Connecting CSR theory and LPJmL 5.3 to assess the role of environmental conditions, management and functional diversity for grassland ecosystem functions

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Abstract. Forage supply and soil organic carbon storage are two important ecosystem functions of permanent grasslands, which are determined by climatic conditions, management and functional diversity. However, functional diversity is not independent of climate and management, and it is important to understand the role of functional diversity and these dependencies for ecosystem functions of permanent grasslands. Especially since functional diversity may play a key role in mediating impacts of changing conditions. Large-scale ecosystem models are used to assess ecosystem functions within a consistent framework for multiple climate and management scenarios. However, large-scale models of permanent grasslands rarely consider functional diversity. We implemented a representation of functional diversity based on the CSR theory and the global spectrum of plant form and function into the LPJmL dynamic global vegetation model forming LPJmL-CSR. Using a Bayesian calibration method, we parameterised new plant functional types and used these to assess forage supply, soil organic carbon storage and community composition of three permanent grassland sites. These are a temperate grassland, a hot and a cold steppe for which we simulated several management scenarios with different defoliation intensities and resource limitations. LPJmL-CSR captured the grassland dynamics well under observed conditions and showed improved results for forage supply and/or SOC compared to LPJmL 5.3 at three grassland sites. Furthermore, LPJmL-CSR was able to reproduce the trade-offs associated with the global spectrum of plant form and function and similar strategies emerged independent of the site specific conditions (e.g. the C- and R-PFTs were more resource exploitative than S-PFTs). Under different resource limitations, we observed a shift of the community composition. At the hot steppe for example, irrigation led to a more balanced community composition with similar C-, S- and R-PFT shares of above-ground biomass. Our results show, that LPJmL-CSR allows for explicit analysis of the adaptation of grassland vegetation to changing conditions while explicitly considering functional diversity. The implemented

mechanisms and trade-offs are universally applicable paving the way for large-scale application. Applying LPJmL-CSR for
20   different climate change and functional diversity scenarios may generate a range of future grassland productivity.

1   Introduction

Permanent grasslands have various important ecosystem functions, one of which is their role as a source of feed for livestock -
in the following defined as forage supply - across the globe (White et al., 2000). Another function is their soil organic carbon
(SOC) storage which has the potential to contribute to climate change mitigation (e.g. Godde et al., 2020; Yang et al., 2019).
25   These two important ecosystem functions of permanent grasslands depend on the climatic conditions, soil properties, manage-
ment and functional diversity. The climatic conditions and soil properties determine the availability of important resources for
photosynthesis and plant growth. While irrigation and fertiliser management are applied to increase the availability of specific
resources and thereby productivity, grazing or mowing remove biomass which can affect leaf and root growth and SOC stocks
(Bai and Cotrufo, 2022; Conant et al., 2017). Even though functional diversity of the vegetation is not an independent factor but
30   depends on environmental conditions (Fei et al., 2018; Grime, 2001) and management (Guo, 2007), it also affects forage supply
and SOC (Yang et al., 2019; Chen et al., 2018). Furthermore, functional diversity plays an important role for the resistance
and resilience of an ecosystem towards the impacts of changing conditions and might be essential to maintain the ecosystem
functions of permanent grasslands under climate change (Isbell et al., 2015). Therefore, it is important to understand the role
of functional diversity in permanent grasslands and its role for ecosystem functions such as forage supply and SOC storage.

35   ### 1.1   The role of environmental conditions and management for grassland vegetation and SOC storage

Forage supply, SOC and community composition are dependent on environmental conditions and management. Important
factors for plant growth are radiation, temperature, water and nutrient supply. Provided with sufficient availability of water
and nutrients, grasslands can produce large amounts of biomass, while drought and nutrient stress lead to lower productivity.
Since large amounts of biomass can lead to high carbon inputs, this highlights the importance of temperature and precipitation
40   for SOC storage (Wiesmeier et al., 2019). High precipitation also favours the formation of SOC-stabilising mineral surfaces
(Doetterl et al., 2016; Chaplot et al., 2010) and leads to lower decomposition rates (Meier and Leuschner, 2010). On the other
hand, high temperatures can lead to an increase of microbial decomposition and a decrease in SOC stock (e.g. Koven et al.,
2017; Sleutel et al., 2007). Highest SOC stocks are generally found in cool humid climates but decrease towards warmer and
drier climates (Jobbágy and Jackson, 2000). Additionally, removal of above-ground biomass through grazing or mowing may be
45   beneficial for grassland productivity depending on its intensity (Oesterheld and Loreti, 1999; Semmartin and Oesterheld, 1996;
Milchunas and Lauenroth, 1993). However, mowing and grazing also affect the below-ground biomass and highly intensive
management may lead to overgrazing and cause SOC loss (McSherry and Ritchie, 2013). Still, global meta-analyses of grazing
effects on SOC did not find uniform trends (McSherry and Ritchie, 2013; Piñeiro et al., 2010).

Together, the environmental factors and the management act as filters for the species that are best suited for the specific
50   conditions. Changes in management or climatic and soil conditions may alter this filtering process and lead to the selection of

different strategies either indirectly through alterations of the resource limitations (e.g. Yu et al., 2015; Tilman and El Haddi, 1992) or directly in the case of management by manipulating the species pool through reseeding and weeding (Weisser et al., 2017) or selective grazing (e.g. Wan et al., 2015).

1.2 Functional diversity and ecological strategies

55 Functionally diverse ecosystems contain species that follow different ecological strategies and can be described through a representation of these strategies. We define ecological strategy as the means a plant or species uses to occupy a certain habitat. Plants have evolved a range of different ecological strategies that make different species perform better or worse than others in different habitats. Functional diversity which underpins robustness against environmental and management change of certain ecosystem functions is related to the presence or absence of specific strategies. For example, a community in which multiple

60 strategies are present is less vulnerable to fluctuations or changes in environmental conditions or management. To distinguish between different ecological strategies, several classification schemes have been developed (examples). The competitor, stress-tolerator, ruderal (CSR) theory (Grime, 2001; Campbell and Grime, 1992; Grime, 1977), distinguishes three main strategies: Competitive (C), stress tolerant (S) and ruderal (R) strategies can be placed at the nodes of a triangle, while intermediate strategies are placed in between. This scheme can be used to classify the overall strategy of a community (e.g. Caccianiga

65 et al., 2006) as well as the strategies of single species (e.g. Grime, 1974). These overall strategies are associated with different plant behaviour. C species are efficient resource users and grow fast but do not deal well with resource limitations or frequent disturbances. Opposite are S species which invest resources in more robust tissue which grows slower but enables them to cope with resource limitations. While both C and S species are vulnerable towards disturbance, R species use the time frames between disturbances to complete their life cycle and have an advantage in disturbance prone environments. This different

70 behaviour is expressed through different trait values which in turn can be used to classify plants according to the CSR theory. A prominent example is the global spectrum of plant form and function which explains differences in ecosystem function using traits related to growth economics, stature and life cycle (Díaz et al., 2016) and has been combined with the CSR theory and applied to single but also multi species communities (Pierce et al., 2017, 2013). Additionally, several other CSR analysis methods have been developed (Hodgson et al., 1999; Grime et al., 1988) and applied to compare vegetation function

75 (e.g. Schmidtlein et al., 2012; Hunt et al., 2004) and to assess various community processes (Pierce et al., 2017) for example resistance, resilience and coexistence (Lepš et al., 1982), succession (Caccianiga et al., 2006), and the biodiversity-productivity relationship (Cerabolini et al., 2016). Pierce et al. (2017) provided a method to compare the CSR strategies of vascular plants at the global scale which is useful to assess community assembly in different environments. However, additional methods are needed to also predict ecosystem functioning of the assembled communities.

80 1.3 Modelling ecosystem functions of permanent grasslands

To assess forage supply and SOC storage of permanent grasslands under different environmental conditions and management, models of grassland dynamics can be useful tools (e.g. Jebari et al., 2022; Chang et al., 2021; Rolinski et al., 2018). Models at the community and plot scale that incorporate very detailed approaches to simulate functional diversity in a specific context

already exist (e.g. Schmid et al., 2021; May et al., 2009). In contrast, large-scale vegetation models generally use a very simple
representation of the community and do not consider the trade-offs described by the global spectrum of plant form and function
(Díaz et al., 2016) at all or only partially (Pfeiffer et al., 2019; Sakschewski et al., 2015, e.g.). However, large-scale models
provide the means to assess functional diversity in a wide range of environmental conditions and management interventions
to improve projections of ecosystems functions under future climate change (e.g. Herzfeld et al., 2021; Sitch et al., 2008).
In addition, such models could be useful to improve knowledge on the mechanisms underlying the *global spectrum of plant*
form and function and help better distinguish local variability from large-scale patterns. To overcome current limitations of
large-scale models, simplifications such as the CSR theory provide the opportunity to incorporate ecological strategies and
functional diversity into large-scale models.

The dynamic global vegetation model (DGVM) LPJmL contains a detailed soil module and is able to simulate different
grazing or mowing management (Rolinski et al., 2018), irrigation (Schaphoff et al., 2018), application of manure and synthetic
fertiliser (von Bloh et al., 2018) and tillage (Lutz et al., 2019). The CSR strategies and their relationship to specific plant traits
provide a simple way to incorporate functional diversity into the LPJmL model to include its effects in the assessment of forage
supply and SOC storage of grasslands for different environmental conditions and management. To this end, we implemented
the trade-off associated with the three main strategies of the CSR theory (Grime, 1977) for managed grasslands in LPJmL
using the *global spectrum of plant form and function* (Díaz et al., 2016) to assess:

– how important functional diversity is for forage supply and SOC dynamics in different climates and under different
 management regimes.

– how changing resource limitations affect forage supply, SOC and community composition.

2 Methods

We conducted our assessment at three permanent grassland sites in different climates: a temperate meadow in northern Germany
with favourable climatic conditions for grassland productivity, as well as a hot steppe pasture in South Africa and a cold steppe
pasture in Inner Mongolia (China) with less favourable climatic conditions. Throughout the remaining manuscript we refer to
the sites as temperate grassland, cold and hot steppe. At each site, we assessed two levels of management intensity which either
differed with respect to the amount of fertiliser (temperate grassland) applied or the defoliation intensity (hot and cold steppe).

We extended LPJmL to account for trade-offs between C-, S- and R-plant species as described by the CSR theory (Grime,
1977) using functional traits. We used two strategy axes to distinguish these three strategies: First, we distinguished between
acquisitive (C and R) and conservative (S) strategies using resource economics. Second, we used reproduction strategies and
stature to distinguish between large investments in reproduction but a small stature (R) and small investment in reproduction
with a wide range of statures (C and S). Both strategy axes are expressed through several model parameters (Sect. 2.4).

To represent the different strategies, we parameterised three herbaceous PFTs - one competitive (C-PFT), one stress tolerant
(S-PFT) and one ruderal (R-PFT) - for each site and management intensity. Strategies that are in between these three main

strategies (e.g. competitive ruderal or stress tolerant ruderal) were not reflected by additional PFTs but should be reflected in the share of the main strategy (e.g. if a competitive ruderal strategy is advantageous in an environment, this results in a higher share of the competitive and the ruderal PFT). We evaluated the new implementation in the following referred to as LPJmL-CSR against forage supply and SOC observations for the different sites.

120 ## 2.1 Overview of managed grasslands in LPJmL

We extended the LPJmL model version 5.3 (LPJmL 5.3), which already included the representation of managed grasslands using a daily allocation scheme (Schaphoff et al., 2018), four different management options (Rolinski et al., 2018) and the nitrogen cycle (von Bloh et al., 2018). In this model version the dynamics of a grassland were simulated using three PFTs: one polar, one temperate and one tropical grass, which were constrained to the respective climatic regions by bio-climatic limits. As

125 a consequence, all grasslands that are not located at the border between climatic regions were simulated using only one of these PFTs to represent herbaceous vegetation. Each PFT was represented by one average plant individuum for which dynamics were simulated with daily time-steps. Processes simulated were: 1) establishment of new PFTs and reproduction of established PFTs (Sect. 2.3.3), 2) biomass accumulation calculated from gross primary production (GPP) and autotrophic respiration limited by environmental conditions, and 3) plant turnover. Different management options are available for irrigation, fertilisation

130 and grazing or mowing. Irrigation options used here are no irrigation (rainfed) or potential irrigation (no water limitation, ref). Manure fertilisation options were adapted from the crop module (see SI) and include the amount and timing of manure application. Manure application can be split over several treatments. In grazed grasslands, 25% of the grazed carbon (Rolinski et al., 2018) and 50% of the nitrogen are returned to the soil as dung or urine of the grazing animals (Huhtanen et al., 2008).

