Review

Effects of grazing management on spatio-temporal heterogeneity of soil carbon and greenhouse gas emissions of grasslands and rangelands: monitoring, modelling and upscaling

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Simple Summary: Grazing lands provide many goods and ecosystem services, such as forage, livestock, soil carbon (C) storage, biodiversity, and recreational opportunities. Ensuring the long-term sustainability of grazing lands requires management to be adjusted to simultaneously balance livestock productivity for sustaining human food and nutritional demands while reducing environmental impacts, such as greenhouse gases (GHG) emissions and soil degradation. In this paper, we review the measurement, monitoring, modeling and upscaling of soil C and GHG flux data, with a special emphasis on grazing lands used by livestock producers. We identify methodological opportunities and challenges including uncertainty in the quantification of soil C stocks and GHG emissions from grazing systems and propose a future framework for managing GHG emissions from lands subjected to grazing.

Abstract: The sustainability of grazing lands lies in the nexus of human consumption behavior, livestock productivity, and environmental footprint. Due to fast growing global food demands, many grazing lands have suffered from overgrazing, leading to soil degradation, air and water pollution, and biodiversity losses. Multidisciplinary efforts are required to understand how these lands can be better monitored, assessed and managed to attain predictable outcomes of optimal benefit to society. This paper synthesizes our understanding based on previous work done on the impacts of grazing on soil carbon (C) and the environment, identifies current knowledge gaps, and research priorities. We review the impacts of multi-paddock rotational grazing on soil carbon, nutrient cycling and GHG emissions. We then examine challenges of incorporating spatial heterogeneity and temporal variability into monitoring and modelling C and nutrient cycling in grazing lands. We revisit two widely-used process-based models: DeNitrification DeComposition (DNDC) and DayCent, and two watershed models: The Soil & Water Assessment Tool (SWAT) and

Variable Infiltration Capacity Model (VIC) to simulate C, nutrient and water cycles and thereby identify main issues in monitoring and modeling relative to grazing lands. Finally, we point out research trend for improving the knowledge base, which is essential to conserve grazing lands and maintain their ecosystem capital and services.

Keywords: grazed grassland; rangelands; grazing management; soil carbon; nutrient cycles

1. Introduction

Grazing lands include grasslands, rangelands, and pasture lands (hereafter called 'grazing lands') that provide many goods and ecosystem services, including forage, livestock, soil carbon storage, biodiversity, and recreational opportunities, among others. Grazing lands cover about 30% of the total global land use area and comprise 70% of the total land used for agriculture [1-2]. The livestock sector is responsible for 14.5% of global all anthropogenic greenhouse gas (GHG) emissions (CO_2 equivalent) [3]. On the other hand, soil carbon stocks in grazing lands contain about 10% of all terrestrial biomass and hold 20–30% of the global pool of soil organic carbon (SOC) [4,5]. Due to rapidly increasing global food demands, particularly for red meats, many grazing lands have suffered from overgrazing, leading to soil degradation, air and water pollution, and biodiversity losses. Ensuring the long-term sustainability of grazing lands requires management be adjusted to simultaneously balance livestock productivity for sustaining human food and nutritional demands with reducing environmental impact. Therefore, it is necessary to reduce soil degradation (i.e., SOC losses, erosion, and pollutants) and environmental impacts (i.e., GHG emissions and water pollution).

Grazing management includes the variety of decision-making tools for optimizing livestock production and environmental sustainability [6]. Grazing systems can be broadly categorized into two types: continuous grazing and multi-paddock rotational grazing, depending on the length of time that animals remain on a given land area, known as the grazing period. A multi-paddock grazing system facilitates animal movement among paddocks (i.e., rotation) to allow for more uniform seasonal forage productivity across the landscape, while balancing animal consumption and plant biomass utilization rates with the ecological requirements of both plants and soils under a given stocking rate (Fig. 1). In contrast, a continuous grazing system allows livestock to graze on a single pasture for an extended period, typically the entire grazing season. Multi-paddock grazing systems involve careful alteration of the timing, frequency, and intensity of grazing events, by altering the number of paddocks, animal density, and severity of defoliation. The length of subsequent rest (i.e., recovery) periods determines whether vegetation is able to recover following defoliation [7]. The ideal layout of pastures to facilitate rotational grazing depends on the local landscape, water provision, vegetation, and soil conditions.

The benefits of multi-paddock rotational grazing on livestock production have been realized [8, 9], but remain contentious in terms of the benefits to improving both, plant community productivity and condition, in comparison to continuous grazing [10, 11]. While grazing at high stocking rates can increase livestock productivity in the short-term, corresponding intensive livestock production practices with inadequate rest periods may increase GHG emissions, nutrient losses and soil degradation. By using high animal densities in small areas, multi-paddock grazing allows for more uniform forage offtake and subsequent manure/urine distribution, thereby cycling nutrients to the soil more effectively and preventing the redistribution of nutrients from grazed areas to loafing/bedding areas. When utilized properly, multi-paddock grazing may maintain plant vigor at higher levels

throughout the landscape, thereby maintaining or increasing livestock productivity per hectare [8].

Grazing system sustainability strives to maintain and increase both livestock and forage productivity for optimal profit. This depends on soil health and SOC stocks in grasslands. However, the plant recovery time under rotational grazing depends on environmental conditions, such as the season of year because grazing lands need an adequate recovery interval prior to re-grazing besides the length of the grazing period and intensity [12]. While the use of high stocking rates can increase livestock productivity, it can also initiate degradation of the plant community, leading to soil erosion and associated water pollution. Therefore, the frequency and timing of livestock movement is very important and demands constant monitoring and adaptive management [8]. Muller et al. [13] showed the importance of an adaptive framework to deal with complex rest period environments, such as unpredictable and stochastic rainfall in (semi-) arid regions. Therefore, adaptive grazing systems can make a farm more productive and profitable by balancing animal growth/production through forage removal, with the inherent defoliation tolerance of the plant community and resilient plant species [14]. Consequently, livestock management and animal husbandry under multi-paddock grazing includes complex processes at the forage-animal interface under a given set of environmental conditions with stocking rate considerations affects pasture productivity and sustainability. Multi-paddock grazing systems essentially control/adapt livestock distribution in time and space to promote adequate pasture recovery. However, this process is complicated by weather and soil conditions and animal movement. Grazing animals do not move and feed at random, but instead favor selected areas, including those near water, available salt and minerals, together with easily accessible areas. Given this, grazing lands are never grazed uniformly within a designated time frame, and the impact of the spatial distribution of animals is rarely uniform.
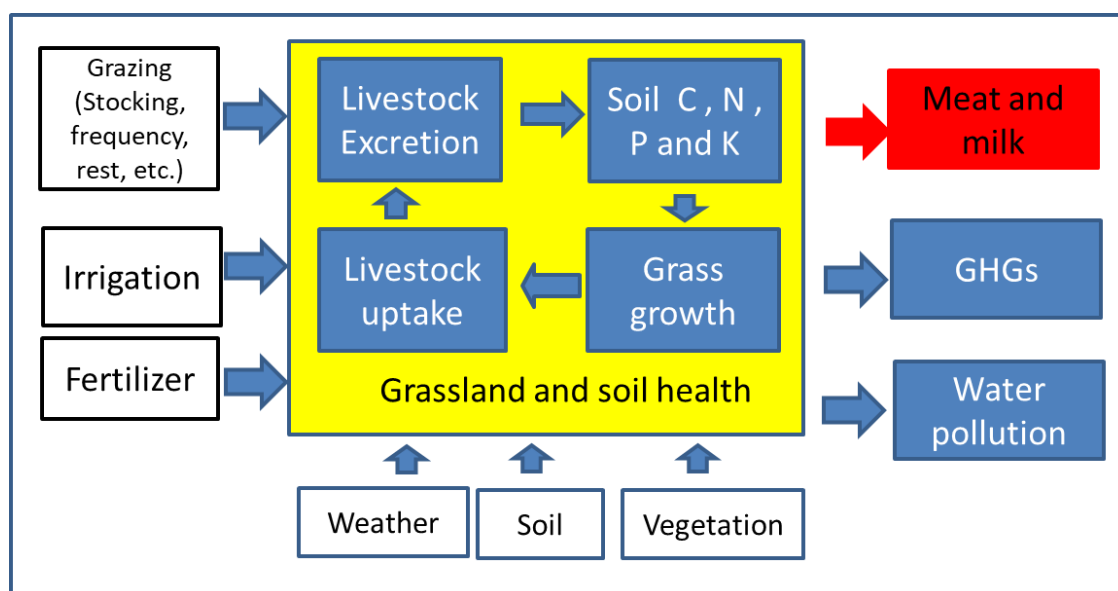

Figure 1. A schematic diagram of the key ecosystem processes within a typical grassland system, including management factors influencing the system (left side), intrinsic conditions (bottom), and key outcomes (right side).

Much effort has been made to control grazing intensity (i.e., stocking rate, frequency and duration of defoliation), the season, distribution of grazing and subsequent resting periods to optimize plant and livestock performance [15]. Although many comparisons of different

grazing systems have been conducted, including multi-paddock systems, these comparisons are complicated due to many uncertainties within the data from different disciplines and across different scales, and a lack of fundamental mechanistic studies identifying the causes of degradation (and comparative improvements) within grazing lands. However, adjustments to grazing are often limited because of complex environments, such as soil and water resources, nutrient and weather conditions [16, 17]. The relevance of broad-spectrum studies conducted by researchers do not reflect a broad understanding of the ranchers' perspectives regarding the efficacy of alternative grazing systems [18] and their rigid treatment structure could omit the adaptive nature of management commonly found on grazing lands at which livestock production occurs [14l, leading to inconsistencies in the outcome of these management investigations. Moreover, the vast majority of research experiments are performed separately in isolation, and thereby lack the systematic connection necessary to reach a consensus of scaling-up strategies due to the inconsistence and very limited context data. This is in part due to limitations of narrowly imposed experimental grazing research due to different soil, landscape and climate conditions and in part resultant knowledge transfer between researchers and ranchers [14, 18,19]. Some studies found that even at low stocking rates, animal preferences lead to patch-selected overgrazing due to inadequate recovery of the palatable species [20, 21]. Petrosillo et al. [22] pointed out that environmental vulnerability is multi-layered, multi-scale and complex, existing in both the objective physical, biological, and social realm, as well as the subjective realm of individual human perception.

Past studies do not effectively scale up to the level at which most grazing lands are utilized and lack the flexibility (i.e. adaptability) of grazing management, thereby leading to a deficiency in the tools/methods available for the assessment of grazing at different scales and locations. As a result, productivity of multi-paddock grazing may not necessarily be superior to that of continuous grazing due to environmental change [19, 23]. A survey of 765 ranchers in California and Wyoming showed that two-thirds of respondents used on-ranch multiple paddock grazing with a range of stocking densities (2.4–8 hectares. per animal unit), implying the benefits of multiple paddock grazing to ranchers [24]. However, only 5% of the respondents used multiple paddock grazing on rangelands for California. They found that most of grazing research are orders of magnitude finer than conditions under which on-ranch adaptive grazing management strategies have been developed. The discrepancy between research and practice was attributed to insufficient communication and coordination because the livestock production may not be the only driver examined in grazing systems research. In Ireland, adoption rates of multi-paddock rotational grazing among ranchers have been low despite extensive promotion of its advantages [25]. This pattern is attributed to various inconsistences between field trials and on-farm/rangeland practices or the inability to use research results effectively, thereby guiding grazing management in the desired direction [8-10]. To address this debate about the benefits of multi-paddock rotational grazing on livestock production we must critically examine the data and methods of grazing management that make a clear distinction between the relations of essence and appearance.

