Future impacts of changing land-use and climate on ecosystem services of mountain grassland and their resilience

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Abstract

Although the ecosystem services provided by mountain grasslands have been demonstrated to be highly vulnerable to environmental and management changes in the past, it remains unclear how they will be affected in the face of a combination of further land-use/cover changes and accelerating climate change. Moreover, the resilience of ecosystem services has not been sufficiently analysed under future scenarios. This study aimed to assess future impacts on multiple mountain grassland ecosystem services and their resilience. For a study area in the Central Alps (Stubai Valley, Austria), six ecosystem services were quantified using plant trait-based models for current and future conditions (in 2050 and 2100) considering three socio-economic scenarios. Under all scenarios, the greatest changes in ecosystem services were related to the natural reforestation of abandoned grassland, causing a shift from grassland to forest services. Although the high resilience potential of most ecosystem services will be maintained in the future, climate change seems to have negative impacts, especially on the resilience of forage production. Thus, decision makers and farmers will be faced with the higher vulnerability of ecosystem services of mountain grassland. Future policies should consider both socio-economic and environmental dynamics to manage valuable ecosystem services.

Keywords

Land-use change; Climate change; Ecosystem service resilience; Mountain farming; Permanent grassland

1 Introduction

Mountain grasslands are the result of long-term human agricultural activities, and they have been used for mowing and livestock grazing for centuries. Many of these permanent grasslands are biodiversity hotspots (Wilson et al., 2012) owing to extensive farming practices (MacDonald et al., 2000; Schermer et al., 2016) that provide a variety of ecosystem...
services. While in the past mountain grasslands were managed principally for forage provision, nowadays their importance for regulating (e.g. soil stability, water provision, carbon storage) and cultural ecosystem services (e.g. aesthetic and recreational values) is increasingly recognised (Bürgi et al., 2015; Lamarque et al., 2011). However, mountain ecosystems are highly vulnerable to climate and land-use changes, both of which affect ecosystem services (Schröter et al., 2005).

During the past few decades, large areas of low-productivity grasslands have been abandoned in the European Alps, located mainly on steep slopes and at high altitudes, where their management is linked to high costs (Rutherford et al., 2008). These abandoned areas were located mainly in the subalpine zone below the actual treeline and originated from former forest clearing (Pecher et al., 2011). Subsequently, they were subject to natural reforestation, which resulted in changing landscape patterns (Tasser et al., 2007).

Conversely, activity in the most productive areas in the valley bottoms has intensified, as they are used as highly intensive meadows with several cuts per season or transformed into annual or permanent cropland. Both the abandonment and the intensification have caused both a decline in biodiversity (Lambin et al., 2000; Zimmermann et al., 2010) and changes to ecosystem services (Bürgi et al., 2015; Crouzat et al., 2015; Egarter Vigl et al., 2016; Schirpke et al., 2013). For example, the abandonment of Alpine grasslands has not only reduced forage provision (Briner et al., 2013) but has also had negative impacts on some regulating services such as fire prevention (Navarro and Pereira, 2012), as well as has been negatively related to cultural ecosystem services, in particular to aesthetic landscape values (Schirpke et al., 2016). Nevertheless, positive effects have been found in terms of carbon sequestration (Levers et al., 2015; Nagler et al., 2015), erosion control (Egarter Vigl et al., 2016), timber production, water regulation, and recreational activities (Navarro and Pereira, 2012). These impacts occur at different time scales (Hein et al., 2016), for example, shrub encroachment several years after the abandonment of grassland will affect aesthetic values (Schirpke et al., 2013), while timber production will only be relevant after forest regrowth, which may take several centuries (Tasser et al., 2017).

In mountain areas, ecosystem services of agricultural land are influenced by socio-economic-driven management decisions at all elevations, whereas climate change affects such services more strongly at high altitudes (Briner et al., 2013). In the French Alps, climate-induced changes have been found more important than management decisions for regulating and cultural ecosystem services of mountain grasslands (Lamarque et al., 2014). Whereas manuring increases nutrient availability and changes vegetation structure (species composition), leading to higher biomass production on mown grassland (Quétier et al., 2007), higher temperatures induce a shift in plant species towards higher altitudes, resulting in reduced species richness and altered species composition (Hoiss et al., 2013). Extreme drought periods are expected to occur more frequently and affect water availability, especially in humid regions (Köplin et al., 2014; Leitinger et al., 2015), while reduced water availability, in turn, influences other ecosystem services, for example, leading to reduced forage quantity (Briner et al., 2012; Schirpke et al., 2013), altered carbon allocation (Hasibeder et al., 2015), increasing soil stability, and a lower risk of erosion (Tasser et al., 2003).
Although combined effects of land-use and climate change on ecosystem services have been studied in Switzerland (Briner et al., 2013; Bürgi et al., 2015), Austria (Kirchner et al., 2015; Schirpke et al., 2013), and the French Alps (Lamarque et al., 2014), few studies have included climate-induced impacts on the vegetation composition of mountain grassland under future scenarios (Lamarque et al., 2014). Ecosystem services can be assessed using various modelling approaches, including proxy-based and process-based methods (Lavorel et al., 2017), but trait-based models are most promising to analyse climate-induced changes in ecosystem services at the landscape level (Lamarque et al., 2014), including future shifts in species abundances (Cantarel et al., 2013). Trait-based models describe statistical relationships between biotic and abiotic properties and functional traits that can be linked to specific ecosystem services (Lavorel et al., 2011). The focus on the functional composition of vegetation communities helps not only to explain current ecosystem services but also to project future ecosystem services, accounting for functional responses and changes in species composition (Díaz et al., 2013; Lavorel et al., 2017).

Future provision of ecosystem services depends further on ecosystem resilience, defined as the capacity of ecosystems to cope with disturbance without shifting to another regime (Walker et al., 2004). A resilient system will therefore buffer shocks and disturbances, preserving related ecosystem services, but little is known about the resilience of ecosystems to provide ecosystem services in the face of environmental perturbations (Díaz et al., 2013). Such resilience depends not on the resilience of single species but on the maintenance of important ecosystem functions provided by the plant community. In case of the decline or loss of important functional species, these ecosystem functions of present species might be replaced by other species in the future; however, this depends on species richness and functional redundancy (Oliver et al., 2015). Hence, to maintain ecosystem services that depend on ecosystem functions, resilience in the latter is required (Grigulis et al., 2013; Lavorel et al., 2011). As each ecosystem service is linked to different functions, the level of resilience consequently differs between ecosystem services (Kohler et al., 2017a). Although research on resilience and its implications for ecosystem services is growing, as on mountain areas (e.g., Brunner and Grêt-Regamey, 2016; Kohler et al., 2017a; Schermer et al., 2016), the understanding of the dynamics of the resilience of ecosystem services in mountain grasslands under future scenarios is still limited.