2.2 Site description

135 We conducted our assessment at three different sites (Fig. SI 1) which are located in different biomes with substantial differences in precipitation and temperature, covering the warm temperate fully humid (temperate grassland), the arid hot steppe (hot steppe) and the arid cold steppe climate (cold steppe) (Kottek et al., 2006) and are subject to different management intensities (Table 1).

The temperate grassland is located in favourable climatic conditions and provides high forage supply. The vegetation is dom-

140 inated by C strategists with marginal shares of S and R. It is cut four times each year in May, July, August and September. Data on two experiments were available: an unfertilised (N0) control and a fertilised (N1) treatment with 240 $\mathrm{kgN \cdot ha^{-1} \cdot year^{-1}}$ in the form of cattle manure split over four applications at the beginning of the growing season and after the first three cuts (Reinsch et al., 2018a, b).

Arid conditions lead to a lower forage supply for the hot steppe. S-strategists dominate the vegetation here, while the R-

145 strategy is subordinate and the C-strategy is only marginally present. Data for an ungrazed (C0) control and a rotationally (C1) grazed experiment with a livestock density of 0.1 cows per hectare with a body weight of around 450 kg were available (Munjonji et al., 2020).

Table 1. Overview of the environmental conditions and management of the investigated grasslands. [†]Values are observed leaf biomass and not grazing offtake.

Site	Temperate grassland		Hot steppe		Cold steppe	
Location	Lindhof, Germany		Syferkuil, South Africa		Xilin, Mongolia	
Coordinates	$54°27'$N, $9°57'$E		$23°85'$S, $29°7'$E		$43°38'$N, $116°42'$E	
Mean annual temperature [°C]	9.4		20.5		0.9	
Mean annual precipitation [mm]	746		432		329	
Koeppen-Geiger class	Cfb		BSh		BSk	
Soil type	Sandy loam		Loamy sand		Sandy clay loam	
Management	Fertilisation		Cattle grazing		Sheep grazing	
Experiment	unfertilised	fertilised	ungrazed	grazed	low intensity	high intensity
Forage supply [MgDM ha^{-1}]	7.9±1.6	9.2±2.0	1.1±0.6[†]	1.5±0.6[†]	0.4±0.3	0.6±0.2
SOC depth [m]	0.3		0.3		1	
SOC value [MgC ha^{-1}]	69.7±3.7	71.9±3.4	36±20		273±60	
Literature	(DWD 2021, Reinsch et al. 2018a, b)		(Munjonji et al. 2020)		(Hoffmann et al. 2016, Ren et al. 2017; Wiesmeier et al. 2011)	

As a result of the low precipitation and temperatures, the cold steppe is least productive. Similar to the hot steppe, the S-strategy is dominant and C- as well as R-strategists have marginal shares. We used data of experiments with two different
150 livestock densities of grazing sheep with a body weight of around 35 kg: the low grazing intensity (S1) of 1.5 sheep ha^{-1} and the high grazing intensity (S6) with 9 sheep ha^{-1} (Hoffmann et al., 2016).

2.3 Model development

To extend the LPJmL model to simulate different communities in which different ecological strategies are advantageous, we focused on three aspects: First, we adapted resource uptake and distribution (Sect. 2.3.1) to improve niche differentiation (see
155 Hardin, 1960). Second, we implemented the trade-off between fast tissue growth at low construction cost and longevity versus slow tissue growth at high construction cost and longevity described by the leaf economic spectrum (Sect. 2.3.2 Wright et al., 2004). Third, we altered the representation of the plants' life cycle (Sect. 2.3.3) to distinguish different reproductive strategies. We provide a qualitative description of the aspects of recent model development that are important for LPJmL-CSR in the main text and refer to the Appendix A and SI for the technical details and other minor improvements compared to the original code.

160 ### 2.3.1 Resource uptake and distribution

In the LPJmL model, the different PFTs compete for space/light, water and nitrogen. In past model versions these resources were distributed between PFTs dependent on their foliage projective cover (FPC). The FPC is used as a proxy for actual cover,

which would require the explicit simulation of the plants' geometries. Distributing these different resources based on one variable neglected the importance of different traits for the uptake of different resources. In particular, water uptake should also

165   be dependent on root traits such as the extent of the root network and the amount of fine root biomass (Tron et al., 2015). Using root traits to determine access to water enables the model to simulate different strategies for water resource use. Therefore, we adapted the implementation of water supply to make it dependent on root biomass instead of FPC to provide a distinction between the criteria for above-ground and below-ground resource uptake and distribution. Based on the concept of the FPC we implemented a below-ground equivalent based on root instead of leaf biomass (A1). The new parameter (k_{root}) is a proxy for

170   root properties associated with morphological properties of the root network (e.g. branching and spread).

2.3.2   The leaf economic spectrum

The trade-off described by the leaf economic spectrum (LES) is mainly associated with the traits *Leaf N concentration* and *specific leaf area* (SLA) (Wright et al., 2004). In LPJmL, both traits are used to calculate the photosynthetic activity. The leaf nitrogen concentration is a function of the assimilated carbon and nitrogen and can vary within a PFT-specific range (von

175   Bloh et al., 2018) to account for the growth under different resource availabilities. The SLA is used to calculate the leaf area index (LAI) from the dynamically computed leaf biomass, which is important for the interception of light energy and thus for photosynthesis. While this implementation was sufficient to represent the growth or resource acquisition component of the LES, it did not represent the leaf longevity trade-off. Instead leaf longevity was the same for all PFTs independent of their SLA. Therefore, a PFT with a high SLA corresponding to a resource acquisitive strategy had an advantage even under

180   resource limited conditions, because its competitor with a lower SLA had the same leaf longevity and could not outlive the acquisitive PFT. However, slow growing, resource conservative plants of stress prone ecosystems supply less forage with a lower nutrient content (Lee, 2018; Onoda et al., 2017). Additionally, such ecosystems are more vulnerable to overgrazing (Liu et al., 2013) and recover slower from disturbances (Teng et al., 2020) which is an important property when simulating stressed grasslands. Representing the SLA leaf longevity trade-off was thus essential to incorporate different ecological strategies in

185   LPJmL-CSR. We derived a power law for SLA and leaf longevity from trait data retrieved from the TRY database (Boenisch and Kattge, 2018), which is used to calculate the PFT-specific leaf longevity (A2). Based on the alignment of the resource conservation axis of the root economic space (Bergmann et al., 2020) and the LES (Weigelt et al., 2021), we assume that leaf and root longevity are not independent from each other and maintain a fixed relationship between the two in LPJmL-CSR.

2.3.3   Reproduction and mortality

190   Herbaceous plants are adapted to different growing conditions and therefore have different reproduction strategies and whole plant - or for graminoids phytomere - longevity. In LPJmL, each herbaceous PFT was simulated using only one average individual with specified properties. Age mortality was implicitly included in the representation of turnover of leaves and roots and not as a separate process. Furthermore, a biomass increase of the average individual dependent on the available area was used to simulate reproduction. We argue that this was not sufficient to simulate different reproduction strategies, which differ in

195   the amount of seeds, seed survival and germination rates, and germination requirements (Thompson, 1987; Brown and Venable, 1986).

In the representation of CSR strategies in LPJmL-CSR, we retained the approach of establishing saplings instead of seeds but allow PFTs to establish different numbers of saplings in agreement with their reproductive strategy. To achieve this, we abandoned the approach of using only one average individual to simulate each PFT and introduced a dynamic number of

200   average individuals that share the same properties but form the community together. Based on the existing implementation, we modified the reproduction so that additional individuals are established and thereby increase the number of average individuals simulated. Since space for plant establishment is limited, we prohibit infinite growth of the number of average individuals by adding an age dependent individual mortality to reduce the number of average individuals (A3). While the new approach does not simulate individual and phytomere morphology explicitly, it provides some implicit information on community structure

205   and plant size through the number of average individuals, the area covered by them and their biomass. It can be assumed that few individuals that maintain a high cover and biomass must be larger than more individuals that provide a similar cover and biomass.

2.4   Defining the C, S and R-PFT

We based our new PFTs on the already existing herbaceous PFTs (Schaphoff et al., 2018), from which we inherited the majority

210   of parameter values. For the temperate grassland we used the temperate herbaceous, for the hot steppe the tropical herbaceous and for the cold steppe, the polar herbaceous PFT. To design the new C-, S- and R-PFTs for each of these environments and given management scenarios, we assessed a subset of parameters that define our trait space using two trade-off dimensions represented by functional traits inspired by the *global spectrum of plant form and function* (Díaz et al., 2016) to distinguish the CSR-strategies: The abiotic dimension and the biotic dimension. Based on past sensitivity analyses (Forkel et al., 2019;

215   Zaehle et al., 2005) and expected behaviour of newly implemented trade-offs, we selected four parameters for each dimension to distinguish the CSR-strategies (Tab. 2).

2.4.1   The abiotic and biotic dimensions

We assumed, that the position of a species within the CSR triangle can be determined through trade-offs between functional plant traits within the abiotic and the biotic dimension according to the following relations:

220   According to CSR theory, the stress gradient expresses the level of stress a species is exposed to in a certain habitat. It ranges from unstressed to severely stressed, but does not distinguish individual stress categories (e.g. temperature, water or nutrient). Different traits and their values are associated with the ability of a plant to cope with the different stress levels. The traits of the leaf economic spectrum (LES, Wright et al., 2004) together with different strategies for water-resource use can be used to distinguish C- and R-strategists (low stress tolerance) from S-strategists (high stress tolerance). To represent the stress

225   gradient, we used functional traits associated with resource use, which we defined as the abiotic dimension of our trait space. To represent the abiotic dimension, we selected the maximum transpiration rate (E_{max}), the minimum canopy conductance (g_{min}), the specific leaf area (SLA) and the leaf to root mass ratio ($lmro$). The maximum transpiration rate (E_{max}) and the

Table 2. Parameter names, units, ranges and criteria for the hierarchy for the C-, S- and R-PFTs.

Parameter	Abbreviation	Unit	Min	Max	Dimension	Hierarchy
Specific leaf area	SLA	[$\mathrm{m^2 \cdot gC^{-1}}$]	0.01	0.1	abiotic	S < C < R
Light extinction coefficient	k_{beer}	[-]	0.2	0.8	biotic	R < C & S
Establishment rate	k_{est}	[$\mathrm{Ind. \cdot m^{-2} day^{-1}}$]	3000	6000	biotic	R > C & S
Root efficiency coefficient	k_{root}	[-]	0.005	0.025	biotic	R < S < C
Leaf to root mass ratio	$lmro$	[-]	0.6	1	abiotic	S < R < C
Maximum transpiration rate	E_{max}	[mm]	4	12	abiotic	S < R < C
Sapling leaf area index	LAI_{sapl}	[-]	0.01	0.15	biotic	R < S < C
Minimum canopy conductance	g_{min}	[$\mathrm{mm \cdot s^{-1}}$]	0.3	2	abiotic	S < R < C

minimum canopy conductance (g_{min}) determine the amount of water transpired and distinguish between strategies that save more or less water. The SLA is used to determine the trade-off between short-lived, acquisitive and long-lived, conservative

230 leaves. The leaf to root mass ratio ($lmro$) describes the investments into above- vs. below-ground biomass and reflects the resulting plant specific resource availability.

