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Greenhouse gas balance in global pasturelands and rangelands

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Abstract

Grassland ecosystems play an essential role in climate regulation through carbon (C) storage in plant and soil. But, anthropogenic practices such as livestock grazing, grazing related excreta nitrogen (N) deposition, and manure/fertilizer N application have the potential to reduce the effectiveness of grassland C sink through increased nitrous oxide (N₂O) and methane (CH₄) emissions. Although the effect of anthropogenic activities on net greenhouse gas (GHG) fluxes in grassland ecosystems have been investigated at local to regional scales, estimates of net GHG balance at the global scale remains uncertain. With the data-model framework integrating empirical estimates of livestock CH₄ emissions with process-based modeling estimates of land CO₂, N₂O and CH₄ fluxes, we examined the overall global warming potential (GWP) of grassland ecosystems during 1961–2010. We then quantified the grassland-specific and regional variations to identify hotspots of GHG fluxes. Our results show that, over a 100-year time horizon, grassland ecosystems sequestered a cumulative total of 113.9 Pg CO₂-equivalent (CO₂-eq) in plant and soil, but then released 91.9 Pg CO₂-eq to the atmosphere, offsetting 81% of the net CO₂ sink. We also found large grassland-specific variations in net GHG fluxes, with pasturelands acting as a small GHG source of 1.52 ± 143 Tg CO₂-eq yr⁻¹ (mean ± 1.0 s.d.) and rangelands a strong GHG sink (−442 ± 266 Tg CO₂-eq yr⁻¹) during 1961–2010. Regionally, Europe acted as a GHG source of 23 ± 10 Tg CO₂-eq yr⁻¹, while other regions (i.e. Africa, Southern Asia) were strong GHG sinks during 2001–2010. Our study highlights the importance of considering regional and grassland-specific differences in GHG fluxes for guiding future management and climate mitigation strategies in global grasslands.

1. Introduction

Rising concentrations of greenhouse gases (GHGs) such as carbon dioxide (CO₂), nitrous oxide (N₂O) and methane (CH₄) in the atmosphere have immediate and long lasting effect on the global climate system through increased radiative forcing (Ciais et al. 2014, Tian et al. 2016b). As an effective GHGs mitigation strategy, grassland ecosystems offer long-term climate benefits because they are increasingly resilient to climate extremes (Hoover et al. 2014), and are more reliable C sinks compared to forests (Dass et al. 2018). Management practices such as irrigation and fertilization can further mitigate atmospheric CO₂ accumulation by conserving and sequestering C (Ryals and Silver 2013, Smith 2014, Conant et al. 2017). But, such practices have the potential to stimulate N₂O and CH₄ emissions (Soussana et al. 2007), and negate any C gains due to increased productivity and soil organic matter retention (Leahy et al. 2004). While much efforts have been made to assess the GHG balance of grassland ecosystems at site and regional/continental scales (Leahy et al. 2004, Jones et al. 2005, Soussana et al. 2010, Chang et al. 2015, Hörltnagl et al. 2018), reliable estimates of net GHG balance at the global scale are lacking.