This study addresses two research questions related to expected changing environmental and socio-economic conditions: 1) What are the potential future impacts on multiple ecosystem services in mountain grasslands? 2) How is the resilience of ecosystem services affected? We used climate (Gobiet et al., 2014) and socio-economic scenarios (Kohler et al., 2017b) to project the possible consequences of climate and land-use changes on ecosystem services for the Stubai Valley in Austria. First, we modelled current multiple ecosystem services of grassland under different management intensities and at various altitudes, including forage production, forage quality, carbon storage, soil fertility, water quality, and aesthetic value, using plant functional traits at the ecosystem scale. Then, we analysed future changes in these ecosystem services under three socio-economic scenarios, taking into account projected climate variations until 2050 and 2100 (Gobiet et al., 2014), as well as the natural reforestation of abandoned grassland (Tasser et al., 2017). Finally, we calculated ecosystem
service resilience indicators to assess the resilience of the mountain grassland ecosystem services to future climate and land-use changes.

2 Materials and methods

2.1 Conceptual approach

To project the possible consequences of climate and land-use changes on future ecosystem services of the selected mountain grassland, we combined three different socio-economic scenarios with a climate scenario and analysed two different dates: 2050 and 2100 (Fig. 1). The three socio-economic scenarios (Section 2.3) were discussed with local stakeholders, who also mapped future land-use changes. These land-use change maps were combined with grassland trajectories (Section 2.3.1) to set up a sample design for the field measurements (Section 2.4.1). To assess the impacts on ecosystem services and their resilience under future changes, we first modelled current ecosystem services using trait-based models (Section 2.4.2), which were combined with future ecosystem services accounting for accelerating climate change, using a moderate climate scenario (Section 2.5). Finally, we used the ecosystem service values to calculate resilience indicators for current and future conditions (Section 2.6).

2.2 Study area

In this study, we focused on managed and abandoned grassland in the Stubai Valley, which is located between longitude 11.6°–11.25° E and latitude 46.55°–47.15° N in the Central Alps (Tyrol, Austria) (Fig. 2). The Stubai Valley comprises the municipalities of Fulpmes and Neustift and extends over an area of about 265 km², with elevation ranging between 887 m and 3484 m a.s.l. The various grassland types in the region (see Appendix Table A1) evolved from a long tradition of farming systems adapted to marginal grasslands and cover about 24% (total ~ 64 km²) of the total area. Of these, 24 km² are currently used, with different management intensities. Whereas meadows in the valley bottom are fertilised and cut several times per year, grassland above 1500 m a.s.l. is cut one to two times per year or used for pasture. Abandoned grassland covers an area of about 40 km² and is located mainly above 1500 m a.s.l. Other land-cover types include 28% forest, 1% settlement, and 47% non-usable land (i.e. glacier and rock).

The economic situation in the Stubai Valley is positively influenced by its proximity to Innsbruck, attracting commuters (Schermer et al., 2016), and by tourism, thanks to the presence of a glacier that is skiable almost all year round, good infrastructure for both summer and winter tourism, and an appealing landscape (Schirpke et al., 2016).

2.3 Land-use scenarios

We used three different socio-economic scenarios for the year 2050, with a focus on decision-making processes related to the management of mountain grassland (Kohler et al., 2017b). The scenarios were developed based on key drivers (see Table 1), which were identified in expert interviews and subsequently discussed with local stakeholders. Assuming that these key drivers influence farming conditions in a positive or negative way, two contrasting storylines were developed (positive and negative scenarios). Under the
positive scenario, a diversified agriculture is maintained in cooperation with the tourism industry and its products and related services are valued, although the community concentrates on further developing tourism. In contrast, farming is rather unimportant for food production, and tourism promotes the ‘new wilderness’ under the negative scenario. Further, a trend scenario was used, projecting the current trend continuing in the future, where the community recognises the ecological and societal services of the farming activities.

In a participatory workshop, scenarios were discussed with local farmers who also mapped probable land-use changes related to each scenario (Kohler et al., 2017b). Based on these three scenario maps, changes in the areas of each land-use type were derived (Figs. 1, S1–3 in the supplementary material). Under the trend scenario, the managed grassland area decreased by 8% due to conversion to cropland in the valley bottom and the abandonment of alpine grassland. Under the negative scenario, 17% of currently used grassland was abandoned, mainly in the subalpine belt and the valley bottom. The positive scenario led to the conversion of grassland to cropland in the valley bottom (−2%) and a reactivation of abandoned land in subalpine and alpine grasslands (+8%), slightly increasing its total area (+3%). The area of abandoned grassland decreased by about 18% by 2050 in all scenarios due to natural reforestation.

2.3.1 Grassland trajectories—To set up a sample design for the field work (i.e. vegetation and soil surveys) carried out for functional trait assessment and the subsequent calculation of ecosystem services, we defined grass-land trajectories (i.e. homogenous ecological units that account also for previous land use) as land-use history as well as environmental aspects that influence an ecosystem’s species composition (Quétier et al., 2007). Trajectories were identified according to six key parameters that determine current vegetation types: parent rock (carbon, silicate), altitude (<1200 m, 1200–1500 m, >1500 m), exposition (N, S, E + W), slope (<9°, >9°), fertilisation (yes, no), and former use (e.g. always used as grassland, formerly used as cropland). Data were then combined and mapped, resulting in 27 different trajectories (exclusion of trajectories smaller than 20 ha, ~2.5% of total grassland area) within the study area (Appendix Table A1). To facilitate interpretation of the results, trajectories were associated to different grassland types (Appendix Table A1, Fig. 1).

