Optimising ecosystem services provided by grassland systems

Huguenin-Elie O.[1], Delaby L.[2], Le Clec'h S.[3], Moreno G.M.[4], Teixeira R.F.M.[5] and Schneider M.K.[1]
[1] *Agroscope, Forage Production and Grassland Systems, 8046 Zurich, Switzerland;* [2] *INRA, UMR Physiologie, Environnement et Génétique pour l'Animal et les Systèmes d'Elevage, 35590 Saint-Gilles, France;* [3] *Agricultural Economics and Policy, ETH Zurich, 8092 Zürich, Switzerland;* [4] *Department of Plant Biology, Ecology and Earth Science, Forestry School, University of Extremadura, 10600 Plasencia, Spain;* [5] *Marine, Environment and Technology Centre (MARETEC), Instituto Superior Técnico, Universidade de Lisboa, Lisboa, Portugal; olivier.huguenin@agroscope.admin.ch*

Abstract

Grasslands are multifunctional, producing forage for livestock while providing a wide array of ecosystem services. The value of grasslands for society thus extends far beyond their direct economic value for livestock production. Nevertheless, some antagonisms exist between benefits that can potentially be provided by grasslands. Hence, optimising the delivery of multiple benefits requires consensual decisions. This paper gives an overview of the current state of knowledge about ecosystem services provided by grasslands and discusses trade-offs at the field and the farm scales and how to evaluate them. Management options are available at the field scale to improve provisioning, regulating or maintenance services (biodiversity conservation, climate change mitigation or the regulation of nutrient cycles). Multi-species swards with an optimal legume abundance stand out as facilitators of multiple ecosystem services. However, the different services are maximised at different levels of grassland intensification and none of the options available at the field scale alleviates the conflicts occurring along the intensification gradient. We conclude that multiple services can be optimised by combining specific improvement measures at the field scale with heterogeneity in management intensity, involving multiple grassland types, at the farm or landscape scale.

Keywords: production, biodiversity, carbon storage, nutrient cycling, multi-criteria analysis

Introduction

Society expects much more from grassland-based agricultural systems than milk and meat products (Dumont *et al.*, 2016). Grasslands supply multiple ecosystem services (ESS – provisioning, regulating, maintenance and cultural) to a greater or lesser extent depending on their management, botanical composition, location within the landscape and pedo-climatic conditions (Duru *et al.*, 2015). In the scientific literature, the term 'ecosystem services' (ESS) is used to refer to various concepts (Plantureux *et al.*, 2016). Here, we follow the definition of the 'Common International Classification of Ecosystem Services – CICES' (Haines-Young and Potschin, 2013). This definition distinguishes ESS from ecosystem goods and benefits, which is crucial to properly evaluate ESS of agro-ecosystems. Indeed, the delivery of agricultural goods is supported by both ESS and agricultural inputs (Palomo *et al.*, 2016). According to the CICES, ESS 'retain a connection with the underlying ecosystem functions, processes and structures that generate them', while goods do not. The supply of goods is a primary goal of agriculture and so we also discuss it in this paper.

The relationships among ESS can be synergetic or antagonistic, non-linear and scale sensitive (Bennett *et al.*, 2009). The simultaneous evaluation of multiple ESS is essential to assess multifunctionality but it is also challenging. Moreover, the valuation of ESS is influenced by the socio-economic context and is thus highly sensitive to local conditions (Dumont *et al.*, 2016). We distinguish between 'antagonism' among ESS, when the provision of one ESS declines concomitantly with the increase of another one, and 'trade-off', when beneficiaries choose a balance between various benefits provided by the system. It follows that trade-offs imply a weighting of the benefits, while antagonisms simply describe a type of relationship among ESS. This nuance is a complement to the typology of ESS relationships proposed by Bennett *et al.* (2009). In addition, we define 'field' as a uniformly managed land area. In this review article we highlight options to enhance the supply of ESS from grasslands. We focus on the effect of agricultural management, often through its impact on the grassland plant community, on the delivery of provisioning, regulating and maintenance services. Methods to evaluate multiple ESS and the supply of ESS at the landscape scale are also discussed.