Similar to the stress gradient, the disturbance gradient ranges from undisturbed to severely disturbed. Reproductive traits and plant stature (Westoby et al., 1996; Grime, 1974; Salisbury, 1943) can be used to distinguish C- and S-strategists (low disturbance tolerance) from R-strategists (high disturbance tolerance). Functional traits associated with reproduction and plant

235 geometry can be used to represent the trade-off associated with the disturbance gradient. We selected the following functional traits involved in the direct interaction of the different PFTs, which we defined as the biotic dimension: The root efficiency coefficient (k_{root}), the light extinction coefficient (k_{beer}), the establishment rate (k_{est}) and the leaf area index of a sapling (LAI_{sapl}). We introduced the root efficiency coefficient (k_{root}) as a substitute for information on root functional traits such as branching and density of the root network to account not only for root biomass but also the below-ground morphology

240 of different species. The light extinction coefficient (k_{beer}) is a determinant for shading and used to distinguish large from small stature plants. The establishment rate (k_{est}) reflects the potential amount of offspring and the leaf area index of a sapling (LAI_{sapl}) represents the offspring size.

In total, we used eight parameters to distinguish the strategists for each PFT and defined plausible ranges for their parameterisation.

245 **2.4.2 Parameterisation and evaluation of new PFTs**

To parameterise the new PFTs we had to assess the model performance for different parameter sets. We included several variables in the calculation of a likelihood ($logLI$): forage supply, SOC and C-, S- and R-strategy cover (Table 3). Data on forage supply and SOC were available from several field experiments conducted at the respective sites. For C-, S- and R-PFT

Table 3. Variables used for parameterisation (para) and evaluation (eval) of the new PFTs at the study sites

Variable	Site	Resolution	Usage	Source	Literature
Dry matter yield	Temp. grassland	per cut	para/eval	field observations	(Reinsch et al., 2018b)
Soil carbon	Temp. grassland	annual	para/eval	field observations	(Reinsch et al., 2018b)
Cover of C-, S- and R-PFT	Temp. grassland	constant	para	expert estimate	-
Leaf biomass	Hot steppe	annual	para/eval	field observations	(Munjonji et al., 2020)
Soil carbon	Hot steppe	monthly	eval	field observations	(Munjonji et al., 2020)
Cover of C-, S- and R-PFT	Hot steppe	constant	para	expert estimate	-
Leaf biomass	Cold steppe	monthly	para	field observations	(Schönbach et al., 2012)
Grazing offtake	Cold steppe	monthly	eval	field observations	(Schönbach et al., 2012)
Soil carbon	Cold steppe	constant	para	field observations	(Wiesmeier et al., 2011)
Cover of C-, S- and R-PFT	Cold steppe	constant	para	expert estimate	-

cover, data were only available for the hot steppe and we defined values based on our knowledge of the site specific conditions
250   to agree with CSR theory for the other sites.

We optimised the $logLI$ using a Markov Chain Monte Carlo (MCMC) method with a Metropolis algorithm (Wirth et al., 2021; Van Oijen et al., 2005). This method evaluates the performance of a sequence of sampled parameter sets. In the following, we refer to a sequence as a *chain* and to an iteration as a *link*. At the beginning of the chain, a first parameter set is drawn from a multivariate Gaussian distribution with its modes at the centre of the parameter ranges for each parameter and its variances as
255   a fraction of the parameter ranges. A fraction of the ranges is used to limit the difference between parameter sets of subsequent links which improves the performance of the algorithm. The width of this fraction is controlled through a tuning parameter and is fixed for the entire chain, while the modes of the Gaussian distribution are updated throughout the chain if the model performance calculated as the total likelihood ($logLI$, Eq. 1) improves.

$$logLI_i = logPrior_i + logLink_i \tag{1}$$

260   The total likelihood $LogLI$ is calculated for each link i. It consists of a prior likelihood ($logPrior_i$, Eq. 2) and the likelihood of the current link ($logLink_i$, Eq. 3).

$$logPrior_i = \sum_j B(\theta_{i,j}, p, q) \tag{2}$$

The prior likelihood $logPrior$ is calculated from the prior distribution, which represents an initial guess on the resulting posterior distribution. We chose a geometrical prior distribution (B) with the shape parameters $p = 1 + 4 \cdot (\hat{\theta}_{centre} - \hat{\theta}_{min} / \hat{\theta}_{max} - $
265   $\hat{\theta}_{min})$ and $q = 6 - p$. Here, $\theta_{i,j}$ are the parameter values of each parameter j of the current link i, $\hat{\theta}_{centre}$ are the values at the

centre of the parameter space, and $\hat{\theta}_{min}$ and $\hat{\theta}_{max}$ are the lower and upper limits of the parameter space, respectively.

$$logLink_i = \sum_k -0.5 \cdot \frac{y_{sim,i,k} - y_{obs,k}}{\sigma_{obs,k}}^2 - 0.5 \cdot log(2\pi) - log(\sigma_{obs,k}) \tag{3}$$

The likelihood of the current link, $logLink_i$, is a measure of the model performance of a simulation using $\theta_{i,j}$. $logLink_i$ incorporates the difference between simulation results (y_{sim}) and observations (y_{obs}) for all variables k including also the
270 uncertainty of the observations (σ_{obs}). The overall likelihood ($logLI_i$) is compared to the highest likelihood that was achieved so far ($LogLI_{max}$) to decide about the acceptance of the current parameter set. If the difference between the current likelihood and the highest likelihood ($\Delta LogLI = LogLI_{max} - LogLI_i$) is positive, the parameter set is always accepted. For negative $\Delta LogLI$, it is only accepted if it exceeds the natural logarithm of a random number between 0 and 1. This mechanism prohibits that the algorithm is trapped in local optima. At the end of the chain, the algorithm returns a posterior parameter distributions
275 whose modes are the parameter values with the best model performance.

We used the same parameter space for all three new PFTs but to ensure that we select parameter values consistent with the traits associated with CSR theory, we prescribed a hierarchy (Tab. 2) for each parameter that defines whether a PFT has to obtain a higher or lower value compared to the other PFTs. For example S-PFTs must have a lower SLA than C- and R-PFTs because the S-strategy is associated with slower growth and longer living tissue than the C- and R-strategies.

280 Our approach included two steps represented by two subsequent chains. The first chain was short and used a large tuning parameter so that the sampling covered the entire parameter space and an area of good model performance could be identified. The second chain started in the area discovered by the first chain, was longer and used a smaller tuning parameter to find the optimal parameter values within the area.

We evaluated the new PFTs using the mean square error (MSE). For the evaluation, we either used a different data set or split
285 the data into different sets for parameterisation and evaluation if the number of replicates was at least eight for the majority of observations (Table 3). Observations with less than eight replicates were only used for the parameterisation. For the hot steppe we used the difference in SOC between the ungrazed and grazed scenario for the evaluation because the current representation of some processes within the model made it unable to simulate the overall SOC level adequately (Sect. 4.1.2). For the cold steppe, SOC data were only available for one year and the common management for the examined region (Wiesmeier et al.,
290 2011). While this is comparable to our extensive grazing intensity, for the intensive grazing intensity we assumed a 25% lower SOC level. We based this assumption on Kölbl et al. (2011) who reported around 25% lower SOC content of the topsoil under heavy grazing compared to areas without or with periods of moderate grazing.

2.5 Modelling protocol

Simulations with LPJmL are driven by data on climate variables and management. If available, we used climate data obtained
295 at the sites (see SI). For missing climate variables we supplemented data from the GSWP3-ERA5 data set for the temperate grassland and bias adjusted data from the MRI-ESM2-0 (Lange and Büchner, 2022) for the hot and cold steppe. To design the new PFTs and evaluate the model development, we reproduced the management under which the experiments were conducted (Sect. 2.2 and Table 1). For none of the sites, data on the land use history were available and we assumed livestock grazing

Table 4. Scenarios names and management (mowing/grazing intensity, irrigation, fertilisation) used for the simulations at the Lindhof (temperate grassland), Syferkuil (hot steppe) and Xilin (cold steppe) site.

Name	Mowing/grazing	Irrigation	Fertiliser application
Temperate grassland N0	Mowing (4 Cuts)	rainfed	unfertilised
Temperate grassland N1	Mowing (4 Cuts)	rainfed	fertilised $240\,\mathrm{kgN \cdot ha^{-1} \cdot year^{-1}}$
Hot steppe C0 R U	Grazing ($0.0\,\mathrm{Cows \cdot ha^{-1}}$)	rainfed	unfertilised
Hot steppe C1 R U	Grazing ($0.1\,\mathrm{Cows \cdot ha^{-1}}$)	rainfed	unfertilised
Hot steppe C0 I U	Grazing ($0.0\,\mathrm{Cows \cdot ha^{-1}}$)	irrigated	unfertilised
Hot steppe C1 I U	Grazing ($0.1\,\mathrm{Cows \cdot ha^{-1}}$)	irrigated	unfertilised
Cold steppe S1 R U	Grazing ($1.5\,\mathrm{Sheep \cdot ha^{-1}}$)	rainfed	unfertilised
Cold steppe S6 R U	Grazing ($9\,\mathrm{Sheep \cdot ha^{-1}}$)	rainfed	unfertilised
Cold steppe S1 I U	Grazing ($1.5\,\mathrm{Sheep \cdot ha^{-1}}$)	irrigated	unfertilised
Cold steppe S6 I U	Grazing ($9\,\mathrm{Sheep \cdot ha^{-1}}$)	irrigated	unfertilised
Cold steppe S1 I F	Grazing ($1.5\,\mathrm{Sheep \cdot ha^{-1}}$)	irrigated	fertilised
Cold steppe S6 I F	Grazing ($9\,\mathrm{Sheep \cdot ha^{-1}}$)	irrigated	fertilised

with a moderate density for 390 years to account for the transition from natural vegetation to managed land. A detailed list of all inputs and settings to reproduce the conditions of the sites and experiments is provided in the SI.

In addition to the simulations done for the parameterisation of the new PFTs, we simulated several scenarios to analyse forage supply and SOC for different water or nitrogen limitation levels. For each site, we simulated the two management schemes also used to derive the new PFTs. To evaluate the changes of forage supply, SOC and community composition in response to different resource limitations, we simulated our three sites additionally without the prevailing site-specific limitations. For this, we removed water limitation for the hot steppe and water and nitrogen limitation separately for the cold steppe (Table 4).

Pre- and postprocessing of the data and figure creation were conducted using R (R Core Team, 2019). A list of all R packages used is provided in the SI.

3 Results

We evaluated LPJmL-CSR for the selected variables (Sect. 3.1) - results of the parameterisation are shown in the SI. Afterwards, we assessed the effect of removing the resource limitations (Sect. 3.3), compared the traits and trade-offs within and across sites (Sect. 3.2) and analysed the community composition (Sect. 3.4).