2.3.2 Scenario maps—To obtain future grassland trajectories, land-use scenario maps from the stakeholder workshop were incorporated into the current trajectory map by adjusting the shape of the grassland patches and updating the trajectory type. Large areas of current and future abandoned grassland are located below the potential treeline (currently about 2350 m a.s.l.; Pecher et al., 2011), and these areas are just in a successional stage or likely subject to natural reforestation (Tasser et al., 2007). The abandonment of grassland in the Stubai Valley started in the 1950s, and forest has already succeeded in some areas. However, reforestation processes of abandoned areas were not discussed or mapped in the stakeholder workshops. Therefore, we needed to model the likely spatial distribution of forest under the scenarios. Although reforestation and the appearance of new forest areas proceeds rather slowly (Schirpke et al., 2012; Tasser et al., 2007), we calculated the time...
span for reforestation (t; in years) for all abandoned areas as a function of geographical position, elevation, time since abandonment, and distance from existing forest (Tasser et al., 2017). Therefore, we first created two maps for 2050 and 2100, depicting the time since abandonment (in years) at the corresponding date, which was derived from historical land-use maps, available for different time steps since 1861 (see Schirpke et al., 2012; Tasser et al., 2009). Finally, we selected all areas that were attributed to be reforested until 2050 and 2100 and converted them to forest in the scenario maps.

2.4 Modelling current ecosystem services

We quantified the current and future ecosystem services of the grassland using quantitative plant trait-based models (Grigulis et al., 2013; Lavorel et al., 2011), which combine abiotic characteristics, plant traits, and the contribution of plant species to standing biomass at the ecosystem scale. A focus was placed on the following six ecosystem services, which are perceived as the most important in the studied grassland (Lamarque et al., 2011): forage production and forage quality (provisioning services); carbon storage, water quality, and soil fertility (regulating services); and aesthetic value (cultural service).

2.4.1 Data collection—Field measurements undertaken included at least four vegetation surveys on each trajectory type in order to characterise its vegetation community. According to the biomass ratio hypothesis (Grime, 1998), the traits of the most abundant species (~highest contribution to total plant biomass) have the greatest effect on ecosystem processes and services (Lamarque et al., 2011). We therefore concentrated on plant species making up at least 60% of the total abundance within the obtained vegetation samples. Data on plant functional traits (vegetative height [VegHt], leaf dry matter content [LDMC], leaf nitrogen concentration [LNC], flowering colour) were obtained from our own measurements and integrated with data from the TRY database (e.g. from Everwand et al., 2014; Kerkhoff et al., 2006; Kleyer et al., 2008; Lavorel et al., 2011; Louault et al., 2005; Müller et al., 2007; Poschlod et al., 2003; Wright et al., 2004; complete list of contributions in the Supplementary material). For each plant trait, we then calculated the community-weighted mean (CWM) (Garnier et al., 2004). Further, associated information (e.g. coverage of flowering species, length of flowering period) was also collected. At least three soil samples were taken for each trajectory in order to measure soil abiotic parameters (denitrifying enzyme activity [DEA], water holding capacity [WHC], pH, soil texture, soil organic matter [SOM], total soil carbon, and nitrogen).

2.4.2 Trait-based models—Based on the current grassland trajectories, we quantified current ecosystem services (with the exception of aesthetic value) using plant trait-based models from Grigulis et al. (2013) and Kohler et al. (2017a) (Table 2). All ecosystem services were referred to annual ecosystem service provision by calculating mean values for forage quality, soil fertility, water quality, and carbon storage. Forage production in the plant trait-based model represented the peak biomass during the harvesting season and was thus extrapolated to total annual production depending on the type of grassland (meadow, pasture) and management intensity (number of cuts per year) following Egger et al. (2004) and Galler (2002). The type of grassland as a function of plant species and length of growing season was considered to determine the productivity of grassland according to Egger et al.
(2004), while management intensity was used to identify the percentage forage quantity of the measured peak biomass; for example, the first cut of intensively used meadows in the valley bottom (3–4 cuts per year) represented about 35–40% of the total annual production (Galler, 2002).

The aesthetic value (index) was calculated using the following equation:

\[ AV = r_{SDI} + r_{LFP} + r_{MC} + r_{CFS}, \]

where

- \( r_{SDI} \) = relative Simpson Diversity Index, i.e. Simpson Diversity Index of the trajectory divided by maximum Simpson Diversity Index of all trajectories
- \( r_{LFP} \) = relative length of flowering period, i.e. length of flowering period of all species in the trajectory divided by maximum length of flowering period of all species in all trajectories
- \( r_{MC} \) = relative maximum number of colours flowering at the same time, i.e. maximum number of colours flowering at the same time in the trajectory divided by maximum number of colours flowering at the same time in all trajectories
- \( r_{CFS} \) = relative coverage of flowering species, i.e. coverage of flowering species in the trajectory divided by maximum coverage of flowering species at the same time in all trajectories

### 2.5 Modelling future ecosystem services

Future ecosystem services were modelled accounting for land-use as well as climate changes. Land-use changes caused by socio-economic drivers were incorporated using the three scenario-based trajectory maps. To analyse climate-induced changes, we used a moderate climate change scenario for the European Alps, A1B in the IPCC AR4 report (Solomon et al., 2007), in line with Gobiet et al. (2014). Under this scenario, temperature is expected to increase by +1.5 K until 2050, along with higher precipitation variability and more frequent summer droughts. This trend is assumed to accelerate in the second half of the 21st century, resulting in a temperature increase of +3.3 K until 2100 and more severe summer droughts (Gobiet et al., 2014). We therefore selected two dates for modelling the impact under the climate change scenario together with the three socio-economic scenarios: the year 2050, which corresponds to the land-use scenarios, and 2100, for which we assumed no further land-use changes. For both dates, farmers expected only a minor impact on farming activities in the face of moderate climate variation unless the next generations change the farming system due to different values or socio-economic conditions (Kohler et al., 2017b). Further, as tourism is of great importance to the study region, the farmers confirmed that they will continue to manage grassland areas because they are crucial for maintaining a traditional landscape picture (Schirpke et al., 2013).