Management options optimising the supply of ESS

Provisioning ESS

Forage for ruminants has been the primary motivation for grassland creation and remains the chief good delivered by grasslands. Huyghe *et al.* (2014) reported maximum yields of up to 20 t DM ha^{-1} and identified temperature and water availability as the main drivers for the huge yield variability in Europe. Often, a large spatial variability in grassland yield also exists at the local scale, depending on soil characteristics, microclimate, topography, botanical composition and management (inputs of nutrients, water, labour and energy). In the Mediterranean region, for example, rainfed semi-natural grasslands produce on average as little as 2 t DM ha^{-1} whereas irrigated and fertilised ones yield up to 15 t DM ha^{-1} (Smit *et al.*, 2008). The yield increase in this case is mainly due to an increase in agricultural inputs, although alleviating drought conditions may also improve ecosystem functioning (for instance nutrient availability). Thus, only a fraction of the yield gains arise from an increase in ESS. The importance of distinguishing provisioning ESS from total yield is well exemplified by the effect of grass-legume mixtures on crude protein yield (Figure 1). If the nitrogen (N) yield of a pure grass sward is enhanced by increased N fertilisation, the increase in provisioning ESS is marginal because nearly the entire additional yield originates from fertiliser (1). If N yield is enhanced by the inclusion of legumes in the sward, the increase in provisioning ESS is tremendous, mainly by symbiotic N_2 fixation (SNF; 2). On the contrary, if grass-legume mixtures receive additional N fertiliser, the provisioning ESS decreases even if the yield is maintained because SNF declines (3).

The botanical composition of the grassland, and especially the presence and proportion of legumes, greatly affects the delivery of provisioning ESS (Figure 1). Legumes, in mixtures with grasses generally boost biomass and crude protein yields (Finn *et al.*, 2013; Suter *et al.*, 2015), as well as weed suppression (Connolly *et al.*, 2017). Forbs may also promote high provisioning ESS by multi-species grasslands as indicated by Husse *et al.* (2017) for *Cichorium intybus* L. and of Cong *et al.* (2017) for *Plantago lanceolata* L. Forbs widens the range of functional trait values in the community (for instance range of rooting depths). However, the positive effect of forbs on provisioning ESS seems weaker and less consistent than that of legumes combined with grasses. Mixture effects on provisioning ESS may differ under grazing as compared to mowing partly because of selective grazing by livestock. Nevertheless, increasing botanical complexity from one to five species (two grasses, two clovers and chicory) has been shown to increase daily milk production of dairy cows under intensive rotational grazing (Roca-Fernández *et al.*, 2016).

The second key aspect of forage provision is its quality for the livestock as determined by its digestibility, palatability and its content in energy, protein and other compounds (Huyghe *et al.*, 2008). Forage digestibility and protein content decrease with increasing plant maturity and the associated formation of structural tissues, but very frequent defoliations aimed at harvesting young forage usually reduce annual biomass yield (Pontes *et al.*, 2007), increase harvesting costs and is detrimental to several ESS (following sections). Annual yield needs to be traded off against forage quality. At the field scale, forage yield can be increased without compromising forage digestibility and protein content by adjusting the relative legume abundance in the sward (Sturludóttir *et al.*, 2014). Otherwise, trade-offs between forage quality

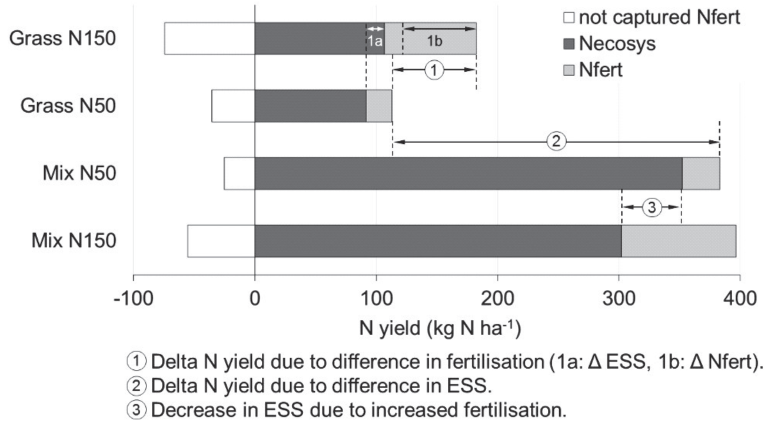

Figure 1. Nitrogen (N) yield of pure grass swards (Grass) and grass-legume mixtures (Mix) fertilized with 50 or 150 kg N ha^{-1} yr^{-1}. Nfert: N from fertiliser applications, Necosys: N from the ecosystem (symbiotic N_2 fixation, soil). Data from Nyfeler *et al.*, 2011.

and other benefits can best be targeted at the farm scale. Indeed, most farms keep animals that differ in energy and protein demand (animals of various growth or lactation stages, sometimes various species or breeds). For example, too intensive feeding of replacement heifers can impair milk production later on (Sejrsen and Purup, 1997). In addition, adapted hardy breeds can be fed on low-quality forage and valorise marginal lands or infrequently defoliated fields by producing valuable meat products (Zehnder *et al.*, 2017). Thus, prioritising forage quality on the area needed for the most demanding livestock and other benefits on the remainder may be an efficient trade-off.