3.1 Evaluation of new PFTs

For each site and management scenario, the new PFTs led to improved model results for forage supply and a reduced MSE compared to a simulation using LPJmL 5.3 (Fig. 1 a, d and g), which did not include the changes described in Sect. 2.3. A
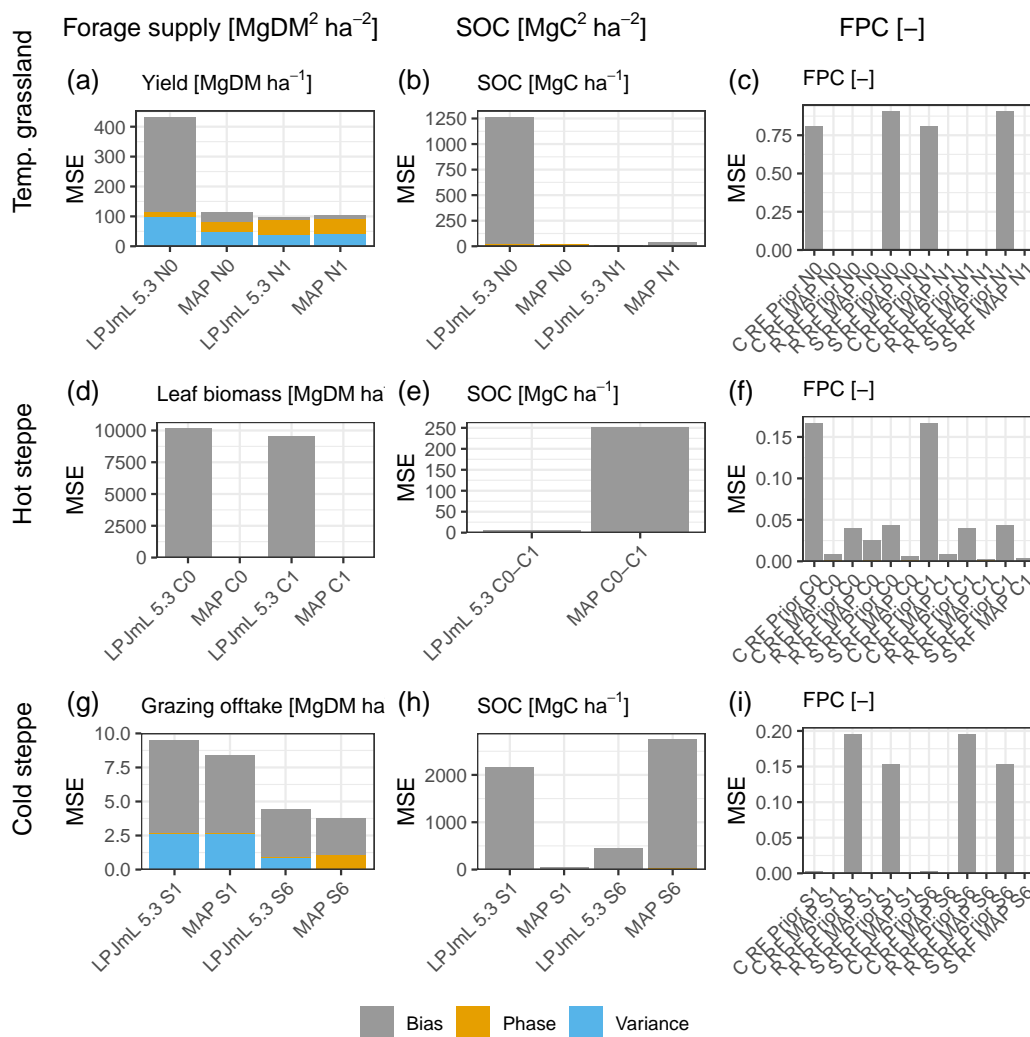
Figure 1. MSE for the different management scenarios (x-axis) for forage supply, SOC and FPC (columns, left to right) for the temperate grassland, hot and cold steppe (rows, top to bottom). For forage supply and SOC, MSEs for the old (LPJmL 5.3) and new (LPJmL-CSR) model version are shown. For FPC MSEs are shown for each PFT separately for LPJmL-CSR before and after the calibration. The colours separate the MSE into three components: the bias (grey) showing the systematic error for each variable, the phase (yellow) showing the temporal shift against observations and the variance (blue) which is the random error not attributable to bias and phase compared to observations.

315 major improvement was the capability of LPJmL-CSR to distinguish between CSR-strategies using different PFTs. For all sites and strategies we were able to find parameter sets for the new PFTs that enable LPJmL-CSR to represent the community well. Annual averages of the C-, S- and R-PFT cover simulated by LPJmL-CSR compared well to the expected cover which we used for the parameterisation. MSEs for the FPC were below 0.02 (Fig. 1 c, f and g) across sites and scenarios. Simulation results for forage supply improved at all sites (Fig. 1 a, d and g). For the temperate grassland and the extensive grazing scenario in the

320 cold steppe, the MSE of SOC was lower in LPJmL-CSR (Fig. 1 b and h) but similar for the hot steppe and moderately higher for the intensively grazed cold steppe (Fig. 1 e and h).

3.1.1 Temperate grassland

Forage supply of the temperate grassland for the unfertilised scenario was strongly underestimated by LPJmL 5.3 (Fig. SI 2 a) and the MSE improved from 431.7 to 112.2 $(\mathrm{MgDM \cdot ha^{-1}})^2$ in LPJmL-CSR (Fig. 1 a). For the fertilised scenario, LPJmL

325 5.3 underestimated forage supply less severely (Fig. SI 2 b) and the MSE was similar with 96.4 $(\mathrm{MgDM \cdot ha^{-1}})^2$ in LPJmL 5.3 to 105.3 $(\mathrm{MgDM \cdot ha^{-1}})^2$ in LPJmL-CSR. For the unfertilised scenario, the representation of SOC improved as well. For the unfertilised scenario LPJmL 5.3 strongly underestimated SOC stocks (Fig. SI 3 a) and the MSE was reduced from 1262 to 21.4 $(\mathrm{MgC \cdot ha^{-1}})^2$. While it remained similar with 11 and 37.9 $(\mathrm{MgC \cdot ha^{-1}})^2$ for the fertilised scenario (Fig. 1 b SI 3 b).

3.1.2 Hot steppe

330 Simulation results for the hot steppe presented a mixed picture showing lower MSEs for forage supply but higher MSEs for SOC in LPJmL-CSR compared to LPJmL 5.3. For the ungrazed (C0) scenario, the MSE of forage supply improved from 10154.1 to 1.9 $(\mathrm{MgDM \cdot ha^{-1}})^2$ (Fig. 1 d). Similarly, for the grazed (C1) scenario, the MSE of forage supply improved from 9522.5 to 40.1 $(\mathrm{MgDM \cdot ha^{-1}})^2$. The MSE for the difference in SOC between the ungrazed and grazed scenario was lower in LPJmL 5.3 and increased from 6.3 to 251.2 $(\mathrm{MgC \cdot ha^{-1}})^2$ (Fig. 1 e). LPJmL 5.3 already simulated the SOC difference

335 between the scenarios well impeding improvements through LPJmL-CSR. Furthermore, improvements in forage supply outweigh degradation in SOC stocks and LPJmL-CSR fits the observations better overall. However, compared to observations, LPJmL 5.3 severely overestimated forage supply and LPJmL-CSR underestimated forage supply (Fig. SI 4) and both model versions overestimated SOC in the ungrazed and grazed scenario (Fig. SI 5).

3.1.3 Cold steppe

340 For the cold steppe, animal feed demand was met in both model versions for the low grazing intensity (S1). Still, the MSE for forage supply improved from 9.5 to 8.4 $(\mathrm{MgDM \cdot ha^{-1}})^2$ (Fig. 1 g). For the high grazing intensity (S6), the feed demand was not always met in both models versions. Here, the MSE improved from 4.5 in LPJmL 5.3 to 3.8 $(\mathrm{MgDM \cdot ha^{-1}})^2$. Both LPJmL 5.3 and CSR underestimated observed forage supply for both grazing intensities but the dynamics at the high grazing intensity were captured better by LPJmL-CSR (Fig. SI 6). Unfortunately, only data on SOC for one year, which did not distinguish

345 between areas of different grazing intensity, were available. Since these data were already used for the parameterisation,
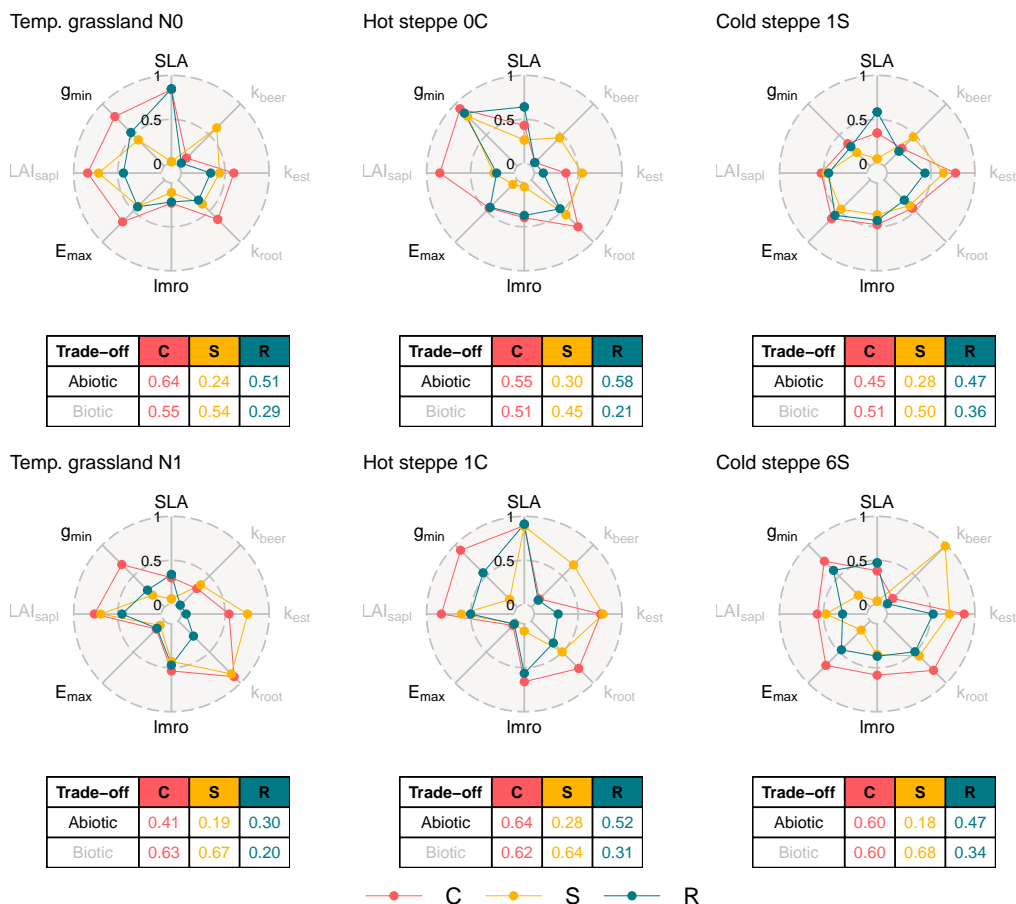
Figure 2. Spiderplots of normalised parameter values after calibration for each site (columns) and management scenario (rows). The centre and the edge represent the low and high end of the abiotic (black labels) and biotic (grey labels) trade-off dimension. The colours of the points distinguish the three PFTs. The tables show the mean of normalised parameter values for each PFT and the two trade-off dimensions.

we were not able to properly evaluate SOC. While LPJmL 5.3 strongly underestimated SOC for the low grazing intensity (S1), LPJmL-CSR captured the observations better but still underestimated observations (Fig. SI 7). Values were within the standard deviation of the observations for the low grazing intensity. For the high grazing intensity (S6), we assumed that 75% of the observed SOC to be an appropriate estimate for calibration (Sect. 2.4.2). However, both LPJmL 5.3 and LPJmL-CSR

350   overestimate this reduced calibration estimate (Sect. 4.1.3). The MSE was reduced from 2157.5 to 60.5 $(\mathrm{MgC} \cdot \mathrm{ha}^{-1})^2$ for the low and increased from 456.7 to 2741.5 $(\mathrm{MgC} \cdot \mathrm{ha}^{-1})^2$ for the high grazing intensity (Fig. 1 h).

3.2 Comparison of parameterisations between sites and different management intensities

The environmental conditions, the management and the communities at the examined sites were different, and each site and management could be placed at a different location within the CSR-triangle. Therefore, we expected different parameterisa-

355 tions across and within the sites for our new PFTs reflected through the PFTs' positions in the abiotic and biotic dimension. Throughout this study, we focus on the two dimensions and discuss parameters in the context of these dimensions.

3.2.1 Management intensities

At all sites, the different management scenarios resulted in different parameter values for the three PFTs. For the temperate grassland, the C and R-PFT shifted to a less resourCce exploitative strategy in the fertilised scenario indicated by the lower

360 value for the abiotic dimension which resulted from higher leaf longevity (lower SLA), while the S-PFT remained similar (Fig. 2). Additionally, all PFTs showed a lower maximum transpiration rate ($E_{,}max$) and higher investments into above-ground biomass (higher $lmro$). In the biotic dimension, the C- and S-PFT had a higher value in the fertilised scenario. For the C- and S-PFT this indicated a shift towards less offspring (lower k_{est}) and a more efficient root network (higher k_{root}). The R-PFT had a lower value caused by an increase in number of offspring (higher k_{est}).

365 For the hot steppe, the S- and R-PFT showed a lower value in the abiotic dimension for the grazed scenario (C1) and shifted to a more water saving (lower E_{max} and/or g_{min}) strategy. These differences were counteracted to some extent by an increased investment in above-ground biomass (higher $lmro$) and a more resource exploitative strategy (higher SLA). The C-PFT showed similar changes except for the reduction of the minimum canopy conductance (g_{min}). In the biotic dimension, all PFTs had higher values from different causes: The C-PFT shifted towards the establishment of less offspring (lower k_{est}).

