansen et al. 2016). The time-lag may result in the presence of the same species in both managed and abandoned semi-natural grasslands and therefore similar food sources for pollinators although this is not explored in detail here.

Livestock grazing affects multiple ES and can also create trade-offs and synergies between different ES (Davidson et al. 2017). Our case study shows that different land use approaches ultimately deliver different amounts of ES. Appropriate management strategies therefore depend on the context and conservation goals at the landscape or field level. There is not always a positive relationship between ES indicators and the biodiversity elements that are the goals for conservation actions (Wehn et al. 2018b) and this case study show that different land use management strategies foster different ES. Ekroos et al. (2014) argue that agri-environmental schemes must include more explicit goals regarding either biodiversity conservation or provision of ecosystem services. Further, the effects of agri-environmental measures depend on regional and local conditions (Kleijn et al. 2011). Our results demonstrate that the effects of management on the provision of ES vary along environmental gradients which has also been highlighted in other studies from Europe (Peco et al. 2017). The interactions between management and environmental conditions such as soil and climate, require that local conditions and spatial scale are considered in assessments of ecosystem services and how these are influenced by management.

The concept of ES is multivariate and combine ecological, biophysical and social values and include both non-material and material services (Millennium Ecosystem Assessment 2005). This makes indicators incommensurable due to multiple units of measure. Composite indicators overcome this by combining several individual indicators into a single numerical value (Burkhard et al. 2012) and make it possible to compare ES in the investigated system. This simplification facilitates communication between policy and science which enables an integration of ES into decision-making and land use management (Burkhard et al. 2012; Alam et al. 2016). Indicators must, however, be relevant to stakeholders (farmers, public, managers, etc.) and should be adapted to the appropriate scale (grassland, farm, region) of the study (Plantureux et al. 2016). The scale used for mapping ES should be based on the questions asked, the type of analysis needed and the level of detail required to answer the questions (Raudsepp-Hearne and Peterson 2016). By using composite indicators based on empirical data, we have assessed and compared several ecosystem services (ES) provided by an ecosystem. We have also gathered empirical-based knowledge on how different management actions can affect ES delivery. This knowledge can help policy makers to further develop agri-environmental schemes that manage ES.

Our results demonstrates that species richness and provisioning of ES of semi-natural grasslands are highly influenced by management. By using composite indicators, we found that the provisioning of the ES nutrient cycling, forage quality and aesthetics are higher in managed as compared to abandoned grasslands. Species richness was also significantly higher in grasslands with management. The effects of management on species richness and ES were however dependent on local environmental conditions related to climate and soil. Policy and management measures that aim to sustain biodiversity and ES of semi-natural grasslands should recognize the ES trade-offs related to different types of management and develop goals for conservation and management that depend on context and are adapted to local conditions.

Acknowledgments

We thank S. Aune, S. Nordal Grenne, P. Thorvaldsen, L. G. Velle and P. Vesterbukt for help during fieldwork, and M. V. Henriksen for proofreading and commenting on the manuscript. We are grateful to anonymous reviewers and the editors for their suggestions and useful criticisms, which significantly improved the paper.

Disclosure statement

No potential conflict of interest was reported by the authors.

Funding

This study was funded by The Research Council of Norway, via the project no 208036/010.

ORCID

Line Johansen http://orcid.org/0000-0002-3904-070X

References

Alam M, Dupras J, Messier C. 2016. A framework towards a composite indicator for Urban ecosystem services. Ecol Indic. 60:38–44. doi:10.1016/j.ecolind.2015.05.035.

Bates D, Maechler M, Bolker B, Walker S. 2015. Fitting linear mixed-effects models using lme4. J Stat Softw. 67 (1):1–48. doi:10.18637/jss.v067.i01.

Betts RA. 2000. Offset of the potential carbon sink from boreal forestation by decreases in surface albedo. Nature. 408(6809):187–190. doi:10.1038/35041545.

Biesmeijer JC, Roberts SP, Reemer M, Ohlemüller R, Edwards M, Peeters T, Schaffers A, Potts SG, Kleukers R, Thomas C. 2006. Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. Science. 313(5785):351–354. doi:10.1126/science.1127863.

Billeter R, Liira J, Bailey D, Bugter R, Arens P, Augenstein I, Aviron S, Baudry J, Bukacek R, Burel F. 2008. Indicators for biodiversity in agricultural landscapes: a pan-European study. J Appl Ecol. 45(1):141–150. doi:10.1111/j.1365-2664.2007.01393.x.