Biodiversity conservation

Some grasslands are among the communities with highest plant species richness, if small scales up to 50 m^2 are considered (Wilson *et al.*, 2012). Recurring disturbance by mowing or grazing is thought to reduce the asymmetry of competition and benefit the coexistence of many plant species (Zobel, 1992). Permanent grasslands also shelter high plant genetic diversity and are a valuable resource for breeding endeavours (Boller *et al.*, 2009). High plant diversity usually promotes diversity at higher trophic levels, although correlations among taxonomic groups are sometimes weak (Lüscher *et al.*, 2015). Moreover, the turnover of stubble and roots sustains complex food webs in grassland soils (Creamer *et al.*, 2016).

Despite the high potential of grasslands for biodiversity, agricultural grasslands are often relatively species poor and populated by ubiquists (Aviron *et al.*, 2009) due to competitive exclusion of specialist species following an increase in nutrient availability (Foster and Gross, 1998). A review by Humbert *et al.* (2016) pointed out that the amount and duration of N addition act additively on reducing plant species richness. Frequency and timing of harvest operations have a major impact on the faunal communities (Buri *et al.*, 2013). Thus, at the field scale, there is a clear antagonism between management intensity and biodiversity (Dumont *et al.*, 2016). A number of specific measures have been suggested to sustain biodiversity at the field scale. First, plant species may be directly added to existing swards. The success of these interventions depends on site conditions, the propagule material and sward management after addition (Kiehl *et al.*, 2010). Adding legumes can benefit both pollinator populations (Williams and Osborne, 2009) and grassland productivity (Teixeira *et al.*, 2015). In any operation of seed addition, care must be taken on its effect on composition and genetic diversity of the native vegetation (Schröder and Prasse, 2013). Secondly, invertebrate diversity can be helped by adjusting mowing time to target species (Buri *et al.*, 2013; Sabatier *et al.*, 2015) by the creation of uncut refuge areas for mobile organisms (Buri *et al.*, 2013) and by avoiding rotatory machinery such as conditioner or flail mowers (Humbert *et al*., 2009). Third, extensive grazing creates an uneven vegetation structure, i.e. various ecological niches, which may favour faunal diversity (Sabatier *et al*., 2015). A prime example for high niche diversity are wooded pastures. Scattered trees produce a fine mosaic of gradients in terms of light, microclimate, soil nutrient, moisture, food availability, and even of disturbance caused by uneven use of space by livestock (Moreno *et al*., 2016). The spatial heterogeneity of wooded pastures contributes to a higher diversity of multiple taxa compared to other adjacent land uses (Plieninger *et al*., 2015). Moreover, as a mixture of forest and grasslands, wooded pastures often conserve a specialised set of species.

At the farm scale, increasing the heterogeneity of grassland management in space and time may benefit biodiversity without impeding overall productivity (Sabatier *et al*., 2015). This has been shown for a rotational grazing system with the exclusion of one paddock from grazing for two months during the main flowering period (Ravetto Enri *et al*., 2017). Huguenin-Elie *et al*. (2014) compared model farms managing the same number of mountain grasslands but with contrasting management heterogeneity among grasslands. The 'intensive bottom' farm (with 20 intensive grassland fields on the bottom of the valley) had a considerably lower total plant species richness than the 'extensive slopes' farm (with 20 extensive grassland fields on the valley slopes) (Table 1). Most importantly, the species richness of the 'four grassland types' farm (comprising of intensive and extensive grasslands on the bottom and slopes of the valley) was similar to the one of the 'extensive slopes' farm, although half of its grasslands were intensively managed. This is explained by the large β-diversity between grassland types. It should be noted that the location of the different grassland types should be taken into consideration to maximise habitat area and connectivity within and between farms (Kremen *et al*., 2007).

Climate change mitigation

Grasslands are key for climate change mitigation because of (1) the amount of carbon (C) stored in their soils, with 340×10^9 t C in the top meter, equivalent to half the atmospheric C (Conant *et al*., 2001; FAO, 2017), (2) the emission of greenhouse gases (GHG) such as N_2O from soil and CH_4 from ruminants associated to grassland-based farming, and (3) the sensitivity of GHG fluxes to farming practices (Soussana *et al*., 2004; Lal *et al*., 2015). Because it is easier and faster for soils to lose than to gain C, it is crucial to at least maintain these stocks (Smith, 2014). Grassland soils are net sinks of 0.60 ± 0.64 SD t C ha^{-1} yr^{-1} in Europe (Freibauer *et al*., 2004) and globally, C sequestration by permanent pastures could offset up to 4% of the global GHG emissions (Lal, 2004). Hence, C sequestration by grasslands could partly mitigate GHG emissions from livestock production. In nine European grasslands, C sequestration did offset 56% of the emissions of N_2O from soil and of CH_4 from enteric fermentation (weighed in CO_2e; Soussana *et al*., 2010). If accounting for the digestion of the harvested forage, the net GHG balance reached 0.1 t C-CO_2e ha^{-1} yr^{-1}, indicating a moderate net sink activity. In contrast, Chang *et al*. (2015) estimated a net sink of 0.15 ± 0.07 t C ha^{-1} yr^{-1} for European grasslands and identified

Table 1. Plant species richness at the field scale for four types of mountain grasslands and at the farm scale for three model farms, all with 20 grassland fields, but comprised of only one or of all four of the grassland types. At the farm scale, the species richness is the aggregated number of species. The model farms were constructed by randomly sampling fields within the corresponding grassland types (100 resampling per model farm; Huguenin-Elie *et al*., 2014).