370 The S-PFT increased its stature (higher k_{beer}), and sapling size (higher LAI_{sapl}). The R-PFT only increased its sapling size (higher LAI_{sapl}).

Consistent with the findings for the other sites, for the cold steppe all PFTs showed different strategies for the different management intensities. While the value for the abiotic dimension was the same for the R-PFT and only differed for the C- and S-PFT, all PFTs showed a shift within strategies. The C-PFT shifted to a less water saving strategy (higher E_{max} and g_{min}),

375 while the S-PFT shifted to a more water saving strategy (lower E_{max} and g_{min}). In the biotic dimension, the C- and S-PFT showed a higher value and the R-PFT a lower value in the intensively grazed scenario (S6). While for the C-PFT this was the result of an increase in the efficiency of its root network (higher k_{root}), for the S-PFT this was a result of an increase in stature (higher k_{beer} Sect. 2.4.1). In contrast, the R-PFT had a smaller stature (lower k_{beer}) and sapling size (lower LAI_{sapl}).

3.2.2 Site-specific conditions

380 Across sites, we found a large variation within both dimensions which ranged from 0.30 to 0.64 for the abiotic and from 0.18 to 0.68 for the biotic dimension (Fig. 2). As a consequence of our assumptions for the parameterisation, the sorting of the parameter values for the three PFTs had to match the hierarchy defined in Table 2 (Sect. 2.4.2) for each site. Between sites however, we did not make any assumptions that would predetermine an order, meaning that each site could occupy a different

area of the two dimensions. For example, an R-PFT had to have a higher value in the abiotic dimension compared to the S-PFT

385 for the same site, but could have a lower value compared to the S-PFT of another site, as is the case when comparing the temperate grassland to the hot steppe. For the biotic dimension, the same case can be made.

However, if averaged over all sites and management scenarios, the C-PFT still was the most resource exploitative with a value of 0.55 in the abiotic dimension, while the R- and S-PFT were more resource conservative with values of 0.48 and 0.25. Similarly, the R-PFT produced most offspring and had the smallest stature with a value of 0.29 compared to 0.57 and 0.58

390 for the C- and S-PFTs. While this general pattern emerged clearly for the two dimensions, there were substantial differences between the sites when comparing the contributing parameters. Most similar was the $lmro$ determining investments into above- versus below-ground biomass, which contributed to high values of the C- and R-PFT in the abiotic dimension for several scenarios. For the steppe sites, there was some alignment within the S-PFTs, which all had a larger stature (higher k_{beer}). The remaining parameters were not discernibly aligned across sites.

395 ### 3.3 Effects of resource limitation

To assess the effect of resource limitation, we compared different scenarios with LPJmL-CSR. In addition to the scenarios using the prevailing climatic conditions (resource limited), we simulated scenarios where we removed the limitation of water or nitrogen supply.

3.3.1 Temperate grassland

400 The temperate grassland already is a productive site where water and nitrogen are not limiting productivity and we did not simulate any additional scenarios but focused on comparing the two fertilisation levels (N0 and N1). For both scenarios total annual forage supply was similar and between 5.3 and 7.4 $\mathrm{MgDM\ ha^{-1}\ year^{-1}}$ for the unfertilised and 4.7 and 8.9 $\mathrm{MgDM\ ha^{-1}\ year^{-1}}$ for the fertilised scenario (Fig. 3 a). The first cut was the most productive, yielding between 1.8 and 2.8 $\mathrm{MgDM\ ha^{-1}\ year^{-1}}$ for the unfertilised and between 2.5 and 3.9 $\mathrm{MgDM\ ha^{-1}\ year^{-1}}$ for the fertilised scenario. The

405 subsequent cuts contributed substantially to the overall forage supply except in 2018, which was a drought year. Here, the forage supply from all cuts was reduced. In all cuts the dominant C-PFT contributed the majority of the forage supply. In both scenarios, the S- and R-PFT barely contributed - 2 and 8% share of forage supply on average - in all cuts (Fig. SI 8). Overall, DMY was more stable between years (except 2018) in the fertilised scenario because of higher yields during the regrowth stages (cuts 2 to 4). These compensated the slightly lower DMY of the first cut compared to the unfertilised scenario.

410 The SOC showed no significant trend for the unfertilised scenario, where the annual average decreased by 0.02 $\mathrm{MgC\ ha^{-1}\ year^{-1}}$ on average (τ=0.09, p-value 0.1). In contrast, SOC in the fertilised scenario increased by 0.96 $\mathrm{MgC\ ha^{-1}\ year^{-1}}$ on average (τ=0.56, p-value <0.001), respectively (Fig. 3 d). Intra-annual SOC dynamics, which are driven by the litter production and C input from manure, were stronger in the fertilised scenario.
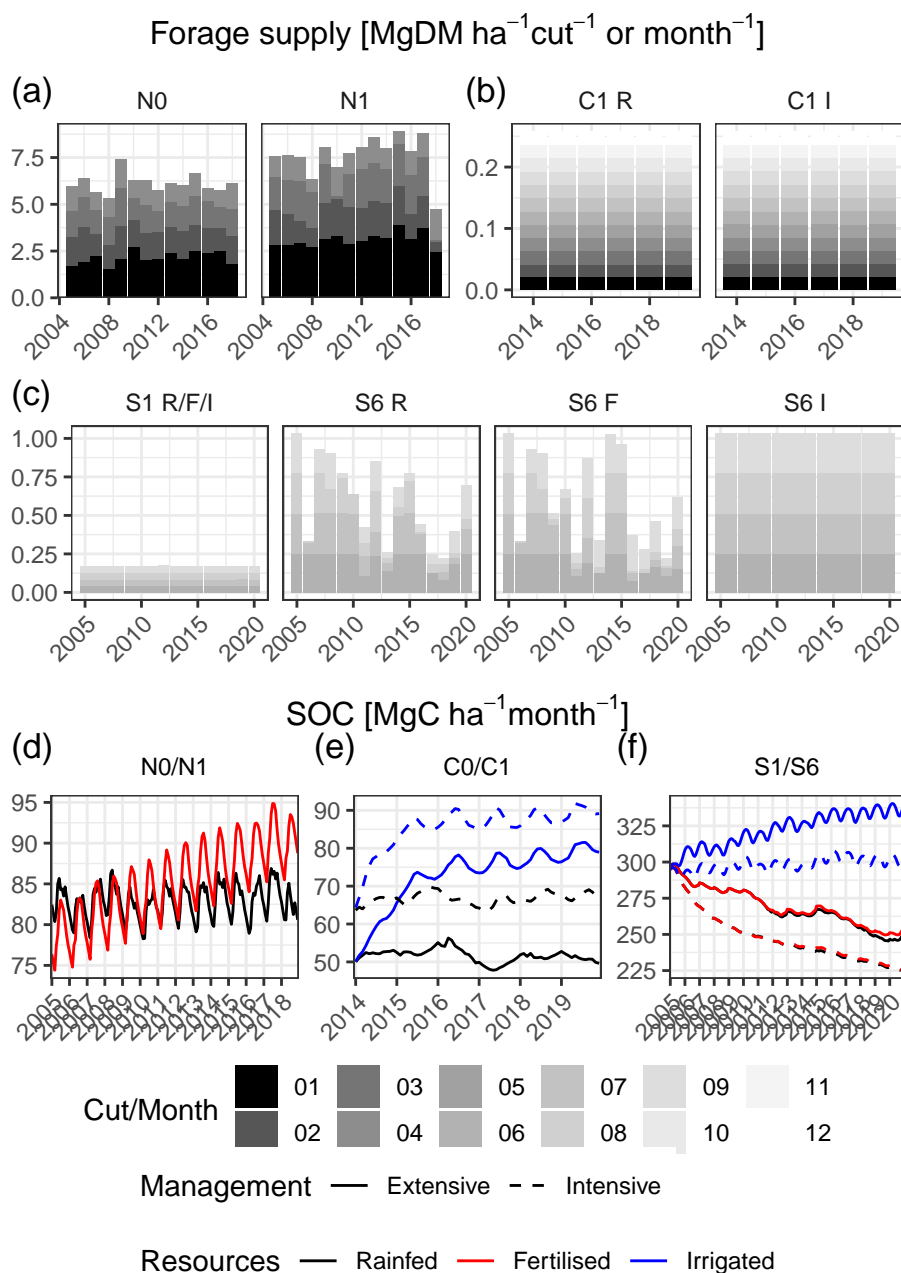
Figure 3. Simulated forage supply (a,b,c) and SOC (d,e,f) for all sites, management levels and resource limitation scenarios. Bars show the annual forage supply and coloured segments the forage supply for each cut/month. Line colours differ between rainfed (prevailing conditions, black), rainfed fertilised (red) and irrigated unfertilised (blue) while line-types show the grazing management intensity low (ungrazed/C0 or extensively grazed/S1, solid) and high (grazed/C1 or intensively grazed/S6, dashed).

3.3.2 Hot steppe

For the hot steppe, we simulated an irrigated (I) scenario in addition to the rainfed (R) scenario which was used for the calibration of our PFTs for the ungrazed (C0) and grazed (C1) management. Annual forage supply was 0.26 MgDM ha^{-1} year^{-1} in the rainfed scenario (Fig. 3 b) and animal feed demand was always met (Fig. SI 9 a). Similarly, the feed demand was always met in the irrigated scenario (Fig. SI 9 b). However, between the two scenarios the composition of forage supply strongly differed: In the rainfed scenario, the S-PFT contributed the majority in most years whereas in the irrigated scenario, the community composition changed and all PFTs contribute to forage supply similarly. A shift also occurred in the ungrazed scenario, which was still dominated by the S-PFT but showed a higher share of the C- and R-PFT after several years as well. This change is related to the changing community composition (Sect. 3.4.2) and increased leaf biomass in the irrigated scenario. In the ungrazed scenario, 55% of the leaf biomass increase from irrigation resulted from elevated growth of the S-PFT, 21% from the C- and 23% from the R-PFT. This was different in the grazed scenario, with -18% (S-PFT), 82% (C-PFT) and 37% (R-PFT) respectively.

The SOC of the rainfed scenarios did not show strong trends (Fig. 3 e). However, the negative trend in the ungrazed scenario (C0) was still significant (τ=-0.27, p-value <0.001). In the irrigated scenario, SOC increased strongly with little differences between the grazing scenarios - on average by 4.9 MgC ha^{-1} year^{-1} in the ungrazed and 4.2 MgC ha^{-1} year^{-1} in the grazed scenario. However, SOC did not increase linearly but showed a much stronger increase which was unrealistically high in the first one to two years after the start of irrigation (10.4 MgC ha^{-1} year^{-1} in the ungrazed and 10.6 MgC ha^{-1} year^{-1} in the grazed scenario) than in the remaining time-series (2.1 MgC ha^{-1} year^{-1} in the ungrazed and 1.0 MgC ha^{-1} year^{-1} in the grazed scenario).

3.3.3 Cold steppe

For the cold steppe, we simulated an irrigated (I) and a fertilised (F) scenario in addition to the rainfed (R) scenario used for the parameterisation for both the low (S1) and high (S6) grazing intensities. Total forage supply was 0.17 MgDM ha^{-1} year^{-1} for all scenarios with low grazing intensity because the feed demand of the animals was always met (Fig. 3 c). In all scenarios the forage supply was almost entirely attributed to the dominant S-PFT (Fig. SI 10 a-c). For the high grazing intensity, total forage supply was 1.03 MgDM ha^{-1} year^{-1} if the feed demand was met. This was always the case in the irrigated scenario but not in the rainfed and fertilised scenarios. In the latter two, the model simulated very similar forage supply, indicating that nitrogen addition was not sufficient to increase productivity because water was the main limiting factor. In all three scenarios, the S-PFT was dominant (Fig. SI 10 d-f). However, in the rainfed and fertilised scenarios the share of the S-PFT decreased in months where the feed demand could not be met and mainly the share of the C-PFT increased. In the irrigated scenario, only the S-PFT contributed to the forage supply.