For intensively managed (pasturelands) and extensively managed (rangelands) grasslands, the nature, frequency and intensity of management practices play an essential role in determining the net GHG balance. Although management practices such as irrigation and fertilization can become a
viable option for enhancing grassland growth and CO₂ sequestration, such benefits are counterbalanced when the grazing pressure are close to or exceed the carrying capacity of grasslands (Hilker et al 2014, Dangal et al 2016). Nitrite oxide fluxes, on the other hand, is strongly linked to nitrogen (N) availability and controlled by various factors including soil water content, temperature, pH and substrate (C and N) availability. For grasslands, additional N input originated from livestock excreta (urine and feces) results in soil N₂O emission peaks, predominantly driven by microbial processes of nitrification (oxidation of ammonium to nitrate) and denitrification (reduction of nitrate to molecular N) (Dangal et al 2019). The magnitude of N₂O emissions also depend on the type, amount and timing of applied N as well as the legacy effect of previous N accumulation (Abraha et al 2018). Likewise, grasslands can be a small CH₄ sink (Davidson 1991, Mosier et al 1997) or small to moderate CH₄ sources (Chan and Parkin 2001, Flessa et al 2002), depending on the factor that regulates soil microbial activity responsible for CH₄ production and consumption. In many grassland ecosystems, the magnitude of CH₄ fluxes from soil is relatively small compared to fluxes from grazing animals due to fermentation of feed reserves, which varies as a function of livestock type, forage and digestibility (Allard et al 2007). Manure management practices can further affect CH₄ fluxes with lagoon and liquid management yielding higher emissions due to anaerobic conditions that promote methanogens (CH₄ producing microbes) activity compared to dry manure management (Dangal et al 2017). Studies focused on assessing the net global warming potential (GWP) of GHGs at site and regional/continental scales have shown that grasslands can be a net GHG source or a small sink (Liebig et al 2010, Chadwick et al 2011, Chang et al 2015). However, to date, there is limited understanding about how grassland management alters the fluxes of CO₂, N₂O and CH₄ at the global scale. As the capacity of grassland ecosystems to act as a net GHG sink or source depend on the relative effects of multiple environmental factors including land management practices and their potential interaction with both plant physiological and soil microbial processes, an integrated assessment of all GHG fluxes is essential to better estimate the overall GHG balance of grassland ecosystems.

With the data-model framework integrating empirical estimates of animal-based CH₄ emissions with process-based modeling estimates of CO₂, N₂O and CH₄ fluxes from the land surface (Schulze et al 2009, Tian et al 2015), we examined the overall GWP of grassland ecosystems during 1961–2010. We simulated grassland CO₂, N₂O and CH₄ fluxes using a highly integrated process-based model (the Dynamic Land Ecosystem Model; DLEM) driven by changes in climate, land cover and land use change (LCLUC), atmospheric chemistry, N deposition and land management practices (i.e. fertilizer and manure N application and excreta N deposition) (Tian et al 2015, 2019). We estimated livestock related CH₄ emissions from enteric fermentation and manure management at the global level and extracted all grid cells that represented intensively managed pasturelands and extensively managed rangelands based on Historical Database of Global Environment (HYDE v3.2). We then determined the net GWP from grassland ecosystems over a 100-year time horizon (Tian et al 2016b) followed by attribution of changes in net GHG fluxes by grassland types (pasturelands and rangelands). Finally, we assess the variations in net GHG balance by dividing the global land area into seven regions.

2. Materials and methods

2.1. Enteric fermentation related CH₄ emissions

To quantify CH₄ emissions from enteric fermentation, we used IPCC Tier II approach, which relies on the estimation of net energy requirement and energy availability of the diet (Dong et al 2006). This includes energy for maintenance, work, activity, lactation, pregnancy, growth and wool production (for sheep only) (text S1 (available online at https://stacks.iop.org/ERL/15/104006/mmedia)). The most important parameters that affect the growth and maintenance energy requirements of each livestock category are the live weight and milk production potential of the animals. Energy requirements vary significantly for different categories of livestock or for different live weight even within the same category (Dangal et al 2017). As a result, we calculated the energy requirements using equations 10.3–10.9 and 10.11–10.16 (Dong et al 2006) for different livestock categories as described in supporting information (SI) text S1.

2.2. Manure management related CH₄ emissions

To estimate CH₄ emissions from manure management, we used IPCC tier II guidelines that vary as a function of manure characteristics and manure management systems (Dong et al 2006). Manure characteristics refers to total volatile and non-volatile solids produced in manure, while manure management includes solid vs liquid manure handling systems, where a system specific conversion factor is used to estimate CH₄ production. Additionally, CH₄ emission from manure management is modified by local climatic conditions. Therefore, we combined CRUNCEP global gridded mean annual temperature at 0.5° × 0.5° spatial resolution and equation 10.23 (Dong et al 2006) to estimate CH₄ emission from manure management as described in SI text S2.
2.3. Global grassland GHG simulations using DLEM