Climate changes were expected to affect both species composition and plant traits, as reported in several studies (e.g. Cantarel et al., 2013; Debouk et al., 2015; Niedrist et al., 2016; Stampfli and Zeiter, 2004). Under the changes expected until 2050, we assumed that
effects on species composition and plant traits were negligible due to species adaption potential, phenotypic plasticity, and nonlinear shifts (Benot et al., 2013; Jump and Penuelas, 2005; Oliver et al., 2015). Variations in species relative abundances and plant traits were therefore included only in the 2100 scenarios. Based on Cantarel et al. (2013), the fraction of grasses was reduced by 10.7%, whereas the fraction of legumes was increased by 7.7% and the fraction of remaining functional groups by 3%. Furthermore, we changed the mean values of two plant traits (LNC −18.5% and LDMC +3.8%) according to Cantarel et al. (2013).

Rising temperatures will also likely result in a longer growing season (Fuhrer et al., 2014). Considering a temperature lapse rate of 6 K per km of elevation (Körner, 2003), 1.5 K and 3.3 K temperature rises will result in an upward shift of vegetation zones (Niedrist et al., 2016) by 250 m and 550 m, respectively. These shifts were considered here by changing the altitude (i.e. values of the digital elevation model [DEM]) by the respective altitudinal difference. Based on the modified DEMs, the length of the growing season was then modelled according to Harflinger and Knees (1999), and mean values were calculated for each trajectory, leading to an extension of the growing season by 15–22 days until 2050 and by 31–48 days until 2100, depending on the trajectory.

Future ecosystem services of the grassland were quantified based on the same plant trait-based models as for current ecosystem services (Table 2). The reduction in grassland area under the socio-economic scenarios implicated that some grassland parcels were replaced by forest or cropland. However, the trait-based models could not be applied to these land cover types due to missing data and because the models were established based on the value range of grassland habitats only. Table 3 provides an overview of ecosystem services for different land cover types based on previous research in the study area, with the data for forest and cropland replacing former grassland areas in the analysis.

### 2.6 Ecosystem service resilience indicators

We calculated resilience indicators for each ecosystem service according to Kohler et al. (2017a), quantifying the resistance and resilience of each trajectory. Resistance is defined as the observed range of ecosystem service values within each trajectory under current conditions, expressed by the normal operating range (NOR), whereas resilience refers to the potential range of ecosystem service values within each trajectory, expressed by the community potential operating range (Com-POR). These operating ranges represent the boundaries of ecosystem services, i.e. minimum and maximum ecosystem service values determined by species composition, which were identified using the same models as described in the previous section. The NOR was calculated based on actually observed data (species abundances and abiotic parameters) for each trajectory resulting from the field measurements. The Com-POR was obtained from simulation (25,000 iterations, software @RISK 7, Palisade DTools 2016, http://www.palisade.com/risk/) of observed data, i.e. all combinations of species abundances and abiotic parameters related to the observed species pool in each trajectory (application of community assembly rules: species were assigned as dominant, subdominant, or subordinate in relationship to observed median abundances [Grime, 1998]). Further details can be found in Kohler et al. (2017a). Resilience potential of
ecosystem services at the ecosystem level can be derived by comparing the nested overlap between the NOR and Com-POR (Fig. 3), i.e. the smaller the overlap, the higher the potential to return to pre-disturbed conditions because of a greater functional pool (Kohler et al., 2017a). Even if, in case of human or environmental disturbance, the NOR is exceeded due to changes in species composition, ecosystem service provision remains resilient as long as it does not also exceed the Com-POR, which maintains the functional pool and, thus, the potential to recover. On the contrary, a high overlap between the NOR and Com-POR indicates a high potential for change because disturbances cannot be buffered any more, i.e. the smaller the overlap, the higher the potential for transition to another ecosystem state (e.g. from mown to grazed grassland) or transformation to another ecosystem type (e.g. from grassland to forest), implying a change in ecosystem service provision. In addition, the position of the NOR within the Com-POR indicates the potential for resilience (NOR in the middle of Com-POR) or for transition/transformation (NOR at either the lower or upper end of Com-POR) (Fig. 3).

For each trajectory and ecosystem service, we therefore identified the NOR, the future NOR (NOR-Future) considering climate-induced changes in species composition and plant traits as described above, and the Com-POR, which remained unchanged under future climatic conditions as it represents the whole functional range of the species pool; however, potential migration or extinction of species could not be considered. We tested if the NOR was nested in the Com-POR to verify that the extension of the range of community functional composition was represented. All cases for which the NOR did not exist within the Com-POR were excluded. We calculated the overlap between the current and future NOR (NOR-Future) and the Com-POR to obtain the resilience potential. Future impacts on the resilience potential of each trajectory and ecosystem service were then analysed by comparing current and future overlap (Fig. 3). Furthermore, to assess shifts in the position of the NOR within the Com-POR, we first identified the relative current (NOR) and future (NOR-Future) position of each ecosystem service within the Com-POR for each trajectory by calculating the percentage relative deviation of the ecosystem service values from the median of the Com-POR. We then calculated differences between current and future absolute deviations, indicating relative changes of position within the Com-POR, i.e. a greater or smaller deviation in the future (Fig. 3). Finally, we tested whether both the NOR and NOR-Future were below or above the median or whether they were on opposite sides of the median in order to analyse the direction of shifts, i.e. a further shift to the lower or upper end of the Com-POR, or a stable position in the middle of the Com-POR.

To assess future changes in resilience at the landscape level, we mapped both the differences in overlap and those in current and future deviations as described above for the three scenarios before calculating the area-weighted mean for each scenario.

3 Results

3.1 Current ecosystem services

Ecosystem services vary spatially based on the influence of the defined trajectory parameters (topography, management, former use) (Fig. 4). Whereas current forage production is up to four times higher in intensively used meadows in the valley bottom than on alpine pastures,
forage quality is higher on unfertilised grassland above 1500 m a.s.l. than on fertilised meadows, and is highest on abandoned grassland. Abandoned grassland also has on average the greatest ecosystem service values for all regulating services (carbon storage, soil fertility, water quality), followed by unfertilised grassland. On average, the aesthetic value of grassland above 1500 m a.s.l. is slightly higher than that of grassland at lower altitudes.

3.2 Future impacts on ecosystem services

We assessed future impacts at the trajectory level, landscape level, and land-use level. At the trajectory level, forage production was projected as increasing by between 9% and 18% until 2050 due to a longer growing season, while all other ecosystem services remained unchanged (Appendix Table A2). After 2050, variations in species composition and plant traits resulted in changes to all ecosystem services, with varying change rates between the different trajectories (Appendix Table A2). At higher elevated trajectories, forage production rose by up to 200%, but was reduced for grassland below 1200 m a.s.l. Forage quality, soil fertility, and carbon changed between −6.6% and +4.1%, with none exhibiting any specific spatial pattern. In contrast, water quality and aesthetic value changed for almost all trajectories by 14.8–27.0% and -0.4–13.7%, respectively.