SOC was similar for the rainfed and fertilised but differed for the irrigated low and high grazing intensity scenarios (Fig. 3 f). Both the rainfed and fertilised scenarios showed a significant negative trend for SOC, which was similar between the high grazing intensity where SOC decreased by roughly 4 MgC ha^{-1} year^{-1} on average (τ=-0.99, p-value <0.001) and the low

grazing intensity with SOC losses of 3 $MgC\ ha^{-1}\ year^{-1}$ on average (τ=-0.87, p-value <0.001). For the irrigated scenarios, SOC increased by 2.5 $MgC\ ha^{-1}\ year^{-1}$ on average (τ=0.79, p-value <0.001) for the low and 0.4 $MgC\ ha^{-1}\ year^{-1}$ on average (τ=0.47, p-value <0.001) for the high grazing intensity.

3.4 Community composition

450

We compared expected and realised shares of the C-, S- and R-PFT for the three sites using leaf biomass, explored seasonal and inter-annual dynamics and analysed shifts under different resource limitations.

As already evidenced by the low MSE values for the FPCs of all PFTs after calibration (Fig. 1), LPJmL-CSR captured our expert estimates on C-,S- and R-PFT cover, which defined the position of the ecosystem within the CSR triangle, well.

455 However, these were annual averages and did not prescribe any intra-annual variability. Since above-ground biomass and FPC are directly related and above-ground biomass is the less abstract variable to interpret, we present results on above-ground biomass from here on.

3.4.1 Intra-annual variability

Each site showed substantial intra-annual dynamics of total above-ground biomass (Fig. SI 11 a, c, 12 a, e, 13, a, g) and the

460 monthly average of the above-ground biomass share of the C- S- and R-PFTs (Fig. 4). However, the intra-annual dynamics were different between sites. In the temperate grassland, the C-PFT was dominant throughout the year, however, after the end of a growing season, the marginal PFTs had an increasing share until after the first cut (Fig. SI 11 b, d). While in the unfertilised (N0) scenario, the share of the S-PFT increased, the share of the S- and R-PFT increased in the fertilised (N1) scenario (Fig. 4 a).

465 In the hot steppe, the community was dominated by the S-PFT in both management scenarios (Fig. 4 b and d). In the ungrazed (C0) scenario, the C-PFT made up almost the entire remainder of the above-ground biomass (Fig. SI 12 a). However, the C-PFT was replaced by the R-PFT in the grazed (C1) scenario (Fig. SI 12 c).

For the cold steppe, PFT shares of above-ground biomass did not show strong intra-annual variation for the extensive (S1) grazing scenario (Fig. 4 c). However, for the intensive (S6) grazing scenario the C- and R-PFT strongly contributed to the

470 overall leaf biomass and the C-PFT was even dominant in during and after the grazing period (Fig SI 13 h).

3.4.2 Effects of irrigation and fertilisation

Removing resource limitations led to a shift of the community composition for the hot and cold steppe.

The hot steppe transitioned from an S dominated community to a community with more balanced CSR shares that was still dominated by the S-PFT (Fig. 4 b and d). This transition occurred within the first one to two years after the beginning of

475 irrigation for both scenarios (Fig. SI 12 c, d), which was reflected through the shift of the community average in Fig. 4 b and d.
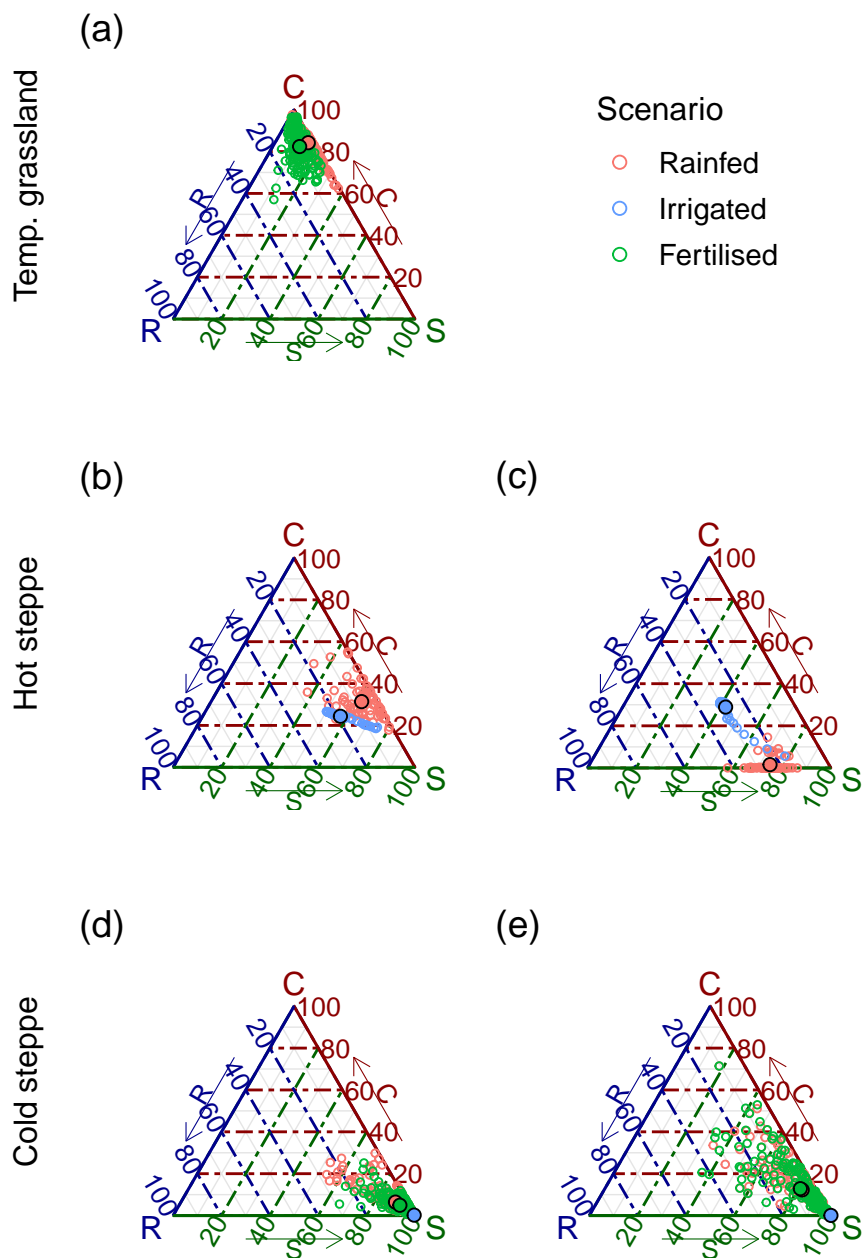
Figure 4. Ternary plots of the share of above-ground biomass of the C-, S- and R-PFT for the temperate grassland (a), the ungrazed (b) and grazed hot steppe (c), and the extensively (d) and intensively (e) grazed cold steppe. Colours differ between the rainfed (red), irrigated (blue) and fertilised (green) scenarios. Points with a black border show the mean composition of the time-series.

While removing the nitrogen limitation did not alter the community composition of the cold steppe under extensive (S1) and intensive (S6) grazing, irrigation had an effect (Fig. 4 c and e).The S-PFT out-competed the other PFTs entirely in the both grazing scenarios throughout the time-series (Fig. 4 c and e and SI 13 e, f, k, l).

4 Discussion

4.1 Forage supply, SOC and community composition under different management and resource limitations

At all sites, forage supply, SOC, and community composition differed between the different management intensity and resource limitation scenarios. The implemented model extension enabled the model to successfully simulate differences between C-, S- and R-strategists (Sect. 4.2). We were able to define new PFTs using a Bayesian calibration method that led to improved simulation of forage supply and/or SOC at three sites under different environmental conditions and management. Our implementation is a major advancement because:

1. it allows for explicit analyses of the adaptation of the vegetation to changing conditions compared to the model version in which only productivity changed.

2. changes in the productivity of the community caused by changing conditions are the result of a changing community composition and should therefore not only be quantitatively different to those in LPJmL 5.3 but also more reliable.

3. this allows to assess the adaptive capacity under different levels of functional diversity by adding or removing specific strategies.

4.1.1 Temperate grassland

While the fertilised scenario for the temperate grassland was already well simulated in LPJmL 5.3, the unfertilised scenario underestimated forage supply (Sect. 3.1.1). In LPJmL-CSR, growth of the vegetation was faster than in LPJmL 5.3 which led to higher yields for all cuts.

The temperate grassland is neither water nor nutrient limited and since we only assess scenarios with reduced resource limitations, we only compared the fertilised and unfertilised scenarios. Despite the additional nitrogen input in the fertilised scenario, the unfertilised scenario achieved a similar forage supply. Missing nutrients were acquired through biological nitrogen fixation, which was much higher in the unfertilised scenario which is in line with the higher share of legumes observed in the field experiments (Reinsch et al., 2020). Despite the higher share of legumes in the unfertilised experiments, the share of C-, S- and R-strategists was similar and both fertilisation levels were dominated by C-species, which was well-represented by the model.

The simulated SOC was strongly dependent on the land use history for which available data were limited. For simplicity we did not simulate crop rotations for the land use history but selected a livestock density of $1.0 \, \mathrm{cows \cdot ha^{-1}}$ (see SI) that led to an underestimation of observations in unfertilised scenario in LPJmL 5.3. This indicated, that carbon inputs into the soil were

too low in LPJmL 5.3. LPJmL-CSR showed smaller deviations from observations and an adequate representation of the trends (Sect. 3.1.1).

The community composition showed some intra-annual variability, and higher shares of the marginal PFTs at the end and the beginning of a growing season in the unfertilised and fertilised scenarios (Sect. 3.4.1). The S-PFT gained higher shares
510   in the unfertilised scenario showing an advantage of the S over the R-PFT despite the fact that strong nitrogen stress was avoided through biological nitrogen fixation. In contrast, if nitrogen stress was removed entirely, the S-PFT lost its advantage and the R-PFT could increase its share. After the first cut, these shares of the S- and R-PFT became smaller because a cut is a disturbance that directly removes part of the above-ground biomass. One strategy to cope with this is grazing (or in this case mowing) tolerance (Briske, 1986; Stuart-Hill and Mentis, 1982), which requires fast regrowth of the leaves to compensate for
515   the removed biomass as is typical for a C strategist (Grime, 1977), and Sect. 4.2.2).

4.1.2   Hot steppe

For the hot steppe, LPJmL 5.3 performed better for SOC while LPJmL-CSR performed better for forage supply. We identified several reasons for the inconsistent results: First, LPJmL does not distinguish between leaves of different age classes and therefore not between alive, senescent and moribund tissue (Schaphoff et al., 2018). All tissue is either alive and associated
520   with the plant or moribund and part of the litter layer. However, observed forage supply contained senescent biomass (Munjonji et al., 2020). This predisposed the model to underestimate forage supply when accounting for realistic turnover rates, which was observed in the low biomass values simulated in LPJmL-CSR. Second, litter decomposition is a function of soil moisture, temperature and litter composition (Schaphoff et al., 2018). However, the PFTs do not differ in their persistence of the litter which is the case for different plant species and across ecological strategies (Brovkin et al., 2012). Considering this may help
525   to improve the simulation of SOC dynamics in the future. Third, the vegetation was described as an open thornbush savanna (Acocks, 1994) which includes a woody component. However, in LPJmL managed grassland vegetation does not include bushes or trees and therefore only partially represents the observed community.

The S-PFT was dominant in the grazed and ungrazed scenario, while the remainder of the above-ground biomass was contributed by different PFTs depending on the scenario (Sect. 3.2.1). The R-PFT was better suited to withstand the grazing
530   and out-competed the C-PFT in the grazed scenario due to its higher ability to deal with disturbances.

Removing the water limitation led to an increase of forage supply and SOC which can be expected when removing the main resource limitation. However, the majority of the SOC increase occurred in the first two years after the start of irrigation which is not realistic. This can be explained with the missing representation of senescent tissue in combination with the adaptation of the community composition: Removing the water limitation led to a strong increase of leaf biomass, which was substantially
535   higher than the feed demand of the simulated grazing intensity and contributed to the litter layer. Furthermore, the share of the R- and C-PFT which have a lower leaf longevity than the S-PFT increased leading to faster inputs into the litter layer. After one to two years the community composition reached a new equilibrium and inputs into the litter layer decreased. Introducing senescent tissue would increase the competition for light and space and likely slow down this transition.