Gridded, georeferenced datasets for DLEM were compiled from various sources at a spatial resolution of 0.5° × 0.5° (SI table S1). These datasets include daily climate data, atmospheric chemistry (CO$_2$ concentration, AOT40 O$_3$ index, and N deposition), soil properties, land management practices (Excreta N deposition, manure N application and fertilizer N application) and other ancillary data such as river network, cropping system and topography maps. Daily climate data during 1901–2010 were based on CRU-NCEP climate forcing, while atmospheric CO$_2$ concentration was obtained from Carbon Dioxide Information Analysis Center CDIAC. Annual N deposition was retrieved from the outputs of multiple atmospheric chemistry transport models (Dentener et al. 2006). The effects of tropospheric ozone concentration on grassland dynamics was developed using an AOT40 index (Felzer et al. 2004), while elevation, slope and aspect were derived from Global 30 arc-second elevation product GTOPO30. Soil texture was derived from Food and Agricultural Organization (FAO) Soil Database System (Reynolds et al. 2000). Excreta N deposition, manure N application and fertilizer N application in grassland ecosystems are consistent with the Nitrogen Model Intercomparison Project (Tian et al. 2018, Xu et al. 2019). In addition, we used HYDE 3.2 (Klein Goldewijk et al. 2017) grassland datasets to create maps of global pasturelands and rangelands distribution during 1961–2010 (text S3).

For each grid cell, we first ran the DLEM to determine the equilibrium state of C, N and water using the early 20th century (30 year; 1901–1930) daily climate, while other input data (atmospheric CO$_2$, N deposition and land cover) were kept at 1900 level. The equilibrium run is carried out for the maximum of 10,000 years or until the net carbon exchange between the atmosphere and the terrestrial ecosystems is less than 0.5 g C m$^{-2}$ yr$^{-1}$, the change in soil water pool is less than 0.5 mm, and the change in total N content is less than 0.5 g N m$^{-2}$ during two consecutive 20 years. We then carried out a model spin up for 100 years by randomly selecting 20 years of climate data, repeated five times, during 1901–1930. The purpose of model spin up is to account for the influence of inter-annual variability on the initial conditions of C, N and water pools and to smooth the transition from equilibrium state to transient run.

2.4. Estimation of the full GHG budget in grasslands

We estimated net CO$_2$, N$_2$O and CH$_4$ fluxes using their respective GWP. The GWP is a metric developed to compare the global warming impacts of multiple GHGs on the planetary energy budget over a certain period (20 or 100 years). In this study, we adopted a 100-year GWP of 265 and 28 for converting N$_2$O and CH$_4$ to CO$_2$ equivalents, respectively. The net GHG balance using a 100-year GWP is obtained as:

$$\text{GHG} = \left(\text{NEE} \times \frac{44}{12}\right) + \left(N_2O \times \frac{44}{28} \times 265\right) + \left(CH_4 \times \frac{16}{12} \times 28\right)$$

NEE is the net ecosystem exchange and is estimated as the difference between C gains through gross primary production and loss through plant and heterotrophic respiration based on the model simulation. A negative value of NEE indicates a net sink of CO$_2$ to soils while a positive value indicates a net source of CO$_2$ to the atmosphere. A similar term net ecosystem production (NEP) is commonly used in literature, where a positive NEP (negative NEE) is a CO$_2$ sink to the soils and a negative NEP (positive NEE) is a CO$_2$ source to the atmosphere, largely dependent on whether the atmosphere or ecosystem is used as the reference (Luyssaert et al. 2009). N$_2$O and CH$_4$ are the net source/sink of nitrous oxide and methane, respectively. A positive value of N$_2$O or CH$_4$ indicates that land is a source of N$_2$O or CH$_4$ to the atmosphere, while a negative value indicates that land is sink. N$_2$O source/sink from soils due to nitrification and denitrification is estimated using the DLEM simulation. Natural CH$_4$ source/sink associated with methanotrophs (CH$_4$ consuming) and methanogens (CH$_4$ producing) is also based on DLEM simulations, while CH$_4$ source due to enteric fermentation and manure management is based on the IPCC tier II approach. Details on the estimation of NEE, CH$_4$ and N$_2$O fluxes from soils using the DLEM can be found in SI text S4, and comparison of model simulated CO$_2$, CH$_4$ and N$_2$O fluxes against observations and other data products are available in SI text S5–S6, figure S1, tables S2–S7. The constants 44/12, 44/28 and 16/12 are used to convert the mass of CO$_2$, N$_2$O-N and CH$_4$-C to CO$_2$, N$_2$O and CH$_4$, respectively.