Field scale	**Species richness**	**Farm scale**	
Grassland type		**Model farm**	**Species richness**
IB: Intensive, bottom of valley (n = 40)	31 ± 0.9	'intensive bottom': 20 × IB	110 ± 8.6
IS: Intensive, slopes of valley (n = 18)	40 ± 2.1	'extensive slopes: 20 × ES	175 ± 5.3
EB: Extensive, bottom of valley (n = 14)	40 ± 2.5	'2 grassland types': 12 × IB and 8 × EB	155 ± 8.9
ES: Extensive, slopes of valley (n = 35)	46 ± 1.7	'4 grassland types': 5 × each IB, IS, EB and ES	169 ± 9.3

grassland-based systems as a net source of GHG (0.5 t C-CO_2e ha^{-1} yr^{-1}) when N_2O, CH_4 and CO_2 from forage digestion were included. Grassland C balance largely depends on the current level of soil organic C (SOC) and C sequestration might be a saturating process (Castellano *et al*., 2015). Drained organic soils may release large amounts of C (Soussana *et al*., 2010). Tiemeyer *et al*. (2016) reported a mean source of 8.0 ± 4.7 t C-CO_2e ha^{-1} yr^{-1} for grasslands over peats and other organic soils.

Integrating trees in ruminant production systems favour C sequestration (Mutuo *et al*., 2005; Franzluebbers *et al*., 2017). The SOC content under wooded pastures is significantly higher than under tree-less pastures (Lorenz and Lal, 2014), and C is on average stored deeper and for longer (Howllet *et al*., 2011a,b). Carbon stocks in Atlantic and Mediterranean wooded pastures are up to 30% higher than in neighbouring tree-less pastures (Howllet *et al*., 2011a,b). On the contrary, woody encroachment of grasslands might lead to SOC losses on wet sites (Jackson *et al*., 2002). Recurrent fires cause losses of soil organic matter from the upper soil layer (Lal, 2004; Wardle *et al*., 2008). Thus, low fire-prone ecosystems such as sparsely wooded pastures can retain SOC for longer than fire-prone woody ecosystems in seasonally dry regions (Rigueiro-Rodríguez *et al*., 1999; Ruiz-Mirazo *et al*., 2011). An intermediate tree density therefore seems optimal for long-term C storage.

Carbon inputs to soil depend on grassland productivity, on the type and intensity of utilisation and on the form and amount of fertilisation (Soussana *et al*., 2004; Jones and Donnelli, 2004). Grassland C sequestration peaks at intermediate management intensity (Ward *et al*., 2016). Although there are divergent opinions on the effect of grazing on C sequestration, appropriate stocking rates and rotational grazing can potentially alleviate grassland degradation and increase C sequestration (Lal *et al*., 2015). Erosion, frequently associated with overgrazing, is a major cause of SOC loss (Diamini *et al*., 2016). Conant and Paustian (2002) estimated a potential C sequestration at a global scale of 45.7×10^6 t C yr^{-1} through rehabilitation of overgrazed pastures. Soussana *et al*. (2004) calculated a sustained GHG sink activity over five years only for low stocking rates with moderate N fertiliser inputs. This is in line with the results of Chen *et al*. (2015) who observed the highest SOC stocks under moderate grazing due to the largest root production and turnover. Grazing seems more beneficial than mowing for SOC storage up to a certain stocking rate (Rumpel *et al*., 2015; Koncz *et al*., 2017). At the field scale, Senapati *et al*. (2014) observed a six times larger net C storage under grazing than under mowing. In this experiment, however, the plant C excreted as dung by the livestock was not returned to the mown field.

Improved plant N nutrition increases C capture by the plants, which could benefit C sequestration. However, the positive effect of N fertilisation on the terrestrial C sink could, in terms of GHG balance, largely be offset by the resulting increase in CH_4 and N_2O emissions from the ecosystems (Liu and Greaver, 2009). In this respect, forage legumes could play a key role. First, they greatly reduce the need of N fertilisation, reducing the risk of N_2O emission and the CO_2 emitted during the production of mineral N fertiliser (Lüscher *et al*., 2014). Second, legumes can reinforce the capacity of soils to accumulate organic C as observed with the establishment of biodiverse grass-legume permanent grasslands in Portugal (Teixeira *et al*., 2015). In the arid and semi-arid regions of Portugal, soils are low in SOC and grasslands are typically low yielding and susceptible to fire-prone shrub encroachment. After replacement of the semi-natural grasslands with biodiverse grass-legume pastures, yields can increase twofold (Teixeira *et al*., 2015) and SOC content increases by 2.4 t C ha^{-1} yr^{-1} during the first 10 years (Teixeira *et al*., 2011). Finally, condensed tannins, frequent in legumes, can reduce enteric CH_4 emission (Mueller-Harvey *et al*., 2017).