The primary goal of this paper is to synthesizes our understanding based on previous work done on the impacts of grazing on SOC, nutrient cycle and environment, identifies current knowledge gaps, and research priorities. We review the impacts of multi-paddock rotational grazing on soil carbon, nutrient cycling, and greenhouse gas emissions. Our intent is not to conduct an exhaustive review of all past literature on the topic but to examine data and methods of grazing experiments and modelling to explore potential ways to identify the knowledge gaps, and foster a path forward using modeling to facilitate data upscaling. This paper is organized as follows. First, we review experiments and monitoring in grazing lands exposed to different grazing systems considering economic and environmental aspects: productivity, soil carbon, nutrient runoff and soil erosion, as well as GHGs. We analyze the

separate experimental data to examine their disconnection points at different scales and environmental conditions in section 2. In section 3, we revisit process-based models and focus on two flagship models of agroecosystems: Denitrification and Decomposition (DNDC) [26] and DayCent [27,28] since they represent state of the art biogeochemical and agroecosystem models. We analyze their structures with respect to their representation of key grazing processes and SOM decompostions and discuss their weakness in hydrological representation. In Section 4, we review two widely used watershed models: the Soil & Water Assessment Tool (SWAT) [29, 30] and the Variable Infiltration Capacity (VIC) models [31]. We analyze their structures and inability to account for key biogeochemical processes, such as grazing management and nitrification and denitrification related to grazing lands. Finally, in Section 5, we present the recent progresses on modelling of SOM decomposition and coupling hydrological and biogeochemical models and point out the current and future needs of grazing system research and management with an emphasis on process-based modelling.

2. Effects of grazing intensity on C, N, SOC, GHG emissions and other indicators

Many field experiments have been performed to compare the effects of contrasting grazing practices on soil health, water infiltration, soil carbon stock, greenhouse gas emissions, nutrient runoff and soil erosion and grass cover [32,33]. Most are controlled experiments emulating multiple paddock grazing. A gradient of grazing intensities was designed with varying from non-grazed to very heavy grazing [34,35]. Wang and Wesche [36] compared grazing intensity on two groups of grassland indicators using a gradient of grazing intensity in northern China and the Qinghai-Tibetan plateau: (1) vegetation (i.e., plant species richness, vegetation cover, aboveground biomass, belowground biomass and root/shoot ratio) and (2) soil (pH, bulk density, SOC, total N, total P and available P). They found that belowground carbon stocks were higher than aboveground carbon in all management regimes. Most indicators deteriorated as stocking rates increased, while soil pH, bulk density and belowground biomass increased linearly with increasing grazing intensity. Elevation and weather conditions had an impact on aboveground biomass and SOC, while grazing effects on belowground biomass were affected by temperature, precipitation, and radiation. Badgery et al. [37] performed a grazing management experiment to determine effects of grazing intensities on the profitability and sustainability of a sheep production system. They compared continuous grazing with 4- and 20-paddock rotational systems. Their results showed large variations in feed availability and quality over the summer in between years. Flexible management was therefore suggested to utilize the greater feed supply in better seasons.

2.1. SOC, nutrient cycle and greenhouse has emissions as a function of grazing Intensity

Savian et al. [38] investigated multi-paddock rotational stocking targeting pre- and post-grazing sward heights of 25 and 5 cm, and 18 and 11 cm, respectively. Their goal was to identify the ideal sward heights for different grass species. They found the target height of 18 and 11 cm had a high potential to mitigate methane emissions by sheep. Jones et al. [39] found that soil respiration from plots receiving manure was up to 1.6 times larger than CO_2 release from control plots, and up to 1.7 times larger compared to inorganic treatments (p<0.05). Hewins et al. [40] found that while climate had the largest impact on SOC concentrations, grazing increased C concentration more in mesic grazing lands of Alberta, Canada. Teague et al. [41] indicated that grass cover under proper management is highly effective in reducing soil erosion and in increasing SOC stocks. A meta-analysis of 115

published studies [42] showed that livestock grazing intensity can alter ecosystem C and nitrogen (N) cycles in grassland ecosystems. It was found that light grazing increased soil C and N stocks whilst moderate and heavy grazing significantly increased C and N losses. The largest decreases in microbial biomass C and N were at 21.62% and 24.40%, respectively. Badiou et al. [43] studied GHGs and soil carbon sequestration potential in restored wetlands of the Canadian prairie pothole region. They found that methane emissions from seasonal, semi-permanent, and permanent prairie wetlands are very high while N_2O emissions from these sites are low. Gourlez de la Motte et al. [44] monitored the net ecosystem exchange (NEE) and CO_2 fluxes using eddy covariance in two adjacent pastures located in southern Belgium during a complete grazing season. Li et al. [45] studied changes of soil C, N, and P in a long-term grazing land, at Stavely, Alberta, Canada. They found that soil NO_3^- and NH_4^+ concentrations were not affected by grazing intensity. SOC and N stocks on rough fescue grassland were not affected by grazing while soil bulk densities (0–30cm) were higher. Phosphorus (P) stocks (15–30 cm) were lower under grazed than non-grazed sites. Labile SOM responds to grazing in dependence of slope positions [46]: at the bottom, SOC, total N, active C, NH_4^+-N, soil protein and water extractable N contents were higher under light grazing than under heavy or very heavy grazing. Counter-intuitive, no significant differences were found among the three grazing treatments in the top of the slope. Gao et al. [47] investigated soil trace gas fluxes from grasslands in the foothills of the Rocky Mountains, Alberta. Grazed grassland soils had 37 to 51% greater CO_2 emissions? compared to non-grazed soils. The N_2O emissions under heavy and very heavy grazing were 122 to 179% greater than non-grazed in the wet season. There were no effects of grazing on N_2O emissions in the normal precipitation season. Bikila et al. [48] compared grazing during the dry season with grazing enclosed for 20 years and burned grazing lands after fired for more than five years in Borana rangelands, Ethiopia. Their results showed that belowground carbon stocks were higher than the aboveground carbon stocks in all management systems. Tree and shrub carbon and soil organic carbon stocks were higher (P < 0.01) in rangelands enclosed for 20 years than other rangeland management systems, whereas grass carbon stocks was higher (P < 0.05) in fired rangelands. Whitehead et al. [49] studied the effect of grazing management on soil carbon stocks in grazing grasslands in New Zealand. They showed grazing practices can change soil carbon stocks through changes in NEE, the biomass removed and decomposition rate. Garnier et al. [50] explored the possibility of a reconnection of livestock and crop farming systems in the Orgeval watershed to reduce nitrate runoff to restore agricultural production and water quality in Seine basin, France. we They found a 50% reduction of surface water nitrate concentrations using organic cropping and organic reconnected livestock cropping systems. McCarthy et al. [51] assessed the impact of stocking rate on soil solution nitrogen components beneath a free-draining dairy production system using three stocking rates from light (2.51 cows/ha), medium (2.92 cows/ha), and intensive (3.28 cows/ha) in Teagasc, Ireland. They found that increasing stocking rate led to increased grazing efficiency and milk production per hectare. Stocking rate had not shown substantial influence on soil solution concentrations of nitrate, nitrite, ammonia, or total N. Coffey et al. [52] studied effect of stocking rate and animal genotype on milk production per hectare within intensive pasture-based production systems in Teagasc Moorepark, Ireland. They found that total pasture utilization per hectare consumed in the form of grazed pasture increased linearly as stocking rate increased.

    McSherry and Ritchie [53] conducted a multifactorial meta-analysis of grazing influences on SOC stocks on 47 independent experimental contrasts from 17 studies. They found that increasing grazing intensity increased SOC by 6-7% on C_4-dominated and C_4-C_3 mixed grasslands, but decreased SOC by an average 18% in C_3-dominated grasslands. Zhou et al. [42] performed a comprehensive meta-analysis of 115 published studies to examine the responses of 19 indicators related to belowground C and N cycling to livestock grazing in

global grasslands. They found that grazing intensity significantly altered belowground C and N cycling. Low grazing intensity contributed to soil C and N sequestration while moderate and high grazing intensity increased C and N losses. Their results showed that moderate and intensive grazing significantly decreased belowground C and N pools, with the largest decreases in microbial biomass C and N. Pools of C and N were also affected by soil depth, livestock type and climatic conditions. Therefore, grazing lands may act as a net sink of emitted GHGs by sequestrating carbon, and thus having a high potential to offset a substantial portion of GHG global mean forcing through the use of optimal grazing management [53-57].

2.2. Inconsistences of grazing effect on SOC and GHGs among regions

Although enormous data of grazing lands have been collected on a large number of variables, including soil properties (SOC, N, soil moisture, density, clay fraction) and management (grazing intensity, irrigation and fertilization), our understanding of underlying processes regulating soil C and N remains piecemeal and quantification of environmental impacts is still insufficient. McSherry and Ritchie [53] suggested that grazers in different regions may need to be uniquely managed to mitigate GHGs because the effects of grazing on SOC stocks and GHGs are highly variable. Inconsistent effects of grazing on SOC stocks in different regions rely heavily on climate/weather conditions, soils, hydrologic processes and vegetation type, and on grazing management and intensity at a variety of scales [53, 58,59]. Agro-climatic regions are particularly important for determining the vulnerability of grazing lands to climate change. For example, warmer and wetter climates can produce a shift in grassland species composition from C_3 to C_4 plant species. White et al. [60] reported that herbage quantity and quality responses to climate changes were mainly influenced by warming and reduced precipitation at three northern temperate grassland sites across the Canadian prairies. Reduced precipitation decreased season-long accumulated herbage. Both, warming and reduced precipitation decreased herbage quality (protein content) but clipping increased quality. In tropical areas with higher rainfall, moderate grazing or higher stocking rates can be positive for SOC storage due to greater pasture growth. This, in turn, can increase soil attributes such as organic matter and water-holding capacity. In contrast, intense grazing in arid and semi-arid grazing lands may reduce vegetation cover due to slow pasture growth, resulting in increases of wind erosion and a coarser soil texture. Hebbs et al. [61] compared effects of land use (native grasslands, introduced pastures and annual croplands) on soil physical properties using indictors (the S-index of the inflection point of the slope of the moisture retention curve, hydraulic conductivity, and pore size fractions) in the Canadian prairies. They found that native grasslands had the greatest soil quality, and croplands the lowest. The change of these indicators can cause degradation of grasslands to become more vulnerable to SOC losses. However, evidence is minimal that a stocking reduction in multi-paddock grazing can lead to improved ecological outcomes, e.g. soil C sequestration. Considerable uncertainty exists about the impact of large herbivore grazing on SOC, especially for subtropical grazing lands and impact of grazing on partitioning of C allocation to root tissue production compared to fine root exudation [57]. Currently, the evidence on C sequestration does not justify extensive promotion and adoption of intensive multi-paddock grazing strategies, especially in arid and semi-arid regions.

2.3. Inconsistences between grazing researchers and rachers' practice

The debate regarding the benefits of multi-paddock grazing to productivity, compared to less intensive continuous grazing, has been prolonged due to inconsistences between grazing study methodologies, as well as the actual practices used by ranchers [8-10]. This includes: (1) nuanced differences in grazing treatments, contrasting methods during soil sampling and chemical analyses, and variation in statistical analyses to compile data over the long-term,

leading to different formats, accuracies, units and structures of long-term data, and ultimately to inconsistent interpretation of results and perception [54], (2) varied vegetation and soil responses to grazing along specific abiotic ecological gradients [36], and (3) high temporal and spatial heterogeneity.

These inconsistences are mainly due to effects of spatial and temporal scales on both economic and ecological processes, in addition to inherent intra- and inter-annual variability of grazing land systems [10,19, 20, 62-64]. Due to the large number of regulating factors within these grazing systems (Climate × Soils × Vegetation × Animal Species × Management), even a large number of field trials, all previously conducted field experiments cover only small parts of the real-world complexity. Overall, most data lack long-term, spatiotemporal and methodological generality, obscuring the mechanisms governing grazing impacts that determine subsequent ecosystem responses to different climates and environments, or 'non-analogue' conditions [54]. Despite of efforts to optimize grazing strategies moving from continuous to intensive multi-paddock grazing, long-term sustainable goals of grazing management have not been reached [19, 65,66]. Further studies should consider specific characteristics of different indicators in the context of their regional environments [36].