BRC. 2015. Biological Records Centre's database of insects and their food plants [accessed 2015 December 08]: http://www.brc.ac.uk/dbif

Bugge HC. 2011. Environmental law in Norway. Alphen aan den Rijn: Kluwer Law International.

Bullock JM, Jefferson RG, Blackstock TH, Pakeman RJ, Emmett BA, Pywell RJ, Grime JP, Silvertown J. 2011. Semi-natural grasslands. Technical report: the UK National Ecosystem Assessment. Cambridge (UK): UNEP-WCMC; p. 161–196.

Burkhard B, Kroll F, Nedkov S, Müller F. 2012. Mapping ecosystem service supply, demand and budgets. Ecol Indic. 21:17–29. doi:10.1016/j.ecolind.2011.06.019.

Butterfield BJ, Suding KN. 2013. Single-trait functional indices outperform multi-trait indices in linking environmental gradients and ecosystem services in a complex landscape. J Ecol. 101(1):9–17. doi:10.1111/1365-2745.12013.

Core Team R. 2015. R: A language and environment for statistical computing. Vienna (Austria): R Foundation for Statistical Computing.

Council of the European Union. 2010. 7536/10 Biodiversity: Post −2010

Crouzat E, Mouchet M, Turkelboom F, Byczek C, Meersmans J, Berger F, Verkerk PJ, Lavorel S. 2015. Assessing bundles of ecosystem services from regional to landscape scale: insights from the French Alps. J Appl Ecol. 52(5):1145–1155. doi:10.1111/1365-2664.12502.

Davidson KE, Fowler MS, Skov MW, Doerr SH, Beaumont N, Griffin JN. 2017. Livestock grazing alters multiple ecosystem properties and services in salt marshes: A meta-analysis. J Appl Ecol. 54(5):1395–1405. doi:10.1111/1365-2664.12892.

de Bello F, Lavorel S, Diaz S, Harrington R, Cornelissen JHC, Bardgett RD, Berg MP, Cipriotti P, Feld CK, Hering D, et al. 2010. Towards an assessment of multiple ecosystem processes and services via functional traits. Biodivers Conserv. 19(10):2873–2893. English. doi:10.1007/s10531-010-9850-9.

de Wit HA, Bryn A, Hofgaard A, Karstensen J, Kvalevåg MM, Peters GP. 2014. Climate warming feedback from mountain birch forest expansion: reduced albedo dominates carbon uptake. Glob Chang Biol. 20:2344–2355. doi:10.1111/gcb.12483.

Díaz S, Lavorel S, de Bello F, Quétier F, Grigulis K, Robson TM. 2007. Incorporating plant functional diversity effects in ecosystem service assessments. Proc Natl Acad Sci. 104(52):20684–20689. doi:10.1073/pnas.0704716104.

Direktoratet for Naturforvaltning. 2009. Handlingsplan for slåttemark. Vol. DN rapport 2009-6.

Duru M, Theau JP, Cruz P. 2012. Functional diversity of species-rich managed grasslands in response to fertility, defoliation and temperature. Basic Appl Ecol. 13 (1):20–31. doi:10.1016/j.baae.2011.10.006.

Ekroos J, Olsson O, Rundlöf M, Wätzold F, Smith HG. 2014. Optimizing agri-environment schemes for biodiversity, ecosystem services or both? Biol Conserv. 172:65–71. doi:10.1016/j.biocon.2014.02.013.

Emanuelsson U. 2009. The rural landscapes of Europe – how man has shaped European nature. Stockholm: Formas.

European Commission. 2016. Natura 2000.[accessed 2016 December 15]: http://ec.europa.eu/environment/nature/natura2000/

European Commission Directorate General for Agriculture and Rural Development. 2005. Agri-environment measures. Overview on general principles, types of measures and application. https://ec.europa.eu/agriculture/sites/agriculture/files/publi/reports/agrienv/rep_en.pdf?

Fontana V, Radtke A, Walde J, Tasser E, Wilhalm T, Zerbe S, Tappeiner U. 2014. What plant traits tell us: consequences of land use change of a traditional agro-forest system on biodiversity and ecosystem service provision. Agric Ecosyst Environ. 186:44–53. doi:10.1016/j.agee.2014.01.006.

Ford H, Garbutt A, Jones DL, Jones L. 2012. Impacts of grazing abandonment on ecosystem service provision: coastal grassland as a model system. Agric Ecosyst Environ. 162:108–115. doi:10.1016/j.agee.2012.09.003.