Mitigating climate change through SOC sequestration is synergetic with agronomic benefits, like increased cation exchange capacity, soil fertility and water holding capacity, and decreased erosion (Conant and Paustian, 2002). SOC is hence favouring the provision of multiple ESS (Lal *et al*., 2015).

However, synergies can transform in antagonisms along the gradient of management intensity. For instance, while C sequestration and biodiversity conservation are both favoured by an intermediate rather than an intensive management, a further extensification supports biodiversity more but may reduce C sequestration. Also, C sequestration promoted by increasing plant productivity and/or tree cover could reduce water yield (Huxman *et al.*, 2005; Kim *et al.*, 2016).

Regulation of nutrient cycles

The effects of grassland management, fertilisation or grazing on the nutrient cycles have been dealt with in comprehensive reviews (e.g. Vertès *et al.*, 2018). Here we focus on the effects of the botanical composition. With respect to N, reaping the benefits of SNF by legumes is a key factor to increase N efficiency (harvested N/fertiliser N) in forage production. There is mounting evidence that this is better achieved with multispecies grass-legume mixtures than by sole cropping of legumes (Lüscher *et al.*, 2014). With grass-legume leys, the legume species and their relative abundance in the seed mixtures can be aligned to the targeted performances. Legume persistence is often poor in such mixtures (Brophy *et al.*, 2017), but this may only be a minor drawback for short-term leys (three years or less), because (1) a legume abundance as low as 30% suffices to achieve very high N efficiency (Suter *et al.*, 2015) and (2) the initial legume abundance still positively influences forage yield during the year following a strong decline in legume abundance (Brophy *et al.*, 2017). However, keeping a sufficient legume abundance for high N input from SNF in permanent grasslands is a major challenge (Phelan *et al.*, 2015). Regular grassland reseeding to rectify legume abundance might offset the benefits of legume-generated fertiliser savings in terms of emissions of reactive N forms. Indeed, depending on the method used and its timing, grass-to-grass reseeding can generate high NO_3^- (Conijn and Taube, 2004) and N_2O (Reinsch *et al.*, 2018) losses. Moreover, Necpálová *et al.* (2013) observed on a soil containing 4.5% organic C, that the loss of total soil N following grassland renovation after ploughing in summer was in the range of 3000 kg N ha^{-1}. Around 20 years would be necessary for this N amount to be symbiotically fixed by a mixed sward (estimation based on 25% legumes and a yield of 10 t DM ha^{-1} yr^{-1}). Overseeding instead of reseeding would help conserve N in the soil. Benefits of overseeding legumes on livestock weight gain have been shown (Graves *et al.*, 2012; Del Pino *et al.*, 2016), albeit short-lasting, but this method is quite unreliable (Phelan *et al.*, 2015). Improved strategies to manipulate legume abundance in permanent grasslands are therefore needed. Such strategies should include endeavouring to favour legume overwintering like cultivar selection and management optimising legume leaf area in late autumn (Lüscher *et al.*, 2001).

Whether the substitution of fertiliser N by SNF maintains or reduces N leaching from grasslands is still contentious (Lüscher *et al.*, 2014; Phelan *et al.*, 2015). However, at a minimum, for a similar level of N inputs (fertiliser + SNF), grass-legume mixtures improve crude protein yield without increasing N leaching compared to sole grass cropping (Nyfeler, 2009). There is some evidence that, in grass-legume mixtures, the companion grass is able to significantly reduce the leaching of inorganic N but not of dissolved organic N leaking from the legumes as compared to legume sole-crops (Kušlienė *et al.*, 2015). There are indications from the biodiversity experiment in Jena that working with higher levels of plant diversity could help reduce leaching of dissolved organic N (Leimer *et al.*, 2016). The number of species and the functional traits that would assist in that respect in productive grasslands are yet to be determined.

In addition to the complementarity between fixing and non-fixing species, complementarities in rooting depth or timing of N uptake may further improve N capture by multi-species swards under various climates (Joffre, 1990; Husse *et al.*, 2017). Thus, combining species differing in multiple features may further improve N capture by grass-legume mixtures. Intraspecific variability might also contribute to asynchronous N uptake, as large differences in precocity exist among varieties. This hypothesis is supported by the observed positive effects of genetic diversity (Prieto *et al.*, 2015). With respect to N efficiency, positive interactions appear more consistent between legumes and grasses than between legumes and forbs (Husse *et al*., 2017) or grass-legume and forbs (Cong *et al*., 2017). How much plant diversity itself affects N dynamics in the soil as compared to the presence of specific functional groups is still under debate (Mueller *et al*., 2013). Species developing a large and deep root system may also influence nutrient cycling by affecting soil hydrology. Macleod *et al.* (2012) observed a significant decrease in water runoff caused by the stronger root growth of a *Festulolium* cultivar as compared to *Lolium perenne*. Such traits would be particularly useful for grasslands having a critical location along the hydrological pathways (Gascuel-Odoux *et al*., 2009). Finally, plant traits can indirectly affect nutrient cycling by shaping soil microbial communities. Differences in root exudation might underlie these effects (Kaštovská *et al*., 2015) and plant species might interact with each other with respect to their influence on the soil microbial community (Zhao *et al*., 2017).