3. Biogeochemical -Agroecosystem (BGC-AES) models

Developing grazing ecosystem models can effectively support efforts to improve productivity and enhancing ecosystem services while tackling climate change impacts. Many process-based models have been developed for grazing lands, such as the Pasture Simulation model (PaSim) [67], NGAUGE [68], Denitrification Decomposition (DNDC) [26, 69], DayCent [28], Roth-C [70]. Here, we do not to perform a full review of various agro-ecosystem models but illustrate some common approaches of simulating grazing lands through several typical BGC-AES models. We selected the DNDC and DayCent models because they represent the current state of the art knowledge as ecosystem framework models [71-73]. This allows to examine what challenges account for grazing lands in the BGC-AES models and identify these problems or knowledge gaps in the grazing modelling.

Both the models require an enormous amount of diverse data, which is a formidable challenge but an essential step towards improving our understanding and ability to simulate C and N dynamics in grazed ecosystems. We are essentially dealing with four domains, i.e., plant, soil, animal (+human), and atmosphere as well as integration of these domain models for forage productivity and nutrient cycles [74, 75]. However, both the models are limited due to poor understanding of the long-term impacts of grazing practice on SOC and GHG emission dynamics.

3.1. Denitrification and Decomposition model (DNDC)

The DNDC is a process-based simulation model of C and N dynamics at daily time step. In the DNDC model, a soil thermo-hydraulic component calculates soil temperature, oxygen and moisture profiles using daily weather data and soil properties with consideration of uptake of water and nutrients by plants. These soil physical variables feed into the model components of SOM decomposition and nitrification and denitrification. In the component of decomposition, manure, fertilizers, and litters are partitioned into N and C pools assuming different decay rates.

The nitrification and denitrification rates are calculated by Michaelis–Menten kinetics for ammonium, nitrite and nitrate, as well as dissolved organic carbon (DOC). Nitrification occurs in well aerated conditions of the water filled pore spaces (WFPS) [76]. Denitrification occurs under anaerobic environment which is formulated in a hierarchy redox reaction. Each step of denitrification is formulated through Michaelis–Menten kinetics controlled by the competition between DOC and NOx. The processes consider different soil state variables,

such as soil temperature, soil moisture, redox potential, and pH. N_2 and N_2O production are calculated using microbial growth and death, nutrient consumption, and gas diffusion. The decomposition component partitions soil N and C pools from SOM (i.e. residues, soil microbes and fertilizers). All organic components distinguish four major SOC pools, namely residues, microbial biomass, humads, and humus. These pools are separated into two or three sub-pools representing the very labile, relatively labile and resistant fractions of the pool. The humus pool represents the passive humus, relatively resistant to decomposition. The plant growth component calculates daily root respiration, plant growth, and water and N uptake by the growing vegetation, controlled by climate, nutrient and soil water status. Effects of agricultural practices (fertilization, irrigation, tillage, crop rotation, and manure amendments) are incorporated into the model. Therefore, input data of land management practices are a large source of uncertainty in the N_2O emissions from grazing lands [26, 77,78].

The DNDC model has been parameterized for the main types of grazing management to describe C and N turnover and GHG emissions under different grazing intensities, including the range of partitioning between above- and below-ground biomass. Livestock manure applied to grazing lands and faeces from grazing animals can stimulate forage production. The combined model has been developed for soil and pasture of different grazing lands. Wang et al. [77] modified the UK-DNDC to represent an enhanced livestock grazing function. Processes in grazed pastures were parameterized according to grazing animal type, number, weight and timing. The amount of dung and urine from each animal was calculated to obtain the total amount of dung and urine produced per site. Grazing intensity per hectare was defined as the product of grazing animal days, with accounted for animal weight during each grazing event.

3.2. DayCent model

DayCent is the daily time-step biogeochemical model based on CENTURY [27,28, 79], which consists of several submodels of SOM cycling, denitrification, nitrification, and phosphorus dynamics. It models soil C and N cycle, GHG emissions (e.g. N_2O, NO, N_2, NH_3, CH_4 and CO_2), plant growth, and net primary production in croplands, grasslands, forests and savannahs. The DayCent can simulate a wide range of crops and grasslands by altering a number of plant specific parameters. Plant production is simulated using a maximal production function constrained by temperature, water, and nutrients. Plant biomass uses five pools for live shoots and roots; standing dead plant material and actual plant biomass production are functions of a genetic maximum defined for each crop. Plant residues are allocated into structural and metabolic pools within the SOM sub-model. Each pool is divided into three SOM pools with different turnover rates. Agricultural practices (such as fertilization, tillage, irrigation, cutting and grazing) can be integrated into grazing lands daily, with a fraction of the total aboveground biomass removed to mimick biomass, and where the latter in turn, is regulated by the type of grazing system.

The total N_2O emissions from nitrification occur only in the dry conditions or low WFPS of the soil. The nitrification rate is based on net nitrogen mineralization, soil ammonium and nitrate level, subjected to WFPS, soil temperature and pH [80]. Denitrification occurs in the wet conditions of the soil WFPS. The denitrification rate is calculated using temperature, soil moisture, and ratio of nitrate to SOC. The gaseous N species are partitioned using soil moisture and temperature [80].

Owen et al. [81] simulated long-term impacts of manure amendments on SOC and GHG dynamics of rangelands using DayCent in California, USA, comparing C and N stocks of manured and non-manured fields on commercial dairy farms. Manure amendments increased SOC pools by 19.0 ± 7.3 Mg C ha^{-1} and N pools by 1.94 ± 0.63 Mg N ha^{-1} in the 0–20 cm

surface soil layer, respectively, compared to non-manured lands. N_2O emissions were proportional to total N additions and offset 75–100% of C sequestration. They also simulated long-term historical (1700–present) and future (present–2100) effects of manure management on soil C and N dynamics, net primary productivity (NPP), and GHG emissions. Modelling uncertainty was due to user adjustable parameters, such as organic matter input rates and the maximum nitrification rate. It is also due to the way the model handles some parameters, e.g. keeping soil bulk density and pH constant through time despite their most likely change. Henderson et al. [82] using the Century and DayCent models for similar simulation scenarios, evaluating the net GHG mitigation potential for the world's native and cultivated grazing lands.

3.3. Comparison of process-based BGC-AES models

Abdalla et al. [71] compared DayCent and DNDC to simulate N_2O emissions from cut and extensively grazed pasture located at the Teagasc Oak Park Research Centre, Ireland. DayCent underestimated N_2O flux for the control plots (deviation of -57% from measured) while DNDC over-predicted the measured flux with relative deviations of +132 and +258%, largely due to overestimation of the SOC. Therefore, both models required calibration for simulating their response to N fertilizer induced and background fluxes. Sandor et al. [83] examined eight process-based models at five grassland sites (in France, New Zealand, Switzerland, United Kingdom and United States) comparing their sensitivity of modelled C and N fluxes to changes in grazing animal numbers (from 100% to 50% of the initial livestock densities) in combination with decreasing N fertilization levels (reduced to zero from the initial levels). They found that simulated patterns of enteric methane emission were characterized by high model-to-model variability and alerted to the limitation to represent management variables and the lack of comprehensive validation data sets. Nevertheless, it is important to compare alternative modeling approaches for grassland and analyze their response uncertainty to variation in grazing management and explore the possibility of using models to determine sound mitigation practices. The DayCent and DNDC models provided good results when used to investigate the effects of arable crop production using data from long-term experiments in three locations on the Canadian prairies [72]. Both models simulated SOC change and N_2O emissions for N fertilized wheat systems well but microbial processes (nitrification or denitrification) and their interactions with one another required further improvement.

Ehrhardt et al. [73] conducted an international model comparison and benchmarking exercise of 24 process-based biogeochemical models for assessing uncertainties in crop and pasture simulations of productivity and N_2O emissions. Across sites and crop/grassland types, 23%–40% of the uncalibrated individual models were within two standard deviations of observed crop yields, while 42 (rice) to 96% (grasslands) of the models were within 1 standard deviations of measured N_2O emissions. However, the principles have to be maintained in spite of the significant uncertainties of underlying processes at multiple spatial and temporal scales [77]. From this point of view, the process-based model can be improved for the relationships (or their absence) through the expression of the spatial and temporal trajectories of the stochastic grazing processes studied by clarifying the ambiguity of the grazing and stocking rates or grazing intensity.

Ehrhardt et al. [73] questioned the use of model ensembles for upscaling projections of agricultural productivity and N_2O emissions from field scale to larger spatial units due to soil spatial variability which is likely to reduce the accuracy of model projections. Current biogeochemical models usually do not differentiate the effects of grazing intensities on hydrological processes [84-85] due to lacking representation of terrain morphology and lateral flows, which often creates a great challenge to the prediction of future feedbacks among the grazing intensity, sediment, nutrient runoff and the C and N cycle.

4. Watershed-scale Modeling

In many watersheds, grasslands above headwater streams and ponds can contribute to flooding control, sediments and pollutants, retaining nutrients, and maintaining biological diversity as water infiltration for downstream reaches, lakes and estuaries [86]. Inversely, hydrological processes can be dominant for plant production and nutrient use efficiency in grazing watersheds. Therefore, watershed-scale modeling can be a powerful instrument for simulating changes in nutrient runoff and water quality from grazing lands under diverse environmental and management scenarios. This may overcome the weakness of hydrological effects in the above BGC-AGC models. The Soil Water Assessment Tool (SWAT) and the Variable Infiltration Capacity (VIC) are two typical of watershed models [31, 87-89]. The SWAT is a land surface-atmosphere interactions model that has its origin in hydrological science, while the VIC is a process-based hydrologic model that considers the energy and water balance across the land surface [31, 90,91]. They are ideal to illustrate the typical of processes in watershed modelling and to examine what challenges and knowledge gaps in grazing watershed modelling.

4.1. Soil & Water Assessment Tool (SWAT)

The Soil & Water Assessment Tool (SWAT) is a hydrological model at a watershed and river basin-scale. Digital Elevation Model (DEM), land use, and soil are used for generation of hydrological response units (HRUs), which includes unique combinations of land use, soil and slope. It is widely used in assessing the quality and quantity of surface and ground water, soil erosion prevention and control as well as in predicting the environmental impact of land use, land management practices, non-point pollutant sources and climate change [30, 87, 89].

SWAT uses conceptual C and N pools with varying turnover rates [30], and in the original SWAT, the addition of manure, fertilizers, and residues are partitioned into five N pools (ammonium, nitrate, active, stable and fresh N) and into one single SOM pool. Nutrient cycling (i.e., C and N) are controlled by soil factors (i.e., temperature, moisture, aeration and clay fraction) using integrated soil, plant and microbial processes. Fresh N is associated with plant residue and microbial biomass. The stable and active pools represent the soil humus.