At the landscape level, forage production for the entire Stubai Valley decreased slightly under the negative scenario and increased under the positive scenario until 2050, but increased between 29% and 42% under all scenarios until 2100 (Fig. 4a). Forage quality evolved negatively under all scenarios, especially under the negative scenario, but with a lower change rate after 2050 (Fig. 4b). The same trends were recorded for soil fertility (Fig. 4c). Water quality initially exhibited a negative trend until 2050 under all scenarios, but increased after 2050 (Fig. 4d). Whereas carbon storage increased continuously by 32–48% until 2100, with lower changes observed under the positive scenario and the greatest changes under the negative scenario (Fig. 4e), aesthetic value increased only a little after 2050 under all scenarios (Fig. 4f).

Fig. 5 illustrates the projected changes in the ecosystem services of managed grassland, as well as the shift from those of abandoned land to ecosystem services provided by forests. Details for different grassland types are depicted in Figs. S4 and S5 in the supplementary material. Under the positive scenario, the ecosystem services of managed grassland showed positive trends for forage production, forage quality, carbon storage, and aesthetic value, especially in unfertilised grassland above 1200 m a.s.l., whereas negative trends appeared mainly for fertilised grassland in the valley bottom. For abandoned grassland, all ecosystem services with the exception of forage production decreased, while regulating and cultural ecosystem services for forest emerged, dominated by water quality until 2050 and carbon storage until 2100. Under the trend scenario, all ecosystem services except water quality decreased for managed grassland, especially in fertilised grassland below 1200 m a.s.l. and unfertilised grassland above 1500 m a.s.l., while abandoned land and forest exhibited similar patterns as in the positive scenario. Under the negative scenario, all ecosystem services with the exception of water quality decreased across all managed grassland trajectories. The same trends as recorded in the other two scenarios were observed for abandoned land and forest under the negative scenario, but with greater magnitudes.
3.3 Resilience of ecosystem services

The NOR, NOR-Future, and Com-POR were identified for each grassland trajectory and each ecosystem service, with the exception of aesthetic value. Resilience indicators were not calculated for the latter, as it was not based solely on a mathematical model and quantitative data, but instead also included qualitative information regarding abundant species.

Fig. 6 illustrates the current and future NOR and Com-POR for selected trajectories of each grassland type (see Appendix Table A1). Trajectories in the diagrams were selected according to their spatial extent to represent the general trend of each grassland type. The delivery of provisioning services ceased for abandoned land (T24) due to a prevalence of shrubs. When the NOR fell within the Com-POR (which was the case for almost all trajectories and all ecosystem services), we calculated the percentage overlap (Appendix Table A3). Under current conditions, overlap was small, ranging from 24% for soil fertility to 34% for forage quality, indicating a high potential for recovery in case of disturbance. Climate change (i.e. changes in species abundances and plant traits) had a rather positive impact on resilience potential, as the overlap of the NOR-Future decreased on average by up to 7% with respect to the current NOR, but also had a negative effect, for example, on carbon storage in abandoned grassland trajectories. Provisioning services were generally more affected than regulating services, with the exception of water quality, for which the future NOR did not fall completely within the Com-POR for several trajectories.

Although all ecosystem services were found to maintain the level of resilience potential until the end of this century under a changing climate, for most trajectories, the NOR-Future shifted towards the lower or upper ends of the Com-POR (greater deviations in the future compared to current conditions, Appendix Table A4), indicating a higher potential for transformation in the future. On average, forage production and water quality exhibited the greatest negative trends (-13% and -24%, respectively), while forage quality was positively influenced (+23%) due to the great shifts of species abundances recorded within abandoned grassland. Soil fertility and carbon storage exhibited on average small negative shifts (−2%).

At the landscape level, we calculated area-weighted mean values for each socio-economic scenario in 2100 in order to analyse impacts on service resilience potential (i.e. overlap of NOR and Com-POR), as well as shifts within the Com-POR in relation to land-use changes. Regarding future impacts on resilience potential, forage production and water quality were mostly positively affected (+25% and +8%, respectively), with the overlap changing only by max. +/-2% for all other ecosystem services. No significant differences were found among the three scenarios. Concerning the position of the NOR-Future within the Com-POR, positive trends were found only for forage quality (+39%), while all other ecosystem services exhibited negative trends, with the greatest impacts recorded for forage production (−77%) and water quality (−19%). The magnitude of these impacts differed slightly between the scenarios and was greatest under the negative scenario and smallest under the positive scenario.
4 Discussion

4.1 Impacts on ecosystem services

In this study, we combined the effects of increasing climatic variation with land-use changes derived from three different socio-economic scenarios to analyse possible spatio-temporal impacts on multiple ecosystem services, thereby facilitating a better understanding of the dynamics of ecosystem services and providing a basis for land-use policies and decision making (Fisher et al., 2009). At the landscape level, variations in ecosystem services until 2050 were mainly linked to land-use changes caused by different socio-economic conditions, with the expected level of climate change (+1.5 K) by the middle of the century. The stakeholder discussions revealed that climate change had no influence on farming system per se; however, farmers may have to consider adapting certain agricultural practices, such as introducing irrigation for grassland management in case of less precipitation during the growing season (Kohler et al., 2017b). Furthermore, higher temperatures and, consequently, a longer vegetation period enabled an increase in the management intensity (number of cuts) of mown meadows (Briner et al., 2012). Although only minor land-use changes were expected compared to past dynamics in the European Alps (Briner et al., 2013; Bürgi et al., 2015; Egarter Vigl et al., 2016; Schirpke et al., 2013), ecosystem services were affected under all scenarios. The abandonment of grassland resulted in a reduction in the total area of managed grassland and, consequently, a decrease in provisioning services, which are related only to grassland (Raudsepp-Hearne et al., 2010). The greatest land cover changes under the scenarios, however, were related to legacy effects, i.e. the expected natural reforestation of large previously abandoned grassland areas. In line with other studies (Briner et al., 2013; Locatelli et al., 2017; Nagler et al., 2015; Schirpke et al., 2013), reforestation had positive effects on carbon storage and water quality, but rather negative impacts on soil fertility and aesthetic value. When grassland was replaced by cropland, all ecosystem services included in this study generally decreased. On the other hand, agricultural systems provide important provisioning services (e.g. food, energy), whereas their contribution to regulating services depends on management practices (Power, 2010).