Evaluation of ESS from the field to the landscape scale

Multi-criteria evaluation of ESS

Synergies among ESS have been shown with biodiverse, legume-rich permanent pastures that enhance regulation and maintenance ESS (soil quality, C stocks, N availability) as well as provisioning ESS (Teixeira *et al*., 2015). However, ESS can be antagonistic to one another (Bennett *et al*., 2009; sections 2.2 to 2.4) thus, optimising the provision of multiple ESS requires a multi-criteria assessment followed by the development of trade-offs adapted to each individual situation (Howe *et al*., 2014). In agriculture, trade-offs have to also consider the supply of goods from the systems. For instance, it is well established that functional plant diversity can enhance provisioning ESS, but the supply of goods from grassland-based systems is greatly enhanced by agricultural intensification, which usually negatively affect plant diversity. Therefore, optimally, multi-criteria evaluations would assess both the level of provisioning ESS and the supply of goods. Unfortunately, grassland performances are often assessed based on the supply of goods alone because not enough data are generally available to quantitatively disentangle provisioning ESS from total yield.

Ultimately, trade-offs are worked out by farmers and decision-makers but they should be aided by decision support tools. One multi-criteria assessment method is Life Cycle Assessment (LCA). LCAs assess the environmental impacts of a good throughout its life cycle, from extraction of raw material to disposal (Hellweg and i Canals, 2014). While their initial focus was on the negative impacts of production rather than on ESS, impact assessment models in LCA are nowadays moving towards assessing ESS together with environmental damages (Othoniel *et al*., 2016; Nemecek *et al*., 2016). Although the field is still hindered with issues such as a lack of time-dependency of effects, as well as clear models of interrelationships among impacts, much progress has recently been made. There are now several LCA models to evaluate effects of the agricultural management on biodiversity (Teixeira *et al*., 2016) or soil quality (Legaz *et al*., 2017), in particular looking at SOC depletion and its influence on production (Morais *et al*., 2016). The main advantage of the LCA method is a unified, consistent framework to assess not only farm-level ESS, but also whole-chain ESS of any farm product. The quantification of the impact of entire production chains on ESS is nevertheless still subject to large uncertainties. A further remaining challenge with the evaluation of grassland-based systems is the fair consideration of the potential of the site for the cultivation of crop for direct human consumption (van Zanten *et al*., 2016) and of its carrying capacity.

In LCA, indicators are often weighted into aggregated results to facilitate decision-making. For instance, the aggregated indicator by Cao *et al.* (2015) quantifies the role of land use on six provisioning and regulating ESS: biotic production, fresh water recharge, erosion resistance, mechanical and physicochemical water filtration and climate regulation potential. The ESS are modelled in biophysical units and converted into monetary units based on economic valuation of ESS reduction. Monetary and non-monetary valuation of ESS are complicated by the still imprecise quantification of the ESS and by the patchy information available to the stakeholders about the gains associated with ESS (Costanza *et al.*, 2017). Although aggregated indicators can be helpful, they carry the risk of leading to decisions that fail to take the specificities of each situation into account. Indeed, the importance, respectively the valuation, of the different ESS clearly differs across situations (Bernués *et al.*, 2015). LCA models aggregate results using default weighing factors for each indicator or more advanced integration methods such as fuzzy logic (Agarski *et al.*, 2016). Another option is to weigh ESS using subjective and context-dependent value judgments of multiple stakeholders (Garrido *et al.*, 2017). Such participatory processes may only be locally applicable, but that may turn into an advantage. It means that when faced with the same trade-offs, different experts may decide, depending on the local conditions, which ESS should take priority and solve potential conflicts.

Mathematical methods to simultaneously analyse multiple ESS at the field scale have recently been developed (Dooley *et al.*, 2015). But much work remains to be done, as the evaluation of some ESS requires cross-scales analyses (Duru *et al.*, 2018). To our knowledge, few studies have assessed the effect of grassland management on multiple ESS. Pan *et al.* (2014) used the balance between grazed and total primary production to discuss antagonisms between meat production, C sequestration and water conservation. However, they did not consider changes in grassland productivity due to management changes. Dumont *et al.* (2016) discussed options to improve the delivery of multiple ESS by livestock production systems and highlighted large differences in ESS delivery among production systems. Yet, the main scale of their analysis was the whole production system and management options at the grassland level were not analysed for multiple ESS simultaneously.