The conceptual C and N pools in the SWAT were originally developed for assessing nutrient runoff and leaching related to water quality. The response of grasslands and rangelands have been simulated using manure applications from domestic animals, direct shedding (excretion) by livestock, estimating directly release to streams or groundwater and runoff to surface water system [87, 88, 92]. Chanasyk et al. [92] simulated surface runoff from grassland watersheds under three grazing intensities: ungrazed (control), intensive and very intensive (2.4 and 4.8 animal unit months per hectare, respectively) in Southern Alberta. Their surface runoff patterns showed large summer storm runoff rates from intensive grazed compared to other watersheds and large snowmelt-induced runoff from very intensive grazed. Dakhalla and Parajuli [93] simulated streamflow, total sediment, total phosphorus (TP), and total nitrogen (TN) load in Big Sunflower River Watershed (BSRW). The animal population densities were calculated using the animal populations divided by the amount of grassland in the watershed. Runoff and sediment in small irrigated watershed in the Canadian prairie was investigated by Rahbeh et al. [94] using SWAT. The irrigation activity did not change the water partitioning among the existing hydrological pathways but had temporal effects on the magnitudes of runoff and, more importantly, deep percolation and the subsequent groundwater discharge in the main reach. Shrestha and Wang [88,89] simulated nutrient runoff and water quality in the Athabasca River Basin (ARB), Alberta, Canada. Future climate of the ARB was projected to be warmer and wetter, relative to the base period. They found that climate change can lead to decreased carbonaceous biochemical oxygen demand concentrations, mainly due to dilution and increased degradation. Further, a warmer future climate was found likely to increase the temperature which in turn will reduce the

dissolved oxygen concentrations. Therefore, under a changing climate, these cold regions could become more vulnerable than others because some special geographic features of the regions, such as glaciers, freezing soils and peatland, are more sensitive to changes in temperature and precipitation [95-97]. This can impose serious threats on the water resources, sustainable goods production and ecosystem services that depend on regional water quality. Melaku and Wang [98] modified the SWAT model to improve the module of evapotranspiration in bi-directional groundwater-surface water exchange. They successfully simulated dynamics of water table at Lethbridge and Barons, Alberta, Canada. The two-way interactions of groundwater-surface water can improve representation of water fluxes in grassing lands. Park et al. [99,100] simulated effects of alternate grazing management practices on water quality at the ranch and watershed scales. They compared four grazing practices: heavy continuous, light continuous and adaptive multi-paddock grazing, and no grazing. Their results showed that grazing management practices can change vegetation cover and soil properties as well as watershed hydrological processes. The sediment, TN and TP loads at the watershed outlet can be reduced by 39.7%, 35.1% and 34.1%, respectively, when the grazing management in the entire rangelands of the watershed was changed from baseline heavy continuous to the adaptive multi-paddock grazing. Therefore, heavy continuous grazing management with high stocking rates can cause substantially higher levels of runoff, sediment and nutrient losses to surface water systems.

However, the biogeochemical processes in the original SWAT are simplified. Residue and manure are two addition sources of SOM, in which SOM from different sources are bulked with existing C and N pools. Only one carbon pool represents all carbon turn over. Animal excretion additions have not considered key components, such as time, duration, animal type and number, rotational and so on for managed grasslands. Therefore, the SWAT needs to be strengthened to simulate grazing lands.

4.2. Variable Infiltration Capacity (VIC) Model

As a large-scale process-based hydrologic model, the VIC model has basic features of land surface models (LSMs) to simulate the exchange of surface water and energy fluxes at the soil-atmosphere interface [101,102]. The VIC model is able to capture subgrid variability of land surface vegetation, soil moisture storage capacity, and topography (e.g., reflecting orographic precipitation and temperature lapse rates). VIC has a notable strength to model hydrologic processes in cold regions (e.g., frozen soil, limited thaw depths, snow accumulation, ablation and melting processes) [103,104]. Over the last decades, VIC has been upgraded many times and expanded to multiple soil layers [102], dynamically coupling with a two-layer energy balance snow model (Andreadis et al., 2009), frozen soil [103], permafrost algorithm [105], soil temperature heterogeneity method [104], water table depth from soil moisture and texture [106], blowing snow algorithm [107], and elevation bands algorithm [108]. VIC has also been recently updated to include a dynamic lake/wetland model that simulates permanent lakes, seasonal flooding of vegetated land and timely-varying exposed fraction of land covers within a grid cell [109]. With these improvements, the VIC model has been extensively applied to assess impacts of climate and land-cover changes on hydrologic systems [31, 90,110] and drought [111], and floods [112-113].

VIC has been modified to dynamically couple processes of the carbon cycle with photosynthesis, autotrophic respiration, and heterotrophic respiration in the three SOC reservoirs [102]. Wetlands are recognized as carbon sinks, which store excess carbon from the atmosphere [114]. Considering the use of wetlands, its C stored is released to the atmosphere, degradation and drainage of wetlands leads to methane release to the atmosphere. All fluxes of GHGs are highly dependent on soil moisture and temperature [115]; therefore, modeling of dynamic lake/wetland areas is crucial to simulate GHG fluxes. Bohn et al. [102] integrated a modified wetland methane emission model for the Western

Siberian Lowlands [116] using the VIC's soil temperature, NPP, and water table distribution modules. To simulate the effects of grazing with the VIC model, biochemical models and parameters would need to be adapted to simulate the essential GHG fluxes affected by grazing management.

5. Current Trends in modelling of grazing grasslands

Grazing systems contain complex interactions between grazing practices, and both biogeochemical and hydrological drivers, often interact in unpredictable ways. The spatio-temporal heterogeneity of grazing lands coupled with highly variable management inputs minimizes the likelihood that a given management practice will consistently produce similar outcomes in all cases [63]. Moreover, management of grazing lands requires highly adaptive approaches. Therefore, grazing management and environmental conditions in specific field trials are unlikely to be reproduced due to the infinite possible combinations of soil, water, climate, pasture species, livestock and farm practices. These field data represent a small part of the huge number of possibilities among grazing management, soil, water, grasses and climate, and therefore remain inadequate to represent key grazing processes. It is necessary that the knowledge of the grazing management is transferable between different trials and practitioners' grazing management with a certain degree of generality. System modelling is key to effectively facilitate the learning required to create management strategies that fit specific economic and ecological conditions, and in turn, accommodate the inherent uncertainties of grazing land systems. However, challenges to couple grazing management practices, monitoring networks and system models exist in four categories: 1) scaling-up of grazing management practices, 2) integrating the extreme complexity of animal movement, grazing intensity and excretion based on the highly adaptive nature of grazing, 3) linking SOM decomposition with soil microbial activities in land surface models, and 4) integrating the biogeochemical and hydrological processes for the spatiotemporal heterogeneity at catchment scale.

5.1. Upscaling grazing management practices

Experimental data from small paddock trials may not be consistent with producer experience on both, spatial and temporal aspects of grazing management. A higher production under multi-paddock rotational grazing trials may not be reproduced by practitioners across all conditions. The conflicting management practices firstly are due to effects of spatio-temporal differences on, both, economic and ecological goals between the trials and on-ranch adaptive grazing management [20, 117, 118]. Fig. 2 illustrates the integration and scale-up of grazing models and management. Generally, the scale-up of grazing management can be categorized to the three types: (1) spatial scale-up, (2) temporal scale-up and (3) implementation and knowledge transfer. Normally, the scale-up denotes spatial and temporal upscaling. A spatial scale-up is a process that the research results are scaled up from plot or field scale to regional scale, e.g. county or country, while a temporal scale-up transfers/extrapolates research results from one to another time period, from present studies to the future. The knowledge transfer is somehow a specific spatio-temporal scale-up for grazing management, in which research results from field trials (Plot A) can be reproduced on-farm trials (Plot B) (Fig. 2).

The productivity and C and N cycles are strongly interrelated with soil, climate, and grazing management conditions as well as with time [119, 14]. Despite the vast number of grazing experiments conducted since the 1980s, the majority follow short-term and rigid rotational protocols to increase replicability [10]. However, in practice, rotations and conditions cannot be the same as ranchers' practices and future environments vary. For example, the environments on Plot B is likely to be different from that on Plot A (Fig. 2). Therefore, these results of grazing studies do not necessarily work for other pedoclimatic

environments and ecological purposes (e.g. biodiversity, SOC stocks and GHGs). They do also do not contain the adaptive/flexible treatments that ranchers actually employ under real world variability. For example, the present benefits of multiple paddock grazing management at Plot A may not be possible to reproduce in the future.
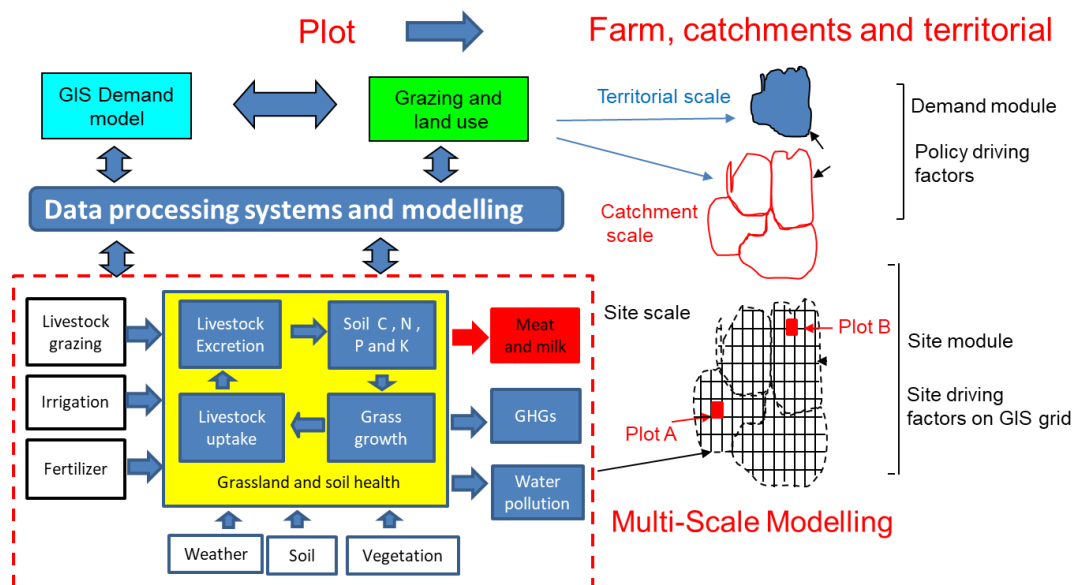

Fig. 2. Upscaling grazing modelling and management: grazing land model (left side) and upscaling (right side)

In-situ monitoring or sampling at field scale is spatially not continuous and is insufficient to cover the full 'landscape' environment [8, 14]. In practice, sampling places and frequency are often limited due to high resource requirement, even at field scale. For example, it is common to use a small number of chamber boxes to measure GHGs from a hectare of grazing land. Soil and vegetation can be highly heterogeneous even at field scale and weather conditions vary. The average result of all samples from these chambers depends on the heterogeneity of the field and the placement and distribution of these chambers. It is often unknown how many samples should be taken and distributed to ensure the real averaged representation of the site. In conclusion, it is difficult to capture spatial and temporal patterns, which can lead to substantial uncertainties. Additionally, grazing management is very complex with variable factors, such as stocking rate, period, number types, animal weight and so on. Even if we fix rigidly these grazing management, GHGs from grazing land and plant production depend on soil, water, grass species, and climate. Therefore, long-term detailed comparisons of multi-paddock grazing methods and practices are useful to resolve spatio-temporal heterogeneity at scale but cost- and time-effective. Therefore, it is not surprising that many researchers have failed to sufficiently account for these management and environmental variations, either in replication of their treatments or in their evaluation.

A quantitative accounting of the potential managerial contributions to the success of multi-paddock rotational grazing systems across scales is a prerequisite for addressing these inconsistences. State of the art agroecosystem models, such as DNDC and DayCent, are potential tools to account for various interactions between grazing management parameters, as listed above. For large scale regions it is common, in the current process-based models, to

use grid systems to subdivide a studied region (Fig. 2). Gridded input variables, such as weather, soil properties, vegetation and agricultural practice, are provided to the models. Therefore, the spatial resolution is determined by a grid node/cell size in which the parameters of soil properties and weather parameters are area-, mass- or volume-averaged and different types of vegetation area is allocated to use the fraction. Therefore, high resolution simulations will depend on data availability of soil and vegetation and computer power. Because all these parameters can be combined for any scenarios, the ranchers' practices can be simulated. As a result, the results of an grassing land model can be used for interpolation and extrapolation of missing field data. These methods are potentially applicable where results of multi-paddock rotational grazing are used for assessing rangelands over decadal time frames at large scale.