Garnier E, Cortez J, Billès G, Navas M.-L, Roumet C, Debussche M, Laurent G, Blanchard A, Aubry D, Bellmann A. 2004. Plant functional markers capture ecosystem properties during secondary succession. Ecology. 85:2630–2637.

Haines-Young R, Potschin MB. 2018. Common International classification of ecosystem services (CICES) V5. 1 and Guidance on the application of the revised structure. EEA URL: www.cices.eu.

Harrison P, Berry P, Simpson G, Haslett J, Blicharska M, Bucur M, Dunford R, Egoh B, Garcia-Llorente M, Geamănă N. 2014. Linkages between biodiversity attributes and ecosystem services: a systematic review. Ecosyst Serv. 9:191–203. doi:10.1016/j.ecoser.2014.05.006.

Harrison PA, Dunford R, Barton DN, Kelemen E, Martín-López B, Norton L, Termansen M, Saarikoski H, Hendriks K, Gómez-Baggethun E, et al. 2018. Selecting methods for ecosystem service assessment: A decision tree approach. Ecosyst Serv. 29:481–498. doi:10.1016/j.ecoser.2017.09.016.

Häyhä T, Franzese PP, Paletto A, Fath BD. 2015. Assessing, valuing, and mapping ecosystem services in Alpine forests. Ecosyst Serv. 14:12–23. doi:10.1016/j.ecoser.2015.03.001.

Hector A, Bagchi R. 2007. Biodiversity and ecosystem multifunctionality. Nature. 448(7150):188–190. doi:10.1038/nature05947.

Holland JM, Douma JC, Crowley L, James L, Kor L, Stevenson DR, Smith BM. 2017. Semi-natural habitats support biological control, pollination and soil conservation in Europe. A review. Agron Sustainable Dev. 37 (4):31. doi:10.1007/s13593-017-0434-x.

Holzschuh A, Steffan-Dewenter I, Kleijn D, Tscharntke T. 2007. Diversity of flower-visiting bees in cereal fields: effects of farming system, landscape composition and regional context. J Appl Ecol. 44(1):41–49. doi:10.1111/j.1365-2664.2006.01259.x.

IPBES. 2016. Summary for policymakers of the assessment report of the Intergovernmental Science-Policy. Platform on biodiversity and ecosystem services on pollinators, pollination and food production. Bonn (Germany): Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

Janžekovič F, Novak T. 2012. PCA–A powerful method for analyze ecological niches. In: Sanguansat P, editor. Principal Component Analysis-Multidisciplinary Applications. Croatia: InTech; p. 127–142.

Johansen L, Wehn S, Hovstad KA. 2016. Clonal growth buffers the effect of grazing management on the population growth rate of a perennial grassland herb. Flora. 223:11–18. doi:10.1016/j.flora.2016.04.007.

Kattge J, Diaz S, Lavorel S, Prentice I, Leadley P, Bönisch G, Garnier E, Westoby M, Reich PB, Wright I. 2011. TRY–a global database of plant traits. Glob Chang Biol. 17 (9):2905–2935. doi:10.1111/j.1365-2486.2011.02451.x.

Kleijn D, Rundlöf M, Scheper J, Smith HG, Tscharntke T. 2011. Does conservation on farmland contribute to halting the biodiversity decline? Trends Ecol Evol. 26 (9):474–481. doi:10.1016/j.tree.2011.05.009.

Kleyer M, Bekker RM, Knevel IC, Bakker JP, Thompson K, Sonnenschein M, Poschlod P, van Groenendael JM, Klimes L, Klimesova J, et al. 2008. The LEDA Traitbase: a database of life-history traits of the Northwest European flora. J Ecol. 96(6):1266–1274. doi:10.1111/j.1365-2745.2008.01430.x,

Lê S, Josse J, Husson F. 2008. FactoMineR: an R package for multivariate analysis. J Stat Softw. 25(1):1–18. doi:10.18637/jss.v025.i01.

Laliberté E, Legendre P. 2010. A distance-based framework for measuring functional diversity from multiple traits. Ecology. 91(1):299–305.

Lamarque P, Lavorel S, Mouchet M, Quétier F. 2014. Plant trait-based models identify direct and indirect effects of climate change on bundles of grassland ecosystem services. Proc Natl Acad Sci. 111(38):13751–13756. doi:10.1073/pnas.1216051111.