In addition irrigation led to a shift in the community composition (Sect. 3.4.2) and an increase in leaf biomass to which the
540 C- and R-PFT together contributed more than the S-PFT (Sect. 3.3.2). We cannot determine whether or not increases under irrigation would be lower for a S-PFT monoculture which does not contain other ecological strategies but strongly suspect so.

In both the ungrazed and the grazed scenario, the community transitioned from strongly S dominated to a community with higher shares of the C- and R-PFT that was still S dominated. According to the CSR theory, this type of community emerges in somewhat stressed and disturbed habitats (Grime, 1977). While this case can easily be made for the grazed scenario, where
545 the disturbance is caused by the animals, the ungrazed scenario does not include such a clear disturbance. However, periods of drought can induce an additional disturbance (Wang et al., 2019) creating a niche for R strategists (Norton et al., 2016; Kooyers, 2015).

4.1.3 Cold steppe

LPJmL 5.3 underestimated the observed forage supply of the cold steppe because the feed demand, which was originally
550 designed to represent large cattle (Rolinski et al., 2018), was scaled down linearly with animal body weight. This led to an unrealistically low feed demand because the feed demand body weight relationship is not linear but follows a power law (Cordova et al., 1978). Our new calculation of feed demand (appendix A5) led to a higher feed demand and forage supply simulations were improved for low and high grazing intensities.

Under observed conditions, the high grazing intensity severely reduced above-ground biomass and feed demand was not met
555 in all years except the year directly after the increase in stocking density. The reduced biomass availability was also observed by Schönbach et al. (2012) in their field experiment.

During and after the grazing period, the C- and R-PFT had a higher share of the community above-ground biomass. Both these PFTs can regrow faster and invest more into above-ground biomass which gave them an advantage over the S-PFT under grazing. In addition to the observed environmental conditions, we simulated two scenarios where we removed the water and
560 nitrogen limitations separately. Removing the nitrogen limitation barely affected biomass availability, and forage supply was similar compared to the rainfed scenario (Sect. 3.3.3). The additional soil nitrogen could not be utilised by the plants because water was the main limiting factor (Li et al., 2011; Bai et al., 2004). In contrast, removing the water limitation led to an increase in leaf biomass, and forage supply met the demand in all years even for the high grazing intensity. This is in line with irrigation and fertilisation experiments conducted in the cold steppe (Li et al., 2011) and other sites with similar conditions (e.g. Shi
565 et al., 2022). Contrary to the results of Li et al. (2011), who reported a lower share of annuals and bi-annuals – that are more likely C than S-strategists – in the rainfed treatments, the S-PFT was dominant in the irrigated scenarios. One reason for this could be that LPJmL does not simulate seedbanks, which play a major role for the establishment and success of the annuals and bi-annuals (Thompson, 1987; Brown and Venable, 1986). Instead, LPJmL simulates establishment of additional saplings dependent on available space assuming that resources for reproduction are available at any time and not dependent on past
570 investments into seed production.

Despite the fact that we did not have separate data on SOC under the two grazing intensities, our results showed a lower SOC storage for the high grazing intensity typical for overgrazed steppes (e.g. Wiesmeier et al., 2012) compared to the low

grazing intensity which constituted the typical livestock density for the region (Hoffmann et al., 2016). Wiesmeier et al. (2012) investigated the effect of high grazing intensities on the SOC, observing significant SOC losses within three years of increased

575 grazing, which is in line with our simulation results. Fertilisation had no effect on SOC while irrigation led to an increase of SOC, which was stronger for the low grazing intensity. This is likely because more biomass was produced and the surplus of the feed demand was not removed but contributes to the litter layer. However, these gains would not justify the effort that would be necessary to irrigate large areas.

Removing the water limitation led to a transition from S dominated to a S monoculture community under both grazing

580 intensities (Sect. 3.4.2).Since the site was still severely nutrient limited and exposed to low temperatures, it seems that an S strategy remained advantageous. Furthermore, the S-PFT showed trait values associated with large investments in roots and more persistent root tissue (Sect. 3.2.1) which provides a likely explanation for its increased dominance: It had an advantage in the competition for the additional water.

4.2 Abiotic and biotic trait dimension across sites and management

585 ### 4.2.1 Across sites

We used a Bayesian calibration method to find suitable parameter values of eight parameters assigned to two trade-off dimensions for the new PFTs. Due to lacking data on starting values and ranges for the three new PFTs, we used the same ranges and starting values for each PFT but prescribed an order of the parameters. Within a site and management scenario, the prescribed hierarchy for specific parameters also predefined the ranking of the PFTs within the abiotic and biotic dimension. Across sites

590 and management we did not constrain the PFTs to positions within the two dimensions. Theoretically, all PFTs of the temperate grassland could have been associated with a more conservative strategy in the abiotic dimension compared to the PFTs of the hot steppe. However, while there were some differences between the sites and management, on average the C- and R-PFTs occupied a more resource exploitative position in the abiotic dimension and the S-PFTs a more conservative one (Sect. 3.2.2). Similarly, in the biotic dimension the C- and S-PFTs occupied a position associated with less but larger offspring and a larger

595 stature compared to the R-PFT. It is an emergent property of the model, that not only the relative position of the PFTs of a site and management scenario determined community composition but also the overall positions within the abiotic and biotic dimension (which we derived from the *global spectrum of plant form and function* Díaz et al. 2016) were important. Our experiences from these three sites showed similar strategies independent of environmental conditions, indicating that LPJmL-CSR is capable of reproducing the empirically derived trade-offs associated with the *global spectrum of plant form and function*

600 (Díaz et al., 2016). However, LPJmL-CSR will benefit from additional testing on larger scales in the future.

4.2.2 Across management

While missing processes such as the representation of seedbanks as at the hot steppe (Sect. 4.1.2) and poor data as at the cold steppe (Sect. 4.1.3) may have determined model dynamics to some extent, we clearly demonstrated the importance of representing different ecological strategies.

605   We found shifts within the abiotic and biotic dimensions between the different management intensities (Sect. 3.2.1), which were related to changes in resource limitations or disturbance level: Generally, a change in resource availability does only change the conditions for the establishment of a community but does not directly affect the established vegetation (changes in environmental filters Bazzaz, 1991; Woodward and Diament, 1991). In temperate grasslands, manure application increases N supply and reduces the number of available niches that can be occupied by different ecological strategies. In the unfertilised

610   experiment, species could satisfy their N demand through two different strategies: Competition for the limited resource in the soil or biological N fixation (BNF). In the fertilised experiment, only the first strategy was advantageous as BNF creates additional costs. In the field experiment this was evidenced through the substantially different amount of legumes between the two experiments (Reinsch et al., 2020). In the model, N-fixing and non-N-fixing species are both collated within each PFT. Therefore, in the unfertilised scenario, a PFT had to apply a strategy combining N uptake and fixation, whereas it could

615   focus on N uptake in the fertilised scenario. Since we calibrated the unfertilised and fertilised scenarios separately using the same data for C-, S- and R-PFT cover, the difference in strategy between the two scenarios is expressed through the different position of the PFTs within the abiotic and biotic dimension: Higher investments into below-ground biomass ($lmro$) provide an advantage in the competition for plant available nitrogen (Johnson and Biondini, 2001). In the model this led to a reduced need of fixing additional nitrogen which constituted an additional investment (Sect. 3.2.1).

620   In contrast to resource availability, a disturbance directly affects the vegetation. In the case of grazing, it also influences resource availability indirectly through removal of nutrients from and spatial redistribution within the system (Liu et al., 2023; Chuan et al., 2018; Wan et al., 2015). In LPJmL, the grazing of the animals at the steppe sites constituted a direct reduction of leaf biomass proportional to the cover of each PFT (Rolinski et al., 2018). Under intensive grazing, strategies of grazing tolerance or avoidance are essential (Briske, 1986; Stuart-Hill and Mentis, 1982). While grazing tolerance is mainly associated

625   with fast regrowth (Briske 1986; Hyder 1972, abiotic dimension), grazing avoidance strategies can operate in time and space. Grazing avoidance in time is possible through the completion of the life cycle between grazing intervals (Noy-Meir 1990, biotic dimension). Grazing avoidance in space is contingent on reducing plant size (Rechenthin, 1956; Branson, 1953). However, since plant size is not explicitly represented in LPJmL, we do not discuss this strategy further (Sect. 2.3.3). In the hot steppe, we simulated a daily grazing system, which makes grazing avoidance through the life cycle impossible and the PFTs had to

630   follow a grazing-tolerance strategy. This was expressed through changes in the abiotic dimension: All PFTs increased their investment into above-ground biomass and faster tissue growth (Sect. 3.2.1).

     At the cold steppe site, grazing only happened during the growing season and both grazing tolerance and avoidance could be useful strategies. Indeed, between the extensive and intensive grazing scenario the differences between the PFTs in both dimensions do support different strategy adjustments (Sect. 3.2.1). The C-PFT increased its investment into above-ground

635   biomass to tolerate grazing, while the S- and R-PFT did not show any adjustment. However, since the high grazing pressure caused degradation of the above-ground biomass, differences between the two management scenarios do not only reflect different strategies to deal with the disturbance, but also for survival outside the grazed period. As such, all PFTs constructed long living tissue to survive unproductive conditions outside the growing season in the intensive grazing scenario. This was

not necessary in the extensive grazing scenario because the PFTs retained substantial above-ground biomass at the end of the
640   growing season and did not need to be as resource conservative.

4.3   Limitations and further need for research

The representation of different CSR-strategies is a new feature in LPJmL, a model, which is mainly used at large to global
spatial scales. Past explorations have pointed out the difficulties of adding new PFTs to DGVMs in general (Yang et al.,
2015) and also to LPJmL (Wirth et al., 2021). We therefore decided to only add a small number of PFTs which should
645   represent the three main CSR-strategies and no sub-strategies. We used expert estimates to determine the shares of the three
strategies. These three strategy shares sum up to 100% and also encompass species that would be added to a sub-strategy
in a less coarse approach. Consequently, our results show a very simplified representation of the different strategies within a
community and across sites, which might be better represented using a small scale model such as IBC GRASS (May et al.,
2009) or GRASSMIND (Taubert et al., 2020a, b, 2012). However, large-scale applications also benefit from the inclusion
650   of universally applicable trade-offs between different ecological strategies and the improved representation of productivity
changes.

Furthermore, we reduced the trade-offs between C-, S- and R-strategists to fit into two dimensions and used a limited
amount of parameters to express these. While this simplification was necessary, this also means that we do not represent all
effects, advantages and trade-offs of functional diversity. However, as LPJmL is a global model, our aim was not to optimise
655   performance for specific sites, but to evaluate and test an approach, which can easily be applied at the global scale without
the need of a global data set on community composition of grasslands. Keeping this in mind, and considering the difficulties
of adding PFTs to a DGVM as well as the global heritage of the model, we find that representing even only the three main
CSR-strategies constitutes a major improvement of LPJmL.

For this study, we only assessed three sites at which our approach worked well. We did not include a site dominated by
660   R-strategists since this is not common for managed grasslands, but also did not include CS and CSR habitats which are typical
for unfertilised and fertilised pastures, respectively (Grime, 1974). Additional research including these intermediate habitats
might provide more insight on the newly-implemented strategies and trade-offs.

In LPJmL, herbaceous plants are represented as a number of average individuals, without an explicit representation of
geometry. Therefore, we used the light extinction coefficient as a proxy for stature, assuming that small stature plants would
665   be less competitive for light. However, this is not sufficient to simulate grazing avoidance in space (Sect. 4.2.2) and an explicit
representation of plant height and area could further improve the representation of ecological strategies (Wirth et al., 2021).

Data coverage for the temperate grassland site was good and observations were available for multiple years and with suf-
ficient replicates. For the two steppe sites, data on SOC were scarce. Especially data on trends and equilibria under specific
management conditions might promote further improvement of the model and help with the parameterisation of new PFTs.
670   We based our parameterisation of the new PFTs on expert estimates for the C-, S-, and R-PFT cover. While we are confident
that these estimates were adequate, data on a small number of traits would be sufficient to calculate the shares for each PFT

Biogeosciences
Discussions

following Pierce et al. (2013), and we would like to encourage including such data as a standard in sampling procedures for future experiments.

The scenarios we examined here only involved the reduction of stress by removing either water or nitrogen limitations.
675 Additional insight might be gained from doing the opposite and imposing additional limitations or looking into gradual changes of environmental conditions.