5.2. Animal movement and grazing intensities

Modelling of different grazing ecosystems is challenging as there are intrinsically different parameterization of grazing intensity that represents grazing animal activities [77, 120-122]. To date, mainstream SOC dynamics models applied to grazing systems have not yet recognized these distinctions due to complex grazing management practices and livestock excreta deposition in their simulations. This could result in uncertainties assessing the soil C and N cycles in managed grasslands. Variables associated with grazing intensity included a mixture of grazing management (stocking rates, pasture rest times) and animal preferences (water and pasture) and farm characteristics (land types, climate regions, and ecoregions).

Variation and movement of animals and pastures are major sources of uncertainties that can bias results if experimental designs or sampling techniques are inappropriate. Multi-paddock grazing requires logistic details: the paddock size, position of water and gates, stocking rate, and timing of movement through the rotation. Timing is dependent on many variables, such as stocking rate and forage quality and quantity. However, the key component required for modelling GHGs, nutrient cycle and soil health and grass regrowth is the movement and preferences of animals within a paddock, which ultimately determines grazing and nutrient heterogeneity. The latter, in turn, will depend on the distribution of palatable species and water sources but also on the favorites of different types of grazing animal. Knowledge of the grazing intensity and distribution is also essential for influencing the frequency of animal moves based on the desired animal number and rate of weight gain to develop process-based models for rotational grazing systems. Failure to integrate plant and animal processes at appropriate temporal and spatial scales can result in an incorrect representation in the process-based models for grazing land management. The lack of effective monitoring animal movement at a large spatial scale severely limits research on the mechanism, model development, and simulation of animal behaviors. Without that, the grazing intensity cannot be quantified, which results in insufficient analysis and simulation of processes dependent on animal movement and grazing intensity. Improved monitoring of animal movement and behavior based on remote sensing (RS) might be helpful to understand and assess the impact of animal movement processes at scale. However, this needs a new multi-disciplinary and multi-partnerships collaborative effort.

The complexity within these models originates from the multiplicity of factors involved and resulting spatial and temporal variabilities of animal activities, their prior use patterns and trajectories. Therefore, causal relationships between animal activity and grazing intensity are challenging to establish but remain fundamental to building the grazing model and understanding these causal links. The process-based model can be improved by building in stochastic or deterministic functions (or their absence) to express the spatial and temporal trajectories of grazing. Deterministic or stochastic to quantify the impact of specific grazing patterns cause by stocking rates or grazing intensity, as well as length of grazing.

5.3. SOM decomposition kinetics

There are three areas of interest for possible improvements of C and N models: i) Decomposition kinetics affected by SOM heterogeneity; ii) soil hydrology: lateral flow and its effects on reaction rates; and iii) grazing management: effects of manure distribution and heterogeneity modifying soil physical, chemical and biological properties [84,85, 123-125].

Current land-surface models (LSMs) usually do not differentiate the effects of grazing intensities on belowground C and N cycles [82, 120, 126,127], which is a potential challenge to the prediction of future feedbacks between climate and C cycle. Future land-surface models may need to differentiate grazing intensity in order to develop a more precise process-based mechanism to forecast the feedback of grassland ecosystems on climate change. Over the past decades, much effort has been made in developing process-based models of key biogeochemical processes of SOM turnover, such as, DNDC [26, 128,129], CENTURY and DayCent [27,28], Roth-C [70], VIC [102], and SWAT [30], are based on conceptual pools of C and N turnover at variable rates. Each pool can be assigned a mean residence time (MRT) within the spatially averaged soil and is affected by the reactivity and environmental constraints on the SOM decomposition [130].

Inconsistent results are due to the underlying assumptions for the kinetics of SOM decomposition rates of these carbon pools and their 'volume averaged' or 'mass averaged' approaches in the conventional process-based models, still subjected to debate [130]. Microbial activities substantially depend on spatio-temporal variations in water, temperature and substrate availability [131,132]. Brilli et al. [123] suggested that future development of C and N models should explicitly account for soil microbial biomass to drive SOM turnover. Further, they must include the effect of N shortage on SOM decomposition and improve the production and consumption of gases using an adequate description of gas transport in soils. This implied possible limitations in the underlying hypotheses of C pools from the literature in the cases where discrepancies between model and observation. Despite this extensive analysis, knowledge basic mechanisms driving C and N cycles in agricultural systems is still far from complete and key questions remain about the cascade of events that finally lead to biological responses. Recent studies [124,125, 131, 132] formulated the sequential oxidation-reduction potential (ORP) and chemical reactions undergoing in the soil-water zone using dual Arrhenius and Michaelis-Menten (DAMM) kinetics to contribute towards more reliable estimates of C and N cycles under grazing systems.

5.4. Coupling biogeochemical and hydrological processe for the spatial and temporal heterogeneity at catchment scale

Grazing lands are either a watershed or parts of a watershed. Therefore, a watershed is a natural ecosystem within natural catchment boundaries, in which natural processes (e.g., hydrological and biogeochemical) and socio-economic processes (e.g., human actives) interact. However, the current studies of such complex ecosystems are artificially segregated by disciplines, such as hydrology, biochemistry, geology and agriculture. Despite clear boundaries between different disciplinary programs, there are no such interfaces occurring in the real-world problems like sustainable productivity grazed watersheds. Biogeochemistry and hydrology are two of the main natural processes in river basins. Because of the historical disciplinary separation, the former is studied by biogeochemist, geologist and soil expert while the latter does by hydrologist, physicist and engineers. Interdisciplinarity is required for the grazing watershed problems of today, and out of an interest for the past unrelated to present day concerns from within the discipline itself or from a more general starting point. Therefore, it is necessary to integrate disciplines to understand such a complex ecosystem, which is a big challenge because disciplines use different methods, tools and terminologies.

This creates difficulties to communicate effectively and to build interfaces and connections between different disciplines.

Developing grassland models can effectively support efforts to enhance ecosystem services and productivity [118]. However, hydrological processes in grazing lands may be dominante to affect biogeochemical processes. Due to different disciplines, many hydrological processes have been simplified much in the widely-used BGC-AES [133], such as snowmelt and freeze-thaw cycle [134, 135], ground water [136] and lateral fluxes [84]. Groffman et al. [133] showed the importance of the effects of hydrological processes on C and N cycles, which may need to be incorporated into regional and global models for predicting effects of human disturbance on global grasslands and assessing the climate-biosphere feedbacks. More importantly, typical water cycle regions are river basins acting as natural water system boundaries. Therefore, models could be able to simulate the lateral flow and distribution of water, nitrogen and carbon within landscapes, stream flows, and sediment and nutrients runoff in river networks. Further work remains on spatially representative from different grazing ecosystems. It is, therefore, of paramount importance that interactions between different processes and controls of soil carbon and nutrient cycles and nutrient runoff to surface water are reflected in the models. Quantifying such interactions between biogeochemical and hydrological processes should be a research priority [84,85].

In grassland ecosystems, the major sources of GHG (N_2O, CO_2 and CH_4) emissions, leaving aside enteric methane production, are livestock excreta deposition, manure application, and mineral fertilizer application [137]. Current challenge is to find connections for estimating the interaction between biogeochemical and hydrological processes. Because of historical reasons, the widely used BGC-AES models do not account for lateral flows. In recent years, a fully coupled soil nitrification and denitrification module of an agroecosystem model, (i.e. DayCent) has been incorporated into SWAT [84,85]. The algorithms of nitrification and denitrification from the DayCent model are explicitly coupled with SWAT model, which simulates carbon and nitrogen partitioning, to modules of soil carbon and nitrogen turnover in grazing lands. A more detailed microbial and thermodynamic kinetics of SOM turnover has been developed by using DAMM kinetics within the SWAT framework [124,125]. Here the variables of the SWAT, NH_4^+ and NO_3^-, are directly coupled with the submodels of nitrification and denitrifications [84, 85,124,125]. These models are based on the SWAT model framework at watershed scale and therefore, the lateral hydrological flow and substrate fluxes have been considered.

Another modelling challenge in grazing modelling of coupled biogeochemical and hydrological processes is the need for high resolution meshes for simulation of large-scale river basins or regions. With the advent of RS techniques, the high-resolution data of land use and cover, soil and DEM will be available for the grazing modelling. To fully appreciate the large grid required, a major focus of future research in grazing modelling will be on interaction of integrated biogeochemical and hydrological processes at watershed scale. Much remains to be learned about the fundamental physical and ecological processes of grazing.

6. Concluding remarks

During the last decades, managed grazing systems have been studied extensively in terms of livestock productivity, pasture diversity, GHG emission, carbon storage and soil erosion. In this review, we synthesized a diverse body of experimental and modelling evidence. We hypothesized that modelling can help to identify the main factors (environmental, management) that affect grassland and rangeland productivity and ecosystem degradation. This review alerts to some important deficiencies, experimental and systematic knowledge gaps in this area. Our aim was to explore directions in which future

experiments, monitoring, and modelling needs to improve knowledge transfer for better grazing management guidelines.

The principle to link well-established multi-paddock grazing experiments with models have widely been applied but need to be challenged and become an effective guideline. This review examined the methods and progresses to describe interactions between grazing management, nutrient cycles and landscape degradation for currently used grazing land systems. The debate about the benefits of multiple paddock grazing management to plant and livestock production, compared to continuous grazing, is not "essence of these inconsistences" but "appearance of these inconsistences". We found that inconsistences are essentially due to (1) effects of spatiotemporal scales on both economic and ecological outcomes, and (2) simplistic representations of grazing systems. This creates inconsistences between the trials and on-farm adaptive grazing management strategies and scaling-up to different environmental conditions, which fails knowledge transfer.

Complex grazing systems need a comprehensive evaluation that explicitly incorporates qualitative and quantitative knowledge of management, hydrology, environment, and technological and biophysical components of grazing systems. System models are key to effectively integrate multi-layer evidence to create management strategies that fit specific socio-economic and agro-ecological settings and account for the inherent uncertainties of grazing ecosystems. Using models, the results of – preferably on-farm – field trials can be extrapolated to other environments (pedo-climatic = soil x weather) and mapped to validate spatio-temporal patterns. Particularly, system models can compare scenarios of differently managed grasslands using multi-paddock or continuous grazing to clarify key ambiguities of grazing management and help to converge directions of future research and knowledge transfer. Effective grazing management offers great potential to achieve sustainable use of grasslands and rangelands to meet increasing food demands, mitigate climate change and provide ecosystem services. The future of grazing research requires an integrated approach to whole systems' analysis for variable outcomes of grazing systems. However, much remains to be done to build these generic models and formulate evaluation criteria for the main challenges, such as high-resolution (RS) data, microbially-mediated SOM decomposition, grazing intensity for animal movement, and hydrological effects. Scientists and stakeholders should collaborate closely with farmers and ecologists to bridge the knowledge gaps and drive progress in this multidisciplinary field.

Conflicts of Interest: The authors declare no conflict of interest.

References

1. Latham, J.; Cumani, R.; Rosati, I.; Bloise, M. Global Land Cover SHARE (GLC-SHARE) database Beta-Release Version 1.0 – 2014. FOA report. http://www.fao.org/uploads/media/glc-share-doc.pdf

2. Oertel, C.; Matschullat, J.; Zurba, K.; Zimmermann, F.; Erasmi, S. Greenhouse gas emissions from soils—A review. Chemie der Erde 2016, 76,327–352.

3. Gerber, P.J.; Steinfeld, H.; Henderson, B.; Mottet, A.; Opio, C.; Dijkman, J.; Falcucci, A.; Tempio, G. Tackling climate change through livestock – A global assessment of emissions and mitigation opportunities. Food and Agriculture Organization of the United Nations (FAO), Rome. 2013. http://www.fao.org/3/i3437e/i3437e.pdf

4. Scurlock, J.M.O.; Hall, D.O. The global carbon sink: a grassland perspective. Global Change Biology 1998, 4(2), 229-233. DOI: 10.1046/j.1365-2486.1998.00151.x

5. Anderson, J.M. The effects of climate change on decomposition processes in grassland and coniferous forests. Ecological Applications 1991, 1, 326–347.