Lavelle P, Dugdale R, Scholes R, Berhe A, Carpenter E, Codispoti L, Izac A, Lemoalle J, Luizao F, Treguer P. 2005. Nutrient cycling. Ecosystems and human well-being: current state and trends: findings of the condition and trends working group. Washington (Covelo, London): Island Press.

Lavorel S, Grigulis K, Lamarque P, Colace MP, Garden D, Girel J, Pellet G, Douzet R. 2011. Using plant functional traits to understand the landscape distribution of multiple ecosystem services. J Ecol. 99(1):135–147. doi:10.1111/j.1365-2745.2010.01753.x.

Layke C, Mapendembe A, Brown C, Walpole M, Winn J. 2012. Indicators from the global and sub-global Millennium ecosystem assessments: an analysis and next steps. Ecol Indic. 17:77–87. doi:10.1016/j.ecolind.2011.04.025.

Lid J, Lid D. 2005. Norsk Flora. Det norske samlaget. Elven R, editor. Oslo:Det Norske Samlaget

Lutz DA, Burakowski EA, Murphy MB, Borsuk ME, Niemiec RM, Howarth RB. 2016. Trade-offs between three forest ecosystem services across the state of New Hampshire, USA: timber, carbon, and albedo. Ecol Appl. 26(1):146–161.

Mace GM, Norris K, Fitter AH. 2012. Biodiversity and ecosystem services: a multilayered relationship. Trends Ecol Evol. 27(1):19–26. doi:10.1016/j.tree.2011.08.006.

Maes J, Teller A, Erhard M, Grizzetti B, Barredo J, Paracchini M, Condé S, Somma F, Orgiazzi A, Jones A. 2018. Mapping and assessment of ecosystems and their services: an analytical framework for ecosystem condition. Luxembourg: Publications office of the European Union.

Maskell LC, Crowe A, Dunbar MJ, Emmett B, Henrys P, Keith AM, Norton LR, Scholefield P, Clark DB, Simpson IC. 2013. Exploring the ecological constraints to multiple ecosystem service delivery and biodiversity. J Appl Ecol. 50(3):561–571. doi:10.1111/1365-2664.12085.

Millennium Ecosystem Assessment. 2005. Millennium ecosystem assessment synthesis report. Washington (DC).

Norderhaug A, Johansen L. 2011. Semi-natural sites and boreal heaths. In: Lindgaard A, Henriksen S, editors. The 2011 Norwegian Red list for ecosystems and habitat types. Trondheim: Norwegian Biodiversity Information Centre; p. 87–92.

Pakeman RJ. 2014. Leaf dry matter content predicts herbivore productivity, but its functional diversity is positively related to resilience in grasslands. PLoS One. 9 (7):e101876. doi:10.1371/journal.pone.0101876.

Peco B, Carmona C, De Pablos I, Azcárate F. 2012. Effects of grazing abandonment on functional and taxonomic diversity of Mediterranean grasslands. Agric Ecosyst Environ. 152:27–32. doi:10.1016/j.agee.2012.02.009.

Peco B, Navarro E, Carmona C, Medina N, Marques M. 2017. Effects of grazing abandonment on soil multifunctionality: the role of plant functional traits. Agric Ecosyst Environ. 249:215–225. doi:10.1016/j.agee.2017.08.013.

Pereira E, Queiroz C, Pereira HM, Vicente L. 2005. Ecosystem services and human well-being: a participatory study in a mountain community in Portugal. Ecol Soc. 10(2):14. doi:10.5751/ES-01353-100214.

Pereira HM, Navarro LM, Martins IS. 2012. Global biodiversity change: the bad, the good, and the unknown. Annu Rev Environ Resour. 37:25–50+. doi:10.1146/annurev-environ-042911-093511.

Plantureux S, Bernues A, Huguenin-Elie O, Hovstad K, Isselstein J, McCracken DI, Therond O, Vackar D. 2016. Ecosystem service indicators for grassland in relation to ecoclimatic regions and land use systems. Grassland science in Europe. Wageningen: Wageningen Academic Publishers; p. 524–547.

Potschin-Young M, Haines-Young R, Görg C, Heink U, Jax K, Schleyer C. 2018. Understanding the role of conceptual frameworks: reading the ecosystem service cascade. Ecosyst Serv. 29:428–440. doi:10.1016/j.ecoser.2017.05.015.

Potts SG, Vulliamy B, Dafni A, Ne'eman G, Willmer P. 2003. Linking bees and flowers: how do floral communities structure pollinator communities? Ecology. 84 (10):2628–2642. doi:10.1890/02-0136.