After 2050, ecosystem services developed similarly across all three scenarios, with the greatest impacts related to shifts in species abundances and trait variations in managed grassland due to climate change (Cantarel et al., 2013), resulting in a reduction in the forage production of certain grassland types, mainly fertilised grassland below 1200 m a.s.l. Given the assumed changes in species composition and plant traits associated with intensive agricultural activities limited to favourable (i.e. flat) areas, ecosystem services for specific grassland types might differ from those presented here, as there is still some debate regarding the impact of climate change on plant functional traits (Cantarel et al., 2013; Stampfli and Zeiter, 2004). Increasing temperatures by the end of this century will also accelerate reforestation processes, replacing the ecosystem services of abandoned grassland with those of forest and augmenting carbon storage in particular. In line with Briner et al. (2013), climate change had an increasing influence on ecosystem services related to agriculture, i.e. forage production, with increasing elevation. Furthermore, climate-induced changes will likely become more important than management changes for future ecosystem service provision in mountain regions (Lamarque et al., 2014), as the highest rates of
abandonment of grassland occurred in many parts of the European Alps already during the second half of the past century (Egarter Vigl et al., 2016). Recent abandonment rates were lower at higher altitudes than in lowland regions in the French Alps (Hinojosa et al., 2016).

4.2 Resilience of ecosystem services

We analysed the resilience potential of ecosystem services by calculating community resistance and resilience for each trajectory and ecosystem service (Kohler et al., 2017a). For almost all trajectories, NOR values lay within their respective Com-POR, indicating that ecosystem service provision remains resilient in case of environmental or human-induced disturbances. The good-to–high potential for resilience can be explained by the high functional diversity of the community with respect to these ecosystem functions. In contrast, a low potential for resilience (high overlap between current and potential operating range) might be linked to higher functional redundancy within the trajectory (Oliver et al., 2015). In contrast to the results of a study area (Lautaret) in the French Alps, we found a higher potential for resilience for regulating services than for provisioning services (Kohler et al., 2017a). Our results also suggest a difference in the resilience potential of different ecosystem services in the same trajectory. For example, current forage production and water quality in fertilised grassland above 1500 m a.s.l. (T13) were positioned at the lower end of their potential range, whereas soil fertility and carbon storage exhibited resilience potential at both the upper and lower boundaries of their Com-POR. In contrast, current forage quality ranged at the upper boundary of its Com-POR. Future ecosystem services exhibited negative trends for forage production, forage quality, and soil fertility in the same trajectory, but positive trends for water quality and carbon storage, leading to a shift in ecosystem services in the future.

Although all ecosystem services maintained their resilience when species composition and plant traits changed over time, most trajectories exhibited a (further) shift towards the lower or upper end of their Com-POR, indicating the direction in which ecosystem service provision might shift, as well as the direction of transformation if resilience is lost. In case it might be necessary to increase fertiliser input to maintain provisioning services of grassland in the long term, a transformation to another regime with other associated ecosystem services might be desirable (Folke et al., 2004). At the landscape level, provisioning services were more affected than regulating services, with impacts greatest under the negative scenario. The observed high resilience potential may be explained by the existence of well-adapted plant communities within the trajectories that have arisen due to centuries of active land-use (Schermer et al., 2016). However, as ecosystems within mountainous terrain are characterised by heterogeneous site conditions, micro-habitats, and continuous disturbance regimes, functional composition might still be diverse and thus affect ecosystem services and their resilience in the future (Díaz et al., 2013).

4.3 Management implications

In mountain regions with persistent grassland farming, farmers will have to cope with stronger climate variations and less resilient ecosystem services in the future. To strengthen the resilience of ecosystem services, species richness and functional diversity should be preserved through extensive management practices and the maintenance of a heterogeneous
landscape pattern (Isselstein et al., 2005; Landis, 2017; Plantureux et al., 2005). Therefore, the Common Agricultural Policy of the EU (CAP) has paid increasing attention to environmental issues. For example, agri-environmental measures (AEM) have been introduced to financially reward farmers for biodiversity-and ecosystem services-related farm management. However, if grassland farming is not subsidised anymore, a decline in provisioning services may force farmers to intensify farming activities or lead to further abandonment, especially at high altitudes (Briner et al., 2013). The maintenance or, in many cases, a relaunch of traditional grassland management practices is therefore essential to maintain and manage valuable grassland ecosystem services in the future. As farming activities are not only influenced by agricultural policies and markets, but depend also on social and cultural values (Schmerer et al., 2016), they should be valued for their contribution to support provisioning and regulating ecosystem services as well as to preserve the cultural landscape and associated aesthetic, recreational, and cultural values (Bernues et al., 2014; Schirpke et al., 2016; Van Berkel and Verburg, 2014). Further research should therefore include socio-economic dynamics related to tourism activities, population dynamics, and societal preferences that influence the management and use of the ecosystems (Zoderer et al., 2016). Trends in societal preferences are difficult to predict, but cultural services as well as biodiversity conservation are increasingly recognised by people in many countries (Bell et al., 2007).

Although a deceleration of land-use changes due to socio-economic drivers can be expected in the European Alps (Briner et al., 2013; Bürgi et al., 2015; Egarter Vigl et al., 2016; Kohler et al., 2017b; Schirpke et al., 2013), land cover will continue changing due to reforestation processes (Tasser et al., 2017). Our results indicate that due to these legacy effects, ‘typical’ ecosystem services of mountain grassland will increasingly shift to those of forest. This trend has already been reported for large areas in the Alps, particularly in Italy (Egarter Vigl et al., 2016; Garbarino et al., 2014; Orlandi et al., 2016), leading to a changed landscape pattern and negative impacts on cultural ecosystem services (Schirpke et al., 2016), but rather positive effects on regulating services (Egarter Vigl et al., 2016; Levers et al., 2015; Nagler et al., 2015; Navarro and Pereira, 2012). These consequences do not appear immediately but are the result of a slow succession process over several years, decades, or even centuries (Tasser et al., 2017). Additionally, impacts may appear already after several years (e.g. aesthetic values, carbon sequestration [Schirpke et al., 2013]) or only after several decades to centuries (e.g. timber production). Landscape planners and decision makers therefore have to deal with long-term effects of past land-use changes (Hein et al., 2016).