6. Jacobo, E.J.; Rodríguez, A.M.; Bartoloni, N.;.Deregibus, V.A. Rotational Grazing Effects on Rangeland Vegetation at a Farm Scale. Rangeland Ecol Manage 2006, 59, 249–257.

7. Thomas, B. Maritime pasture manual. Department of Agriculture, Nova Scotia, Canada, 2009.

8. Teague, R.; Provenza, F.; Kreuter, U.; Steffens, T.; Barnes, M. Multi-paddock grazing on rangelands: Why the perceptual dichotomy between research results and rancher experience? Journal of Environmental Management 2013, 128, 699-717.

9. McDonald, S.E.; Reid, N.; Waters, C.M.; Smith, R.; Hunter, J. Improving ground cover and landscape function in a semi-arid rangeland through alternative grazing management. Agriculture, Ecosystems & Environment 2018, 268, 8-14.

10. Briske, D.D.; Derner, J.D.; Brown, J.R.; Fuhlendorf-Doff, S.D.; Teague, W.R.; Havstad, K.M.; Gillen, R.L.; Ash, A.J.; Willms. W.D., Rotational Grazing on Rangelands: Reconciliation of Perception and Experimental Evidence. Rangeland Ecology & Management 2008, 61(1), 3-17.

11. de Groot, R.S.; Alkemade, R.; Braat, L.; Hein, L.; Willemen, L. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making.   Ecological Complexity 2010, 7, 260-272. https://doi.org/10.1016/j.ecocom.2009.10.006

12. Broadbent, T.S.; Bork, E.; Cooke, J.E.; Willms, W.D. Grass Yields Under Clipping and Watering Explain Varied Efficacy of Management Intensive Grazing on Rangelands. Rangeland Ecology & Management 2019, DOI: 10.1016/j.rama.2018.11.009.

13. Müller, B.; Frank, K.; Wissel, C. Relevance of rest periods in non-equilibrium rangeland systems - A modeling analysis. Agricultural systems 2007, 92, 295-317.

14. Teague, R.; Provenza, F.; Norton, B.; Steffens, T.; Barnes, M.; Kothmann, M.; Roath, R. Benefits of multi-paddock grazing management on rangelands: limitations of experimental grazing research and knowledge gaps. In: Schroder, H.G. (Ed.), Grasslands: Ecology, Management and Restoration. Nova Science, New York, pp. 41-80. 2008.

15. Heitschmidt, R.K.; Walker, J.W. Grazing management: technology for sustaining rangeland ecosystems. Rangeland Journal 1996, 18, 194-215.

16. Alberta Agriculture, Food and Rural Development (AAFRD).   Beneficial Management Practices: Environmental Manual for Alberta Cow/Calf Producers. 2004.

17. Orr, R.J.; Murray, P.J.; Eyles, C.J.; Blackwell, M.S.A.; Cardenas, L.M.; Collins, A.L.; Dungait, J.A.J.; Goulding, K.W.T.; Griffith, B.A.; Gurr, S.J.; Harris, P.; Hawkins, J.M.B.; Misselbrook, T.H.; Rawlings, C.; Shepherd, A.; Sint, H.; Takahashi, T.; Tozer, K.N.; Whitmore, A.P.; Wu, L.; Lee, M.R.F. The North Wyke Farm Platform: effect of temperate grassland farming systems on soil moisture contents, runoff and associated water quality dynamics. European Journal of Soil Science 2016, 67, 374–385.

18. Becker, W.; Kreuter, U.; Atkinson, S.; Teague, R. Whole-Ranch Unit Analysis of Multi-paddock Grazing on Rangeland Sustainability in North Central Texas. Rangeland Ecology & Management 2017, 70, 448-455.

19. Briske, D.D.; Ash, A.J.; Derner, J.D.; Huntsinger, L. Commentary: A critical assessment of the policy endorsement for holistic management, Agricultural Systems 2014, 125,50–53.

20. Mann, C.; Sherren, K. Holistic Management and Adaptive Grazing: A Trainers' View. Sustainability 2018, 10, 1848.

21. De Bruijn, S.J.; Bork, E.; Grekul, C.W. Neighbor defoliation regulates Canada thistle (Cirsium arvense) in pasture by mediating interspecific competition. Crop Protection 2010, 29(12), 1489-1495.

22. Petrosillo, I.; Zaccarelli, N.; Zurlini, G. Multi-scale vulnerability of natural capital in a panarchy of social–ecological landscapes. Ecological Complexity 2010, 7, 359-367. https://doi.org/10.1016/j.ecocom.2010.01.001

23. Briske, D.D.; Sayre, N.F.; Huntsinger, L.; Fernandez-Gimenez, M.; Budd, B.; Derner, J.D. Origin, persistence, and resolution of the rotational grazing debate: integrating human dimensions into rangeland research. Rangeland Ecol. Manage. 2011, 64, 325–334.

24. Roche, L.M.; Cutts, B.B.; Derner, J.D.; Lubell, M.N.; Tate, K.W. On-Ranch Grazing Strategies: Context for the Rotational Grazing Dilemma. Rangeland Ecology & Management 2015, 68, 248–256.

25. Hyland, J.J.; Heanue, K.; McKillop, J.; Micha, E. Factors influencing dairy farmers' adoption of best management grazing practices. Land Use Policy 2018, 78, 562-571.

26. Li, C.; Frolking, S.; Frolking, T.A. A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. Journal of Geophysical Research Atmosphere 1992, 97, 9759-9776.

27. Parton, W.J. The CENTURY model. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of Soil Organic Matter Models. Springer, Berlin. 1996

28. Parton, W.J.; Hartman, M.; Ojima, D.; Schimel, D. DAYCENT and its land surface submodel: description and testing. Glob. Planet. Chang. 1998, 19, 35-48.

29. Neitsch, S.L.; Arnold, J.G.; Kiniry, J.R.; Williams, J.R., 2011. Soil & water assessment tool theoretical documentation, version 2009, Grassland, soil and water research laboratory-agricultural research service. Blackland Research Center-Texas AgriLife Research.

30. Arnold, J.G.; Moriasi, D.N.; Gassman, P.W.; Abbaspour, K.C.; White, M.J.; Srinivasan, R.; Santhi, C.; Harmel, R.D.; van Griensven, A.; Van Liew, M.W.; Kannan, N.; Jha, M.K. SWAT: model use, calibration, and validation. Am. Soc. Agric. Biol. Eng. 201255, 18.

31. Eum, H.-I.; Dibike, Y.; Prowse, T.D. Comparative evaluation of the effects of climate and land-cover changes on hydrologic responses of the Muskeg River Alberta, Canada. J. Hydrol.: Reg. Stud. 2016, 8, 198-221.

32. Wang, X.; VandenBygaart, A.J.; McConkey, B.C. Land Management History of Canadian Grasslands and the Impact on Soil Carbon Storage, Rangeland Ecol Manage 2014, 67, 333–343.

33. Wang, X.; McConkey, B.G.; VandenBygaart, A.J.; Fan, J.; Iwaasa, A.; Schellenberg, M. Grazing improves C and N cycling in the Northern Great Plains: a meta-analysis. Scientific Reports 2016, 6, 33190, DOI: 10.1038/srep33190

34. Biondini ME, Patton BD, Nyren PE (1998) Grazing intensity and ecosystem processes in a northern mixed-grass prairie. USA Ecol Appl 8:469–479.

35. Patton, B.D.; Dong, X.J.; Nyren, P.E.; Nyren, A. Effects of grazing intensity, precipitation, and temperature on forage production. Rangeland Ecol Manage 2007, 60, 656–665.

36. Wang, Y.; Wesche, K. Vegetation and soil responses to livestock grazing in Central Asian grasslands: a review of Chinese literature. Biodiversity and Conservation 2016, 25, 2401–2420.

37. Badgery, W.B.; Mitchell, D.; Millar, G.D.; Broadfoot, K.; Michalk, D.L.; Cranney, P.; Brown, W. Designing a grazing-system experiment for variable native pastures and flexible lamb-production systems. Animal Production Science 2017, 57(9), 1785-1798.

38. Savian, J.V.; Schons, R.M.T.; Marchi, D.E.; Freitas, T.S.; Neto, G.F.S.; Mezzalira, J.C.; Berndt, A.; Bayer, C.; Carvalho, P.C.F. Rotatinuous stocking: A grazing management innovation that has high potential to mitigate methane emissions by sheep. Journal of Cleaner Production 2018, 186, 602-608. https://doi.org/10.1016/j.jclepro.2018.03.162

39. Jones, S. K.; Rees, R.M.; Skiba, U.M; Ball, B.C.   Greenhouse gas emissions from a managed grassland. Global and Planetary Change 2005, 47(2-4), 201-211. https://doi.org/10.1016/j.gloplacha.2004.10.011

40. Hewins, D.B.; Lyseng, M.P.; Schoderbek, D.F.; Alexander, M.; Willms, W.D.; Carlyle, C.N.; Chang, S.X.; Bork, E.W. Grazing and climate effects on soil organic carbon concentration and particle-size association in northern grasslands. Scientific Reports 2018, 8, No 1336 (2018).

41. Teague, W.R.; Apfelbaum, S.; Lal, R.; Kreuter, U.P.; Rowntree, J.; Davies, C.A.; Conser, R.; Rasmussen, M.; Hatfield, J.; Wang, T.; Wang F.; Byck, P. The role of ruminants in reducing agriculture's carbon footprint in North America. Journal of Soil and Water Conservation 2016, 71, 156-164.

42. Zhou, G.; Zhou, X.; He, Y.; Shao, J.; Hu, Z.; Liu, R.; Zhou, H.; Hosseinibai, S. Grazing Intensity Significantly Affects Belowground Carbon and Nitrogen Cycling in Grassland Ecosystems: A Meta-Analysis. Global Change Biology 2017, 23, 1167–1179.

43. Badiou, P.; McDougal, R.; Pennock, D.; Clark, B. Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian prairie pothole region. Wetlands Ecol Manage 2011, 19, 237–256.

44. Gourlez de la Motte, L.; Mamadou, O.; Beckers, Y.; Bodson, B.; Heinesch, B.; Aubinet, M. Rotational and continuous grazing does not affect the total net ecosystem exchange of a pasture grazed by cattle but modifies CO_2 exchange dynamics. Agriculture, Ecosystems & Environment 2018, 253, 157-165.

45. Li, C.; Hao, X.; Ellert, B.H.; Willms, W.D.; Zhao, M.; Han, G. Changes in soil C, N, and P with long-term (58 years) cattle grazing on rough fescue grassland. J. Plant Nutr. Soil Sci. 2012, 175, 339–344.

46. Zhang, B.; Thomas, B.W.; Beck, R.; Liu, K.; Zhao, M.; Hao, X. Labile soil organic matter in response to long-term cattle grazing on sloped rough fescue grassland in the foothills of the Rocky Mountains, Alberta, Geoderma 2018, 318, 9-15.

47. Gao, X.; Thomas, B.W.; Beck,   R.; Thompson, D.J.; Zhao,   M.; Willms,   W.D.; Hao,   X. Long-term grazing alters soil trace gas fluxes from grasslands in the foothills of the Rocky Mountains, Canada, Land Degrad Dev 2018, 29 (2), 292-302.

48. Bikila, N.G.; Tessema, Z.K.; Abule, E.G. Carbon sequestration potentials of semi-arid rangelands under traditional management practices in Borana, Southern Ethiopia. Agriculture, Ecosystems & Environment 2016, 223, 108-114.

49. Whitehead, D.; Schiper, L.A.; Pronger, J. et al. Management practices to reduce losses or increase soil carbon stocks in temperate grazed grasslands: New Zealand as a case study. Agriculture, Ecosystems & Environment 2018, 265, 432-443.