Landscape scale evaluation of ESS

ESS can be supplied at different spatial scales and the beneficiaries of the ESS vary with the considered scale (e.g. local population for water quality regulation, the whole society for climate change mitigation). The landscape is one of the important scales to consider for studying ESS. For ecologists, a landscape is the level of ecological organisation characterised by a specific arrangement of ecosystems (Burel and Baudry, 1999). In social sciences, a landscape is rather considered as a social construction as perceived by a group and produced by economic and cultural practices (Bertrand and Beruchavili, 1978). It thus has a perceptive, aesthetical meaning. Its spatial structure reflects the relations between societies and their environment (Burel and Baudry, 1999). Studying the supply of ESS at the landscape scale is crucial for three main reasons. First, ESS supply by grasslands not only depends on field-scale parameters, but also on farm-scale management (i.e. other available fields, farm structure and strategy) and on the surrounding landscape elements. For instance, ESS with respect to water regulation or pollination depend on the position of the grasslands along the hydrological pathways or within the habitat network, respectively (Kremen *et al.*, 2007). Moreover, relationships between ESS vary across a landscape because of differences in land use and pedo-climatic conditions within this landscape (Li *et al.*, 2017). Second, some ESS, such as recreational opportunities, aesthetic enjoyment or spiritual benefits are mostly generated at the landscape scale (Harrison *et al.*, 2010; Fagerholm *et al.*, 2016). These cultural ESS are very important in grassland-dominated landscapes (Garrido *et al.*, 2017). Grassland landscapes are usually appreciated, for they are perceived more natural than cropland landscapes (Bugalho and Abreu, 2008). In addition, a heterogeneous agricultural landscape is generally more appreciated than less diverse landscapes (Hahn *et al.*, 2017). Third, as a consequence of the two first points, the landscape, interface between the society and its environment, constitutes a privileged scale to incorporate ESS in management strategies and decision making that include a large spectrum of stakeholders (farmers, other economic branches bounded to nature, authorities, citizens), especially through quantifications and mapping procedures.

Quantifying, valuing and mapping ESS is seen as a way to support policy making (Daily and Matson, 2008), especially for the evaluation of trade-offs (Farber *et al.*, 2002). In order to optimise grasslands

ESS supply, landscape-scale spatial analyses are crucial for two main reasons. First, they help to identify hotspots of ESS supply (Figure 2), allow depicting transition areas between grasslands and non-farm habitats (Figure 2), and support the prioritisation of management objectives considering spatially more nuanced criteria. This provides baselines for adjustments of agricultural policies and incentives. Second, they may take into account the structure and features of the landscape (e.g. diversity, openness, connectivity or fragmentation) that affect some ESS and need to be considered in political programs (e.g. financial support of the Swiss federal and regional authorities to implement habitat connectivity measures). Such programs are, however, difficult to evaluate with respect to their effects on multiple ESS (Angelone and Holderegger, 2009) and only few scientific studies have yet considered the potential of landscape-scale approaches on ESS supply (Hodder *et al.*, 2014). Nevertheless, this field is developing quickly. Li *et al.* (2017) for instance, presented a method for the evaluation of the relationship between two ESS at the landscape scale. This method is based on a spatially explicit quantification of multiple ESS at the landscape scale, followed by a pairwise overlay of the ESS on each pixel and a partial correlation analysis. Such approaches will greatly support the evaluation of multiple ESS at larger scales.

Conclusion

Management intensity is a major driver of the delivery of goods and ESS from grasslands. From the available body of literature, we conclude that the supply of goods, C storage and biodiversity conservation have their optimum at different levels of grassland intensification (Figure 3a). Hence, these ESS can generally not all be maximised on the same individual field. At the field scale, various specific options exist to improve the delivery of individual ESS. Some of these options can be beneficial to several ESS simultaneously, like promoting an optimal plant functional diversity, and can be combined. For instance, planting trees to promote C storage, keeping uncut refuges for biodiversity and establishing N_2-fixing legumes could be combined within a single field (Figure 3b). An optimal legume abundance in the sward seems particularly promising to favour multiple ESS. Nevertheless, none of these options allow for alleviating the conflicts occurring along the intensification gradient and at the field scale, an intermediate intensity implies mediocre performances for both biodiversity conservation and the production of high-quality forage (Figure 3a, 3b). The current state of evidence indicates that this challenge must be addressed at the farm or landscape scale. By combining various management intensities and targets involving different types of grasslands at the farm scale (Figure 3c), it seems possible to better reconcile production with biodiversity conservation and pollination than by uniformly managing all fields of the farm at intermediate level of intensity (Figure 3d). We conclude that multiple ESS by grassland-based systems can be optimised by combining 1) specific improvement measures at the field scale with 2) heterogeneity