4.4 Strengths and limitations of modelling approach

To quantify the ecosystem services of the studied mountain grassland, we used trait-based models (Grigulis et al., 2013; Lavorel et al., 2011). These models are considered suitable for the display of fine-scale heterogeneity between grassland types (Eigenbrod et al., 2010), as well as the capture of biophysical site characteristics (e.g. topography, soil) and variability in management (e.g. frequency of mowing) (Bennett et al., 2009; Díaz et al., 2013; Quétier et al., 2007), variables that determine plant community composition at the site. Although it is still uncertain how climate change affects ecological functions (Oliver et al., 2015), trait-based models are able to integrate a range of possible future changes, such as variations in
species composition and abundances, as well as changes in single traits. In our study, although we were able to modify relative species abundances, we could not consider possible species turnover over time or changes in traits other than LNC and LDMC because of missing data. Further field experiments are necessary to understand changes in functional traits driven by climate change.

The application of trait-based models requires much data, usually obtained from field measurement (Lavorel et al., 2011), which is very labour-intensive and time-consuming. Databases such as TRY (Kattge et al., 2011) are promising data sources for ecosystem service assessment at the regional level, although available traits must be used carefully because of the intraspecific trait variability that results from the development and adaptation of species to environmental change (Albert et al., 2011). In this study, while we focused on current grassland within the study area, land-use changes under the scenarios caused a shift from grassland to forest or cropland, for which no trait data were available. Furthermore, although a range of ecosystem service values for each trajectory is obtained using the adopted method, no further spatial differentiation within a trajectory is possible, which is problematic, as, for example, forage production varies highly with elevation (Egger et al., 2004). Consequently, trajectories must be well-defined, especially in mountain regions, in order to adequately depict differences in site characteristics at the landscape level. For assessments with finer spatial resolution in mountain areas, other GIS-based modelling approaches might be more suitable for the mapping of small-scale differences if they rely on DEMs of topography and habitats (e.g. Schirpke et al., 2013).

5 Conclusions

This study quantified the future impact of land-use and climate changes on multiple ecosystem services in mountain grassland, including on their resilience potential. The analysed land-use and climate scenarios had both positive and negative impacts on these ecosystem services. Our results indicate that socio-economic driven land-use changes, especially the legacy effects of abandonment, will have a greater influence than climate change on ecosystem services until the middle of the century. In the second half of the century, however, accelerating climate change will become the more important driver of changes to ecosystem services in mountain grasslands, especially at high altitudes. Moreover, increasing temperatures will accelerate reforestation processes, leading to a shift in ecosystem services in favour of those of forest.

Most of the analysed ecosystem services demonstrated high resilience potential under current conditions. Although our results suggest that resilience will be maintained in the future, climate change will likely lead to higher potential for transformation with negative impacts on ecosystem services for some trajectories, especially on forage production. Consequently, management options for mountain grassland might be restricted, and decision makers and farmers will be faced with the higher vulnerability of ecosystem services. Future policies should consider both socio-economic and environmental dynamics to manage valuable ecosystem services.
**Supplementary Material**

Refer to Web version on PubMed Central for supplementary material.

**Acknowledgements**

This study was supported by ERA-Net BiodivERsA and the national funder Austrian Science Fund FWF (I 1056-B25) as part of the 2011–2012 BiodivERsA call for research proposals (project REGARDS), as well as by the Austrian Federal Ministry of Science, Research and Economy HRSM – cooperation project KLIMAGRO. This study was conducted at the LTER site ‘Stubai Valley’ (Austria, master site, member of the LTSER platform ‘Tyrolean Alps’). UT, GL, US and MK are part of the research focus ‘Alpine Space – Man and Environment’ at the University of Innsbruck. The study was also supported by the TRY initiative on plant traits (http://www.trydb.org), whose database is hosted, developed and maintained by J. Kattge and G. Bönisch (Max Planck Institute for Biogeochemistry, Jena, Germany). TRY is/has been supported by DIVERSITAS, IGBP, the Global Land Project, the UK Natural Environment Research Council (NERC) through its program QUEST (Quantifying and Understanding the Earth System), the French Foundation for Biodiversity Research (FRB) and GIS “Climat, Environnement et Société” France.

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Fig. 1.
Combination of socio-economic scenarios and climate change scenario used to analyse future impacts on ecosystem services (ES) and their resilience. The positive and negative scenarios represent two contrasting storylines, whereas the trend scenario projects current trends continuing in the future. White boxes refer to land-use and climate change scenarios, light grey shaded boxes indicate mapping and modelling approaches, and dark grey shaded boxes are intermediate and final results.

Ecosyst Serv. Author manuscript; available in PMC 2019 May 18.
Fig. 2. Location of study area, current distribution of grassland, and changes in area of different grassland types (see Appendix Table A1) under the three scenarios in 2050 and 2100 based on Kohler et al. (2017b). The decrease in total grassland area in all three scenarios is due to conversion to forest or cropland. Figs. S1–3 in the Supplementary material illustrate future grassland distribution, including converted land-use types.
Fig. 3.
Percentage overlap between the NOR/NOR-Future and Com-POR indicates resilience potential, i.e. the smaller the overlap, the higher the potential for resilience (Kohler et al., 2017a). Differences between NOR and NOR-Future signal changes in future resilience potential. The position of the NOR/NOR-Future within the Com-POR, expressed as the percentage relative deviation from the median of the Com-POR, indicates the potential for resilience (NOR in the middle of Com-POR) or for transition/transformation (NOR at either the lower or upper end of Com-POR). Differences between current and future deviations suggest future shifts within the Com-POR.
Fig. 4.
Left: Current landscape pattern of ecosystem services, represented by quantile-based data classification, for managed and abandoned grassland in terms of a) forage production (FP), b) forage quality (FQ), c) soil fertility (SF), d) water quality (WQ), e) carbon storage (CS), and f) aesthetic value (AV). Right: Future trends in normalised ecosystem services under three socio-economic scenarios and increasing climate change. All scores are normalised by their current levels.
Fig. 5.
Development of ecosystem services until 2050 and 2100 related to different land-use/cover types (FP – forage production, FQ – forage quality, SF – soil fertility, WQ – water quality, CS – carbon storage, AV – aesthetic value). Forest in the scenarios result from the natural reforestation of abandoned grassland. Values are re-scaled to 0–100. Details for different grassland types are depicted in Figs. S4 and S5 in the supplementary material.
Fig. 6. Current (NOR) and future resistance (NOR-Future) and resilience (Com-POR) of selected trajectories for a) forage production, b) forage quality, c) soil fertility, d) water quality, and e) carbon storage. Trajectories represent different grassland types: T1 - unfertilised grassland 1200–1500 m a.s.l., T3 - unfertilised grassland > 1500 m a.s.l., T7 - fertilised grassland < 1200 m a.s.l., T11 - fertilised grassland 1200–1500 m a.s.l., T13 - fertilised grassland > 1500 m a.s.l., T24 - abandoned grassland > 1500 m a.s.l. Boxes represent values within the 5th to 95th percentile; whiskers contain values outside of this range, i.e. minimum to maximum.
values. A small overlap between the NOR/NOR-Future and Com-POR indicates high resilience potential, whereas a great overlap indicates high transformation potential.
Table 1
Description of scenarios based on key drivers (Kohler et al., 2017b).