Queiroz C, Meacham M, Richter K, Norström AV, Andersson E, Norberg J, Peterson G. 2015. Mapping bundles of ecosystem services reveals distinct types of multifunctionality within a Swedish landscape. Ambio. 44(1):89–101. doi:10.1007/s13280-014-0601-0.

Raudsepp-Hearne C, Peterson G. 2016. Scale and ecosystem services: how do observation, management, and analysis shift with scale—lessons from Québec. Ecol Soc. 21:3. doi:10.5751/ES-08605-210316.

Sala OE, Chapin FS, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber-Sanwald E, Huenneke LF, Jackson RB, Kinzig A, et al. 2000. Biodiversity - Global biodiversity scenarios for the year 2100. Science. 287(5459):1770–1774. doi:10.1126/science.287.5459.1770,

Schirpke U, Tasser E, Tappeiner U. 2013. Predicting scenic beauty of mountain regions. Landsc Urban Plan. 111:1–12. doi:10.1016/j.landurbplan.2012.11.010.

Schmid H, Lehmann B, Buller H, Wilson G, Höll A. 2000. Switzerland: agri-environmental policy outside the European Union. In: Buller H, Wilson G, Höll A, editors. Agri-environmental policy in the European Union. London: Ashgate Publishing Ltd; p. 185–202.

Taugourdeau S, Messad S. 2017. TATALE: tools for assessment with transformation and aggregation using simple logic and expertise.Manual (Version March 2017). Montpellier (France): CIRAD-ES-UMR SELMET; p. 1-11.

Taugourdeau S, Villerd J, Plantureux S, Huguenin-Elie O, Amiaud B. 2014. Filling the gap in functional trait databases: use of ecological hypotheses to replace missing data. Ecol Evol. 4(7):944–958. doi:10.1002/ece3.989.

Totland Ø, Hovstad KA, Ødegaard F, Åström J. 2013. State of knowledge regarding insect pollination in Norway – the importance of the complex interaction between plants and insects. Norway: Norwegian Biodiversity Information Centre.

Trilleras JM, Jaramillo VJ, Vega EV, Balvanera P. 2015. Effects of livestock management on the supply of ecosystem services in pastures in a tropical dry region of western Mexico. Agric Ecosyst Environ. 211:133–144. doi:10.1016/j.agee.2015.06.011.

Tscharntke T, Clough Y, Wanger TC, Jackson L, Motzke I, Perfecto I, Vandermeer J, Whitbread A. 2012. Global food security, biodiversity conservation and the future of agricultural intensification. Biol Conserv. 151 (1):53–59. doi:10.1016/j.biocon.2012.01.068.

Van Oudenhoven APE, Schröter M, Drakou EG, Geijzendorffer IR, Jacobs S, van Bodegom PM, Chazee L, Czúcz B, Grunewald K, Lillebø AI, et al. 2018. Key criteria for developing ecosystem service indicators to inform decision making. Ecol Indic. 95:417–426. doi:10.1016/j.ecolind.2018.06.020.

Vinge H, Flø BE. 2015. Landscapes lost? Tourist understandings of changing Norwegian rural landscapes. Scand J Hosp Tour. 15(1–2):29–47. doi:10.1080/15022250.2015.1010283.

Wehn S. 2009. A map-based method for exploring responses to different levels of grazing pressure at the landscape scale. Agric Ecosyst Environ. 129(1–3):177–-181. doi:10.1016/j.agee.2008.08.009.

Wehn S, Burton R, Riley M, Johansen L, Hovstad KA, Rønningen K. 2018a. Adaptive biodiversity management of semi-natural hay meadows: the case of West-Norway. Land Use Policy. 72:259–269. doi:10.1016/j.landusepol.2017.12.063.

Wehn S, Hovstad KA, Johansen L. 2018b. The relationships between biodiversity and ecosystem services and the effects of grazing cessation in semi-natural grasslands. Web Ecol. 18(1):55–65. doi:10.5194/we-18-55-2018.

Wehn S, Taugourdeau S, Johansen L, Hovstad KA. 2017. Effects of abandonment on plant diversity in semi-natural grasslands along soil and climate gradients. J Veg Sci. 28:838–847. doi:10.1111/jvs.12543.

WorldClim. 2015. WorldClim-global climate data, bioclim. [accessed 2015 December 18]: http://www.worldclim.org/bioclim.