5 Conclusions

We presented a new approach for large-scale models and DGVMs to simulate the three main CSR-strategies of managed grassland plant species. In addition to improving the simulation of forage supply and SOC at three different sites, the approach
680 successfully simulated the dynamic community composition at these sites and reproduced the existence of a *spectrum of plant form and function* (Díaz et al., 2016). This is a major improvement allowing to explicitly assess how the presence or absence of specific strategies affects ecosystem functions of managed grasslands. Using this new feature, scenarios for projections of forage supply and SOC under climate change can be complemented with different constraints on the adaptive capacity of the vegetation. Such projections can provide a range of future grassland productivity as decision-support for policy-makers. To
685 further improve these projections, extending the sites by including intermediate habitats as well as the scenarios by including additional resource limitations (e.g. droughts) or gradual changes of environmental conditions (e.g. temperature increase) could be useful to gain additional insights on the model and to study the complex interactions of climate change, management and functional diversity.

Code and data availability. The source code is publicly available under the GNU AGPL version 3 license. An exact version of the code
690 described here and the data used to create the figures is archived under https://zenodo.org/record/7727990 (Wirth et al., 2023).

Appendix A: Model description

We provided a qualitative description of new model development in the main text (Sect. 2.3), for which we supplement the underlying equations and additional minor developments here.

A1 Water uptake

695 To make resource uptake of different resources dependent on different plant traits, we adapted the water uptake routine of the LPJmL model. Available soil water is now distributed between PFTs dependent on their root carbon ($C_{root,PFT}$) and a PFT-specific parameter ($k_{root,PFT}$), which is used as a substitute for information on root functional traits (e.g. branching of the root network, amount of fine roots, number of root tips). These traits cannot directly be incorporated because either the

Biogeosciences

Discussions

EGU

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simplified representation of below-ground plant organs hinders their representation or data are not sufficiently available.

$$f_{root,PFT} = w_{PFT} \cdot (1 - exp(-k_{root,PFT} \cdot C_{root,PFT})) \tag{A1}$$

Eq. (A1-A3) describe an exponential function which follows the approach used for the calculation of the foliage projective cover (FPC, see (Schaphoff et al., 2018)), which was used to distribute water between PFTs in previous model versions.

$$w_{PFT} = (1 - exp(-k_{root,PFT} \cdot \sum_{i}^{Number\ of\ PFTs} C_{root,i})) \cdot f_{root,sum}^{-1} \tag{A2}$$

Each PFT's access to plant available soil water ($f_{root,PFT}$) is weighted using Eq. (A2). Here w_{PFT} is calculated as the fraction of the respective PFT's potential access to the plant available soil water if the entire community root carbon would belong to it and the sum of all PFTs' access to plant available soil water if now weighting would be applied (Eq. A3).

$$f_{root,sum} = \sum_{i}^{Number\ of\ PFTs} 1 - exp(-k_{root,i} \cdot C_{root,i}) \tag{A3}$$

A2 The leaf economic spectrum

To incorporate the trade-offs associated with the LES, we implemented a power law relationship between SLA and leaf longevity (LL) described by Eq. A4

$$LL = a \cdot SLA^b \cdot 12^{-1}, \tag{A4}$$

where $a = 36.3753$ and $b = -0.85384$. a and b were derived from a regression (A1) using trait data for SLA and LL retrieved from the TRY database (Boenisch and Kattge, 2018; Kattge et al., 2011). A detailed listing of the data sets used is provided in SI Tab. 1. The leaf turnover rate is calculated as the inverse of the leaf longevity ($\tau_{leaf} = 1/LL$) and is linearly related to root turnover ($\tau_{root} = k \cdot \tau_{leaf}$ with $k = 2$) assuming that the LES and the conservation gradient (Bergmann et al., 2020) of the root economic space are aligned (Weigelt et al., 2021). Plant biomass is transferred to the litter pools each day only if one of two conditions is met: Under grazing, we assume that depending on the stocking density, leaf tissue is grazed before it becomes senescent, and we define a threshold ($\xi_{leaf} = 5\mathrm{gCm}^{-2}$) for leaf biomass below which no senescent tissue for turnover is available; For mowing, we assume that senescent leaf biomass has to build up again after a mowing event, and we define a threshold for the leaf-to-root mass ratio ($\xi_{lmtorm} = 0.7 \cdot lmtorm_{opt}$) beyond which senescent tissue is built up again.

A3 Reproduction and mortality

To improve the representation of different reproduction strategies and lifecycles, we adapted the establishment and mortality routine of the model. Both establishment and mortality are executed daily. In the new establishment routine, the number of average individuals (n_{ind}) and the carbon ($C_{ind,pool}$) and nitrogen ($N_{ind,pool}$) pools of the leaves and roots for the average
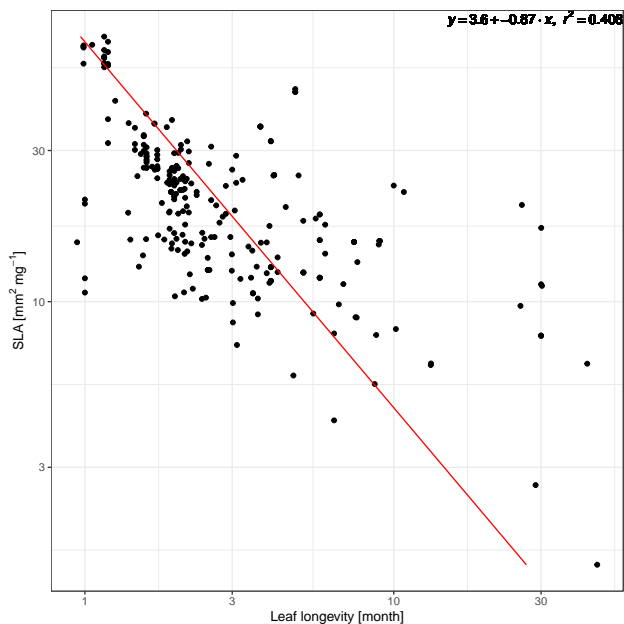
Figure A1. Linear regression of log SLA and log LL using trait data for herbaceous species from the TRY database.

individuals are increased following Eq. (A5-A7).

$$
\Delta n_{ind,PFT} = k_{est,PFT} \cdot 365^{-2} \cdot
$$
$$
(1 - exp(-5 \cdot (1 - FPC_{sum}))) \cdot
$$
$$
(1 - FPC_{sum}) \cdot (k_{est,PFT} \cdot
$$
$$
\left(\sum_{i}^{Number\ of\ PFTs} k_{est,i} \right)^{-1}) \tag{A5}
$$

$$
\Delta C_{ind,PFT} =
$$
$$
(C_{sapling,leaf,PFT} + C_{sapling,root,PFT}) \cdot \Delta n_{ind,PFT} \tag{A6}
$$

$$
\Delta N_{ind,PFT} = \Delta C_{ind,PFT} \cdot NC_{ratio,leaf,PFT} \tag{A7}
$$

Here, k_{est} is the PFT-specific establishment rate, $FPC_{sum} = \sum_{i}^{Number\ of\ PFTs} FPC_i$ is the sum of the FPC of all PFTs, $C_{sapling,pool}$ is the PFT-specific leaf and root pool size of a sapling and $NC_{ratio,leaf,PFT}$ is the PFT-specific nitrogen to

carbon ratio. The new individual properties are calculated following Eq. (A8) and (A9)

$$C_{ind,pool,PFT} = (C_{ind,pool,PFT} \cdot n_{ind,PFT} +$$
$$C_{sapling,pool,PFT} \cdot \Delta n_{ind,PFT}) \cdot$$
$$(n_{ind,PFT} \cdot \Delta n_{ind,PFT})^{-1} \tag{A8}$$

$$N_{ind,pool,PFT} = (N_{ind,pool,PFT} \cdot n_{ind,PFT} +$$
$$C_{sapling,pool,PFT} \cdot NC_{ratio,pool,PFT} \cdot \Delta n_{ind,PFT}) \cdot$$
$$(n_{ind,PFT} \cdot \Delta n_{ind,PFT})^{-1} \tag{A9}$$

Mortality was implemented as an age mortality using the concept of growth efficiencies (Waring and Schlesinger, 1985; Waring, 1983) using Eq. (A10)

$$mort_{PFT} = mort_{max,PFT} \cdot 365^{-1} \cdot$$
$$\Delta bm \cdot (C_{ind,leaf,PFT} \cdot SLA_{PFT})^{-1}, \tag{A10}$$

with

$$\Delta bm = C_{inc,PFT} \cdot n_{ind,PFT}^{-1} - C_{turn,PFT} \tag{A11}$$

where $C_{turn,PFT}$ is the amount of carbon that was transferred to the litter pool that day. The number of average individuals is decreased following Eq. (A12).

$$n_{ind,PFT} = n_{ind,PFT} \cdot (1 - mort_{PFT}) \tag{A12}$$

A4 Biological nitrogen fixation

Symbiotic biological nitrogen fixation (BNF) is an important source, especially in unfertilised grassland systems. We implemented an approach adapted from published models of grain legumes (e.g. LPJ-GUESS, CROPGRO, EPIC, APSIM see Ma et al., 2022; Liu et al., 2011), which considers the potential N fixation rate, the soil temperature and the soil water status. The consideration of the growth stage had to be omitted because LPJmL represents herbaceous vegetation using only leaves and roots, not allowing for a determination of growth stages. The nitrogen fixation rate N_{fix} is calculated using Eq. (A13)

$$N_{fix} = N_{fix,pot} \cdot f_T \cdot f_W \tag{A13}$$

with $N_{fix,pot} = 0.1 \, \mathrm{gNm^{-2}day^{-1}}$ (Yu and Zhuang, 2020). The soil temperature limitation is modeled linearly outside the optimal temperature range (Eq. A14):

$$f_T = \begin{cases} 0, & \text{if } T_{soil} < T_{min} \text{ or } T_{soil} > T_{max} \\ \frac{T_{soil}-T_{min}}{T_{opt,low}-T_{min}}, & \text{if } T_{min} \leq T_{soil} < T_{opt,low} \\ 1, & \text{if } T_{opt,low} \leq T_{soil} \leq T_{opt,high} \\ \frac{T_{max}-T_{soil}}{T_{max}-T_{opt,high}}, & \text{if } T_{opt,high} < T_{soil} \leq T_{max} \end{cases} \tag{A14}$$

Biogeosciences
Discussions

EGU
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with $T_{min} = 0.5$, $T_{op,low} = 18.0$, $T_{opt,high} = 35.0$ and $T_{max} = 45.0$ (Yu and Zhuang, 2020). The soil water limitation is lin-
765 early dependent on the relative soil water content SWC (Eq. A15):

$$
f_W = \begin{cases} 0, & \text{if } SWC \leq SWC_{min} \\ \varphi_1 + SWC \cdot \varphi_2, & \text{if } SWC_{low} < SWC < SWC_{high} \\ 1, & \text{if } SWC \geq SWC_{high} \end{cases} \tag{A15}
$$

with $SWC_{low} = 0$, $SWC_{high} = 0.5$, $\varphi_1 = 0$ and $\varphi_2 = 2.0$ (Yu and Zhuang, 2020).

BNF only happens if the nitrogen uptake from other sources is insufficient and the net primary productivity (NPP) is larger than
zero. The costs of BNF are set at a moderate constant value of 6 gCgN^{-1} (Boote et al., 2009; Kaschuk et al., 2009; Patterson
770 and Larue, 1983; Ryle et al., 1979). If the costs exceed the maximum costs which are set at 50% of the NPP (Kull, 2002) the
nitrogen fixation is reduced to the amount achievable with the maximum costs. A full description of the original approach is
provided in Ma et al. (2022).

A5 Feed demand

We implemented a relationship between metabolic body weight (MBW) and feed demand following (Cordova et al., 1978).
775 This is the same relationship used to calculate the feed demand in LPJmL 5.3, but we replaced the constant $650 \, \text{kg} \cdot \text{animal}^{-1}$
with a parameter BW (Eq.A16) while preserving $intake_{MBW} = 31.07$ (Rolinski et al., 2018).

$$
feeddemand = BW^{0.75} \cdot intake_{MBW} \tag{A16}
$$

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