50. Garnier, J.; Anglade, J.; Benoit, M.; Billen, G.; Puech, T.; Ramarson, A.; Passy, P.; Silvestre, M.; Lassaletta, L.; Trommenschlager, J.-M.; Schott, C.; Tallec, G. Reconnecting crop and cattle farming to reduce nitrogen losses in river water of an intensive agricultural catchment (Seine basin, France). Environ. Sci. Policy 2016, 63, 76–90.

51. McCarthy, J.; Delaby, L.; Hennessy, D.; McCarthy, B.; Ryan, W.; Pierce, K.M.; Brennan, A.; Horan, B. The effect of stocking rate on soil solution nitrate concentrations beneath a free-draining dairy production system in Ireland. Journal of Dairy Science 2015, 98, 4211–4224.

52. Coffey, E.L.; Delaby, L.; Fleming, C.; Pierce, K.M.; Horan, B. Mlulti-year evaluation of stocking rate and animal genotype on milk production per hectare within intensive pasture-based production systems open archive. Journal of Dairy Science 2018, 101(3), 2448-2462.

53. McSherry, M.E.; Ritchie, M.E. Effects of grazing on grassland soil carbon: a global review. Global Change Biology 2013, 19, 1347–1357.

54. Singer, M.J.; Munns, D.N. Soils: an Introduction. 6 edn, (Pearson Prentice Hall), 2006.

55. Abdalla, M.; Hastings, A.; Chadwick, D.R.; Jones, D.L.; Evans, C.D.; Jones, M.B.; Rees, R.M.; Smith, P. Critical review of the impacts of grazing intensity on soil organic carbon storage and other soil quality indicators in extensively managed grasslands, Agriculture, Ecosystems & Environment 2018, 253, 62-81.

56. Williams, J.W.; Jackson, S.T. Novel climates, no-analog communities, and ecological surprises. Front Ecol Environ 2007, 5, 475–482.

57. Wilson, C.H.; Strickland, M.S.; Hutchings, J.A.; Bianchi, T.S.; Flory, S.L. Grazing enhances belowground carbon allocation, microbial biomass, and soil carbon in a subtropical grassland, Glob Change Biol 2018, 24(7), 2997-3009.

58. Belknap, J.; Welter, J.R.; Grimm, N.B.; Barger, N.; Ludwig, J.A. Linkages between microbial and hydrologic processes in arid and semiarid watersheds. Ecology 2005, 86, 298–307.

59. Teague, W.R.; Dowhower, S.L.; Baker, S.A.; Haile, N.; DeLaune, P.B.; Conover, D.M. Grazing management impacts on vegetation, soil biota and soil chemical: physical and hydrological properties in tall grass prairie. Agric. Ecosyst. Environ. 2011, 141, 310–322.

60. White, S.R.; J.F. Cahill, Jr.; Bork, E.W. Implications of Precipitation, Warming, and Clipping for Grazing Resources in Canadian Prairies. Agronomy Journal 2014, 106(1), 33-42.

61. Hebb, C.; Schoderbek, D.F.; Hernandez-Ramirez, G.; Hewins, D.B.; Carlyle, C.N.; Bork, E.W. Soil physical quality varies among contrasting land uses in Northern Prairie regions. Agriculture, Ecosystems & Environment 2017, 240, 14-23.

62. Briske, D.D.; Sayre, N.F.; Huntsinger, L.; Fernandez-Gimenez, M.; Budd, B.; Derner, J.D. Origin, persistence, and resolution of the rotational grazing debate: integrating human dimensions into rangeland research. Rangeland Ecol. Manage. 2011, 64, 325–334.

63. Briske, D.D.; Ash, A.J.; Derner, J.D.; Huntsinger, L. Commentary: A critical assessment of the policy endorsement for holistic management, Agricultural Systems 2014, 125,50–53.

64. Teague, W.R. Forages and Pastures Symposium: Cover Crops in Livestock Production: Whole-system Approach: Managing grazing to restore soil health and farm livelihoods. Journal of Animal Science 2018, 96, 1519–1530, https://doi.org/10.1093/jas/skx060

65. Li, W.J.; Ali, S.H.; Zhang, Q. Property rights and grassland degradation: a study of the Xilingol Pasture, Inner Mongolia. China J Environ Manage 2007, 85, 461–470.

66. Suttie, J.M.; Reynolds, S.G.; Batello, C. Grasslands of the world. FAO, Rome, p514. 2005. http://www.fao.org/3/y8344e/y8344e00.htm (accessed on 20 June 2019).

67. Pulina, A.; Lai, R.; Salis, L.; Seddaiu, G.; Roggero, P. P.; Bellocchi, G. Modelling pasture production and soil temperature, water and carbon fluxes in Mediterranean grassland systems with the Pasture Simulation model. Grass and Forage Science 2018, 72(2), 272-283.

68. Brown, L.; Scholefield, D.; Jewkes, E.C.; Lockyer, D.R.; del Prado, A. NGAUGE: a decision support system to optimise N fertilization of British grassland for economic and environmental goals. Agriculture, Ecosystems & Environment 2005, 109, 20-39.

69. Giltrap, D.L.; Ausseil, A.-G.E. Upscaling NZ-DNDC using a regression based meta-model to estimate direct N2O emissions from New Zealand grazed pastures. Science of the Total Environment 2016, 539, 221–230

70. Jenkinson, D.S.; Coleman, K. The turnover of organic carbon in subsoils. Part 2. Modelling carbon turnover. Eur. J. Soil Sci. 2008. 59, 400-413.

71. Abdalla, M.; Jones, M.; Yeluripati, J.; Smith, P.; Burke, J.; Williams, M. Testing DayCent and DNDC model simulations of N_2O fluxes and assessing the impacts of climate change on the gas flux and biomass production from a humid pasture, Atmospheric Environment 2010, 44, 2961-2970.

72. Grant, B.B.; Smith,W.N.; Campbell, C.A.; Desjardins, R.L.; Lemke, R.L.; Kröbel, R.; McConkey, B.G.; Smith, E.G.; Lafond, G.P.; Del Grosso, S.; Ahuja, L.; Parton,W. Comparison of DayCent and DNDC models: case studies using data from long-term experiments on the Canadian prairies. Advances in Agricultural Systems Modeling 6. Synthesis and Modeling of Greenhouse Gas Emissions and Carbon Storage in Agricultural and Forest Systems to Guide Mitigation and Adaptation, pp. 21–58, 2016.

73. Ehrhardt, F.; Soussana, J.-F.; Bellocchi, G.; Grace, P.; McAuliffe, R.; Recous, S.; Sándor, R.; Smith, P.; Snow, V.; Migliorati, M. de A.; Basso, B.; Bhatia, A.; Brilli, L.; Doltra, J.; Dorich, C.D.; Doro, L.; Fitton, N.; Giacomini, S.J.; Grant, B.; Harrison, M.T.; Jones, S.K.; Kirschbaum, M.U.F.; Klumpp, K.; Laville, P.; Léonard, J.; Liebig, M.; Lieffering, M.; Martin, R.; Massad, R.S.; Merbold, E.M.L.; Moore, A.D.; Myrgiotis, V.; Newton, P.; Pattey, E.; Rolinski, S.; Sharp, J.; Smith, W.N.; Wu,; L.; Zhang, Q. Assessing uncertainties in crop and pasture ensemble model simulations of productivity and N2O emissions. Global Change Biology 2018, 24(2), e603-e616.

74. Barrett, P.D.; Laidlaw, A.S.; Mayne, C.S. GrazeGro: a European herbage growth model to predict pasture production in perennial ryegrass swards for decision support. European Journal of Agronomy 2005, 23, 37-56.

75. van Oijen, M.; Bellocchi, G.; Höglind, M. Effects of Climate Change on Grassland Biodiversity and Productivity: The Need for a Diversity of Models. Agronomy 2018, 8, 14.

76. Li, C.; Aber, J.; Stange, F.; Butterbach-Bahl, K.; Papen, H. A process-oriented model of N_2O and NO emissions from forest soils: 1. Model development. J. Geophys. Res. 2000, 105, 4369–4384, https://doi .org/10.1029/1999JD900949.

77. Wang, J.; Cardenas, L.M.; Misselbrook, T.H.; Cuttle, S.; Thorman, R.E.; Li, C. Modelling nitrous oxide emissions in grazed grassland systems, Environ Pollut 2012, 162, 223–233.

78. Yadav, D.; Wang, J. Modelling carbon dioxide emissions from agricultural soils in Canada, Environ Pollut 2017, 230, 1040–1049.

79. Parton, W.J.; Scurlock, J.M.O.; Ojima, D.S.; Schimel, D.S.; Hall, D.O.; Group Members SCOPEGRAM. Impact of climate change on grassland production and soil carbon worldwide. Global Change Biology 1995, 1, 13–22. https://doi.org/10.1111/j.1365-2486.1995.tb00002.x

80. Butterbach-Bahl, K.; Baggs, E.M.; Dannenmann, M.; Kiese, R.; Zechmeister-Boltenstern, S. Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philosophical Transactions of the Royal Society B: Biological Sciences 2013, 368 (1621):20130122

81. Owen, J.J.; Parton, W.J.; Silver, W.L. Long-term impacts of manure amendments on carbon and greenhouse gas dynamics of rangelands. Global Change Biology 2015, 21, 4533-4547. https://doi.org/10.1111/gcb.13044

82. Henderson, B.B.; Gerber, P.J.; Hilinski, T.E.; Falcucci, A.; Ojima, D.S.; Salvatore, M.; Conant, R.T. Greenhouse gas mitigation potential of the world's grazing lands: Modeling soil carbon and nitrogen fluxes of mitigation practices. Agriculture, Ecosystems & Environment 2015, 207, 91-100. https://doi.org/10.1016/j.agee.2015.03.029.

83. Sándor R, Ehrhardt, F., Brilli, L.; Carozzi, M.; Recous, S.; Smith, P.; Snow, V.; Soussana, J.; Dorich, C.; Fuchs, K.; Fitton, N.; Gongadze, K.; et al. The use of biogeochemical models to evaluate mitigation of greenhouse gas emissions from managed grasslands. Science of the Total Environment 2018, 642, 292-306.

84. Shrestha, N.K.; Wang, J. Current and future hot-spots and hot-moments of nitrous oxide emission in a cold climate river basin. Environ. Pollut. 2018, 239, 648-660.

85. Shrestha, N.K.; Thomas, B.W.; Du, X.; Hao, X.; Wang, J. Modeling nitrous oxide emissions from rough fescue grassland soils subjected to long-term grazing of different intensities using the Soil and Water Assessment Tool (SWAT). Environ Sci and Pollut Research 2018, 25 (27), 27362-27377.

86. Riley, W.D.; et al. Small Water Bodies in Great Britain and Ireland: Ecosystem function, human-generated degradation, and options for restorative action. Science of the Total Environment 2018, 645, 1598-1616.

87. Arnold, J.G.; Srinivasan, R.; Muttiah, R.S.; Williams, J.R. Large area hydrologic modeling and assessment part I: model development. J. Am. Water Resour. Assoc. 1998, 34, 73-89.

88. Shrestha, N.K.; Wang, J. Water Quality Management of a Cold Climate Region Watershed in Changing Climate. Journal of Environmental Informatics 2019, DOI: 10.3808/jei.201900407

89. Shrestha, N.K.; Wang, J. Predicting sediment yield and transport dynamics of a cold climate region watershed in changing climate, Science of The Total Environment 2018, 625,1030–1045.

90. Eum, H.-I.; Dibike, Y.; Prowse, T.D. Climate-induced alteration of hydrologic indicators in the Athabasca river basin, Alberta, Canada. J. Hydrol. 2017, 544, 327-342.

91. Dibike, Y.B.; Eum, H.-I.; Prowse, T. Modelling the Athabasca watershed snow response to a changing climate. Journal of Hydrology: Regional Studies 2018, 15, 134-148. DOI: 10.1016/j.ejrh.2018.01.003

92. Chanasyk, D.S.; Mapfumo, E.; Willm, W. Quantification and simulation of surface runoff from fescue grassland watersheds.   Agricultural Water Management 2003, 59(2), 137-153.