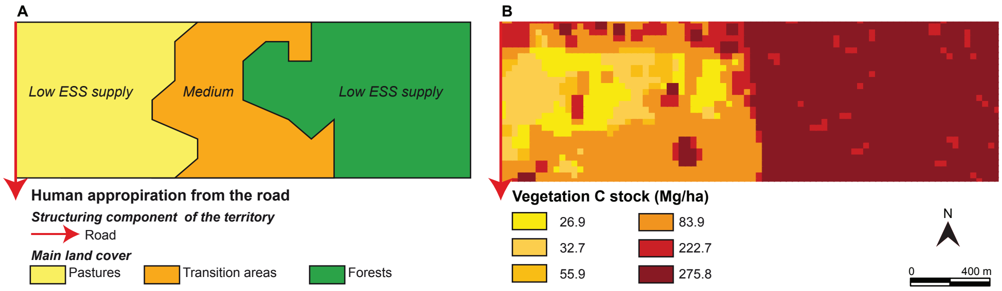

Figure 2. Spatial structure of ESS indicators in a 200 ha landscape of the Brazilian Amazon (landscape including two farms). A gradient in ESS supply can be distinguished from the road, which is the central axis of deforestation. (A) General trend for 6 ESS indicators based on statistical modelling using remote sensing and field data and (B) for the same landscape, detailed mapping of vegetation carbon stock. Adapted from Le Clec'h (2017).

among grassland types at the farm or landscape scale. This heterogeneity would target the right balance between fields managed at very low, intermediate and high intensity, and optimise the spatial location of the different grassland types in accordance with the surrounding nonfarm habitats (Figures 3c and 3d). The high intensity must, of course, remain within the carrying capacity of the site. For C storage, this strategy might be less favourable than an intermediate intensity over the whole farm area, but would allow the integration of optimal elements, like wooded pastures of intermediate management intensity, grazed for instance by replacement heifers. Multi-criteria evaluation tools to quantify ESS and weigh antagonistic services are currently being refined. Methodological advancement in this field is critical to develop multi-scale strategies for multiple ESS.

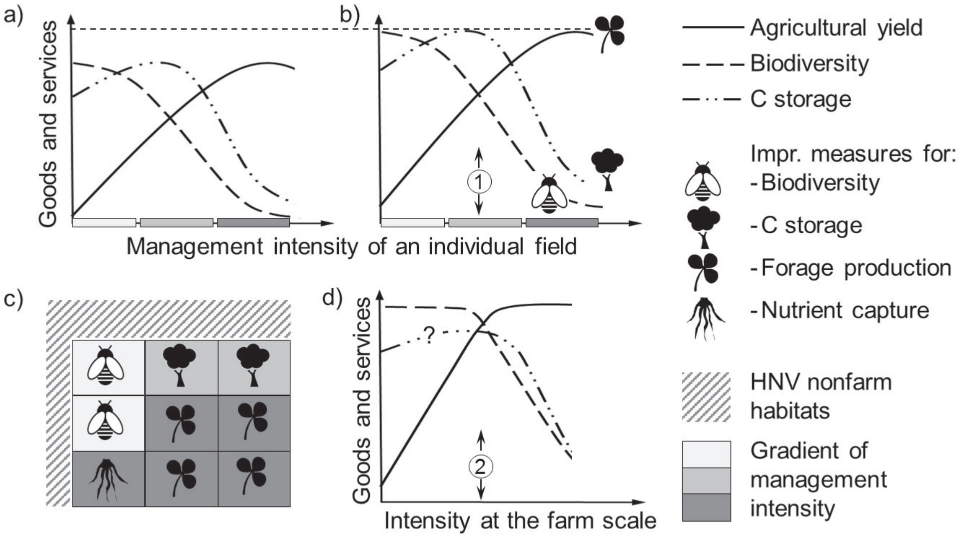

Figure 3. Schematic summary of the effects of grassland management intensity on the delivery of goods and services (a) at the field scale without specific measures, (b) at the field scale with improvement measures, and (d) for the average intensity at the farm scale. Panel (c) illustrates the concept of grassland heterogeneity at the farm scale combined with specific measures at the field scale, which might result in improved multifunctionality for an intermediate average level of intensity at the farm scale (panel d). The main target for each field takes the surrounding nonfarm habitats into consideration (HNV: high nature value). At the field scale, an intermediate intensity level promotes C storage but performs quite poorly in term of agricultural yield and biodiversity. This is expected to be similar for a farm having all its fields uniformly managed at intermediate intensity. A farm combining extensive to intensive grasslands (panel c) has an intermediate average level of intensity at the farm scale but maintains fields of high biodiversity (extensive grasslands), intermediate grasslands for C storage and grazing of less demanding livestock and fields for the production of energy- and protein- rich forage (intensive grasslands). Such a farm would, therefore, supply multiple ESS.

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