| Key driver                                           | Positive scenario                  | Trend scenario                  | Negative scenario                  |
|------------------------------------------------------|------------------------------------|---------------------------------|------------------------------------|
| Touristic services, i.e. managed grassland           | Very important                     | Important                       | Less important                     |
| Demand for local products                           | High demand                        | Moderate demand                 | No demand                          |
| Demand for areas for settlement or energy production| Low demand                         | Low demand                      | High demand                        |
| Supplementary income (on-/off-farm)                 | Important                          | Important                       | Very important                     |
| Farm succession                                     | Guaranteed                         | In most cases certain           | In most cases uncertain            |
| Land-use structural change                          | Intensification of some sites in the valleys/slight increase in abandonment at higher altitudes | Intensification of some sites in the valleys/slight increase in abandonment at higher altitudes | Farmers expand their businesses or stop farming |
| Subsidies                                            | Direct payments important for income | Direct payments important for income | Direct payments do not guarantee secure income |
| Regulations                                          | No retirement of livestock farmers  | Retirement of some small-scale livestock farmers | Retirement of small-scale livestock farmers |
| Cooperation among farmers/with municipality         | Creative cooperation and new income possibilities | Good                            | Less cooperation                   |
| Sustaining cultural landscape                       | Important                          | Important                       | Not important                      |
| Climate change                                      | No influence on farming system, but on some farming practices (e.g. irrigation) | No influence on farming system, but on some farming practices (e.g. irrigation) | No influence on farming system, but on some farming practices (e.g. irrigation) |
## Table 2

Quantification of ecosystem services based on plant trait-based models developed for Alpine grasslands (Grigulis et al., 2013; Kohler et al., 2017a).

| Ecosystem service       | Indicator                     | Unit          | Equation                                                                                                                                 |
|-------------------------|-------------------------------|---------------|-------------------------------------------------------------------------------------------------------------------------------------------|
| Forage production       | Green biomass                 | g.m⁻²         | Green biomass = −2 + 7.53*CWM_LNC¹ + 6.566*CWM_VegHt² + 7.83*WHC³                                                                  |
| Forage quality          | Crude protein content         | XP g.kg⁻¹     | Total digestible N = 201.9 − 0.2691*CWM_LDMC⁴ − 2.013*CWM_VegHt² + 4.6*WHC³                                                                  |
| Soil fertility          | Soil nitrogen mineralisation potential | NH₄-N μg.g⁻¹.d⁻¹ | Nmin log = 1.372 + 0.8993 + (1.916*log(CWM_LDMC⁴)/1000)) + 1.024*DEA_log⁷                                                             |
| Water quality           | Leached nitrogen              | NO₃-N μg.g⁻¹  | Leached_NO₃_N_soil_log = −0.5631 + (0.3851 * NO₃_soil_log⁵) + (0.7690 * Vmax_log⁶)                                                  |
| Carbon storage          | Soil organic matter           | SOM%          | %SOM_log = 1.282 + 0.8266 + (1.494*log(CWM_LDMC⁴)/1000)) + 0.4402*DEA_log⁷                                                          |

¹/CWM_LNC (mg.g⁻¹) = community weighted mean of LNC.
²/CWM_VegHt (cm) = community weighted mean of VegHt.
³/WHC (%) = (0.34*Clay (m%)) + (0.9*OM) + 0.93.
⁴/CWM_LDMC (g dry.g⁻¹ fresh) = community weighted mean of LDMC.
⁵/NO₃_soil_log (μg g⁻¹ soil) = 0.1041 + 0.0361*CWM_LNC.
⁶/Vmax_log (μg N-NH₄ g⁻¹.h⁻¹) = 0.001343 + 0.5705*NO₃_soil_log + 0.2606 * pH.
⁷/DEA_log (N-N₂O μg.g⁻¹.h⁻¹) = 0.4620 + 0.5332*soil_total_N_%_log + 0.2077*NO₃_soil_log.
Table 3
Mean ecosystem service values per hectare for different land-use/cover types. The displayed ecosystem service values for forest and cropland were integrated into our analysis, with values for grassland presented only for comparison.

| Ecosystem service         | Unit          | Grassland (intensive) | Grassland (extensive) | Cropland | Young forest | Mature forest | References               |
|---------------------------|---------------|-----------------------|-----------------------|----------|--------------|---------------|--------------------------|
| Soil fertility            | NH₄ (kg·ha⁻¹) | 0.41                  | 0.28                  | 0.29     | 0.29         | 0.29          | Based on Mantl (2013)    |
| Water quality             | NO₃ (kg·ha⁻¹) | 9.86                  | 6.72                  | 7.04     | 6.98         | 6.98          |                         |
| Carbon storage            | C (kg·ha⁻¹)   | 7.24                  | 11.13                 | 8.38     | 31.00        | 44.80         | Adapted from Tappeiner et al. (2008) |
| Aesthetic value           | Index         | 6.43                  | 7.03                  | 4.59     | 7.09         | 6.06          | Schirpke et al. (2016)   |