93. Dakhlalla, A.O.; Parajuli, P.B. Assessing model parameters sensitivity and uncertainty of streamflow, sediment, and nutrient transport using SWAT. Information Processing in Agriculture 2019, 6, 61-72.

94. Rahbeh, M.; Chanasyk, D.; Miller, J. Modelling the effect of irrigation on the hydrological output from a small prairie watershed, Canadian Water Resources Journal 2013, 38(4), 280-295.

95. Du, X.; Shrestha, N.K.; Ficklin, D.L.; Wang, J. Incorporation of the equilibrium temperature approach in a Soil and Water Assessment Tool hydroclimatological stream temperature model, Hydrology and Earth System Sciences2018, 22, 2343-2357.

96. Du, X.; Shrestha, N.K.; Wang, J. Assessing climate change impacts on stream temperature in the Athabasca River Basin using SWAT equilibrium temperature model

and its potential impacts on stream ecosystem. Science of the Total Environment 2019, 650 (Part 2), 1872-1881.

97. Cui, G.; Wang, J. Improving the DNDC biogeochemistry model to simulate soil temperature and emissions of nitrous oxide and carbon dioxide in cold regions. Science of the Total Environment 2019, 687, 61-70.

98. Melaku, N.D.; Wang, J. A modified SWAT Module for estimating Groundwater table at Lethbridge and Barons, Alberta, Canada. Journal of Hydrology 2019, 575, 420–43

99. Park, J.Y.; Ale, S.; Teague, W.R.; Dowhower, S.L. Simulating hydrologic responses to alternate grazing management practices at the ranch and watershed scales. Journal of Soil and Water Conservation 2017, 72,102-121.

100. Park, J.Y.; Ale, S.; Teague, W.R. Simulated water quality effects of alternate grazing management practices at the ranch and watershed scales. Ecological Modelling 2017, 360, 1-13.

101. Liang, X., Lettenmaier, D.P., Wood, E.F., Burges, S.J., 1994. A simple hydrologically based model of land surface water and energy fluxes for general circulation models. Journal of Geophysical Research: Atmospheres 99, 14415–14428.

102. Liang, X., Wood, E.F., Lettenmaier, D.P. Surface soil moisture parameterization of the VIC-2L model: Evaluation and modification. Global and Planetary Change 1996, 13, 195–206.

103. Cherkauer, K.A., Lettenmaier, D.P. Hydrologic effects of frozen soils in the upper Mississippi River basin. Journal of Geophysical Research: Atmospheres 1999, 104, 19599–19610.

104. Cherkauer, K.A., Bowling, L.C., Lettenmaier, D.P. Variable infiltration capacity cold land process model updates. Global and Planetary Change 2003, 38, 151–159.

105. Adam, J.C., Lettenmaier, D.P. Application of New Precipitation and Reconstructed Streamflow Products to Streamflow Trend Attribution in Northern Eurasia. Journal of Climate 2008, 21, 1807–1828. https://doi.org/10.1175/2007JCLI1535.1

106. Bohn, T.J., Podest, E., Schroeder, R., Pinto, N., McDonald, K.C., Glagolev, M., Filippov, I., Maksyutov, S., Heimann, M., Chen, X., Lettenmaier, D.P. Modeling the large-scale effects of surface moisture heterogeneity on wetland carbon fluxes in the West Siberian Lowland. Biogeosciences 2013, 10, 6559–6576. https://doi.org/10.5194/bg-10-6559-2013.

107. Bowling, L.C., Pomeroy, J.W., Lettenmaier, D.P. Parameterization of blowing-snow sublimation in a macroscale hydrology model. Journal of Hydrometeorology 2004, 5, 745–762.

108. Nijssen, B., Bowling, L.C., Lettenmaier, D.P., Clark, D.B., El Maayar, M., Essery, R., Goers, S., Gusev, Y.M., Habets, F., van den Hurk, B., Jin, J., Kahan, D., Lohmann, D., Ma, X., Mahanama, S., Mocko, D., Nasonova, O., Niu, G.-Y., Samuelsson, P., Shmakin, A.B., Takata, K., Verseghy, D., Viterbo, P., Xia, Y., Xue, Y., Yang, Z.-L. Simulation of high latitude hydrological processes in the Torne–Kalix basin: PILPS Phase 2(e). Global and Planetary Change 2003, 38, 31–53. https://doi.org/10.1016/S0921-8181(03)00004-3

109. Bowling, L.C., Lettenmaier, D.P. Modeling the Effects of Lakes and Wetlands on the Water Balance of Arctic Environments. Journal of Hydrometeorology 2010. 11, 276–295.

110. Shrestha, R.R., Schnorbus, M.A., Werner, A.T., Zwiers, F.W. Evaluating Hydroclimatic Change Signals from Statistically and Dynamically Downscaled GCMs and Hydrologic Models. Journal of Hydrometeorology 2014, 15, 844–860. https://doi.org/10.1175/JHM-D-13-030.1

111. Sheffield, J., Goteti, G., Wen, F., Wood, E.F. A simulated soil moisture based drought analysis for the United States. Journal of Geophysical Research: Atmospheres 2004, 109, D24108, doi:10.1029/2004JD005182.

112. Park, D., Markus, M. Analysis of a changing hydrologic flood regime using the Variable Infiltration Capacity model. Journal of Hydrology 2014, 515, 267–280. https://doi.org/http://dx.doi.org/10.1016/j.jhydrol.2014.05.004

113. Schumann, G.J.-P.; Neal, J.C.; Voisin, N.; Andreadis, K.M.; Pappenberger, F.; Phanthuwongpakdee, N.; Hall, A.C.; Bates, P.D. A first large-scale flood inundation forecasting model. Water Resources Research 2013, 49, 6248–6257. https://doi.org/doi:10.1002/wrcr.20521

114. Fung, I.; John, J.; Lerner, J.; Matthews, E.; Prather, M.; Steele, L.P.; Fraser, P.J. Three‐dimensional model synthesis of the global methane cycle. Journal of Geophysical Research: Atmospheres 1991, 96, 13033–13065.

115. Ringeval, B.; Friedlingstein, P.; Koven, C.; Ciais, P.; de Noblet-Ducoudré, N.; Decharme, B.; Cadule, P. Climate-CH_4 feedback from wetlands and its interaction with the climate-CO_2 feedback. Biogeosciences 2010, 8, 2137–2157.

116. Walter, B.P.; Heimann, M.; Matthews, E. Modeling modern methane emissions from natural wetlands: 1. Model description and results. Journal of Geophysical Research: Atmospheres 2001, 106, 34189–34206.

117. Qi, A.; Murray, P.J.; Richter, G.M. Modelling productivity and resource use efficiency for grassland ecosystems in the UK. Eur. J. Agron. 2017, 89, 148–158.

118. Qi, A.; Holland, R.A.; Taylor, G.; Richter, G.M. Grassland futures in Great Britain – Productivity assessment and scenarios for land use change opportunities. Science of the Total Environment 2018, 634, 1108-1118.

119. Kayser, M.; Müller, J.; Isselstein, J. Grassland renovation has important consequences for C and N cycling and losses. Food and Energy Security 2018, 7(4), e00146.

120. Luo, Y., et al. Toward more realistic projections of soil carbon dynamics by Earth system models, Global Biogeochem. Cycles 2016, 30, 40–56, doi:10.1002/2015GB005239.

121. Dangal, S.R.S.; Tian, H.; Xu, R.; Chang, J.; Canadell, J. G.; Ciais, P., et al. Global nitrous oxide emissions from pasturelands and rangelands: Magnitude, spatiotemporal patterns, and attribution. Global Biogeochemical Cycles 2019, 33. https://doi.org/10.1029/2018GB006091

122. Kipling, R.P.; et al. Key challenges and priorities for modelling European grasslands under climate change. Science of the Total Environment 2016, 566–567, 851–864.

123. Brilli, L.; Bechini, L., Bindi, M.; Carozzi, M.; Cavalli, D.; Conant, R.; Dorich, C.D.; Doro, L.; Ehrhardt, F.; Farina, R.; Ferrise, R.; Fitton, N.; Francaviglia, R.; Grace, P.; Iocola, I.; Klumpp, K.; Léonard, J.; Martin, R.; Massad, R.S.; Recous, S.; Seddaiu, G.; Sharp, J.; Smith, P.; Smith, W.N.; Soussana, J.-F.; Bellocchi, G. Review and analysis of strengths and weaknesses of agro-ecosystem models for simulating C and N fluxes. Sci. Total Environ. 2017, 598, 445–470.

124. Bhanja, S.N.; Wang, J.; Shrestha, N.K.; Zhang, X. Microbial kinetics and thermodynamic (MKT) processes for soil organic matter decomposition and dynamic oxidation-reduction potential: Model descriptions and applications to soil N_2O emissions, Environmental Pollution 2019a, 247, 812-823.

125. Bhanja, S.N.; Wang, J.; Shrestha, N.K.; Zhang, X. Modelling microbial kinetics and thermodynamic processes for quantifying soil CO_2 emission. Atmospheric Environment 2019b, 209, 125–135.

126. Lal, R. Soil carbon sequestration to mitigate climate change, Geoderma 2004, 123, 1 – 22.

127. Tagesson, T.; Fensholt, R.; Cropley, F. et al. Dynamics in carbon exchange fluxes for a grazed semi-arid savanna ecosystem in West Africa. Agriculture Ecosystems and Environment 2015, 205, 15–24.

128. Shen, J.; Treu, R.; Wang, J.; Thorman, R.; Nicholson, F.; Bhogal, J. Modelling of nitrous oxide emissions from three United Kingdom farms following application of farmyard manure and green compost, Sci Total Environ. 2018a, 637–638, 1566–1577.

129. Shen, J.; Treu, R.; Wang, J.; Nicholson, F.; Bhogal, J.; Thorman, R. Modeling Nitrous Oxide from Digestate and Slurry Applied to Three Agricultural Soils in the United Kingdom: Fluxes and Emission Factors. Environmental Pollution 2018b, 243 (Part B): 1952-1965.

130. Davidson, E.A.; Janssens, I.A. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature 2006, 440, 165-173.

131. Davidson, E.A.; Samanta, S.; Caramori, S.S.; Savage, K. The Dual Arrhenius and MichaeliseM enten kinetics model for decomposition of soil organic matter at hourly to seasonal time scales. Glob. Chang. Biol. 2012, 18, 371-384.

132. Davidson, E. A.; Savage, K.E.; Finzi, A.C. A big-microsite framework for soil carbon modeling, Global Change Biol. 2014, 20(12), 3610–3620, doi:10.1111/gcb.12718.

133. Groffman, P.M.; Butterbach-Bahl, K.; Fulweiler, R.W.; Gold, A.J.; Morse, J.L.; Stander, E.K.; Tague, C.; Tonitto, C.; Vidon, P. Challenges to incorporating spatially and temporally explicit phenomena (hotspots and hot moments) in denitrification models. Biogeochemistry 2009, 93, 49-77.

134. Cui, G.; Wang, J. Improving the DNDC biogeochemistry model to simulate soil temperature and emissions of nitrous oxide and carbon dioxide in cold regions. Science of The Total Environment 2019, 687, 61-70.

135. Dutta, B.; Grant, B.B.; Campbell, C.A.; Lemke, R.L.; Desjardins, R.L.; Smith, W.N. A multi model evaluation of long-term effects of crop management and cropping systems on nitrogen dynamics in the Canadian semi-arid prairie. Agric. Syst. 2017, 151, 136–147.

136. Aschenbrenner, F.; Richter, G.M.; Richter, J. Modeling Groundwater Quality in an Agriculturally Used Water Catchment. Environ Geol Water Science 1992, 20, 43-55.

137. Kipling, R.P.; et al. Key challenges and priorities for modelling European grasslands under climate change. Science of the Total Environment 2016, 566–567, 851–864.