2.5. Statistical analysis and uncertainty estimates

We used mean and 1-sigma standard deviation to provide estimates of CO$_2$, N$_2$O and CH$_4$ fluxes and their uncertainty during the study period. The statistically significant trend of GHG fluxes was detected using non-parametric Mann-Kendall method (Mann 1945). In addition, we estimated the uncertainty in N$_2$O and CO$_2$ fluxes associated with N input data by first determining the temporal change in total N input. We then assessed the relationship between these GHG fluxes and N input using bootstrapping method with 10,000 replicates to determine the 95% confidence interval of the slope of that relationship. Uncertainty range of N input data was then multiplied by the slope of the relationship between GHG fluxes and N input to estimate uncertainty in CO$_2$ and
N₂O fluxes due to changes in N input data. The bootstrapping technique uses a random sampling method with replacement, and was implemented using two functions (boot and boot.ci) in R (Canty 2002).

3. Results

3.1. Net GHG fluxes from grassland ecosystems

From 1961 to 2010, grassland ecosystems sequestered a net cumulative total of $-22.0 \text{ Pg CO}_2$-eq (1 Pg = $10^{15}$ g) due to multiple environmental changes and land management practices over a 100-year time horizon. Net cumulative C uptake (NEE) was $-113.9 \text{ Pg CO}_2$-eq, while C release in the form of N₂O and CH₄ were 49.5 Pg CO₂-eq and 42.4 Pg CO₂-eq, respectively. Over the study period, grasslands were a net CO₂ sink of $-2.3 \pm 0.7 \text{ Pg CO}_2$-eq yr⁻¹, while N₂O and CH₄ sources of $1.0 \pm 0.2 \text{ Pg CO}_2$-eq yr⁻¹ and $0.85 \pm 0.2 \text{ Pg CO}_2$-eq yr⁻¹, respectively (figure 1(a); SI table S8). Overall, N₂O and CH₄ were responsible for offsetting 44% and 37% of the net CO₂ sink, respectively. When N₂O and CH₄ are combined together, the grassland ecosystem acted as a small C sink of $-1.2 \pm 0.5 \text{ Pg CO}_2$-eq yr⁻¹, largely because N₂O and CH₄ emissions were responsible for offsetting 81% of the net CO₂ sink.

Model results also show annual and decadal variations in GHG fluxes during the study period (figure 1(b)). Grassland CO₂ sink increased significantly at the rate of $13.3 \text{ Tg CO}_2$-C yr⁻² (1 Tg = $10^{12}$ grams) (R² = 0.99; p < 0.01). Likewise, N₂O and CH₄ fluxes to the atmosphere increased significantly at the rate of $0.03 \text{ Tg N}_2\text{O}-\text{N yr}^{-2}$ (R² = 0.99; p < 0.01) and $0.28 \text{ Tg CH}_4$-C yr⁻² (R² = 0.98; p < 0.01), respectively. Compared to the 1960s, the net cooling effect due to grassland CO₂ sink increased by 157% in the 2000s, while the net warming effect due to N₂O and CH₄ release to the atmosphere increased by 57% and 66%, respectively.

3.2. Contribution of pasturelands and rangelands to net GHGs

Carbon dioxide, CH₄ and N₂O emissions show large variations in terms of their contribution to net GHG fluxes in pasturelands and rangelands (figure 2). N₂O and CH₄ fluxes to the atmosphere were responsible for offsetting 100% of the CO₂ sink resulting in a net GHG source of $1.52 \pm 143 \text{ Tg CO}_2$-eq yr⁻¹ in pasturelands (figure 2(a), 2(b)). The net GHG source in pasturelands was largely dominated by N₂O emissions, which contributed to 72% ($0.86 \pm 0.16 \text{ Pg CO}_2$-eq yr⁻¹) of the net fluxes to the atmosphere during 1961–2010. However, in rangelands, CH₄ emissions was the major driver of net GHG sources, contributing to 79% ($0.51 \pm 0.09 \text{ Pg CO}_2$-eq yr⁻¹) of the net GHG fluxes to the atmosphere (figure 2(c), 2(d)). Overall, N₂O and CH₄ emissions were responsible for offsetting 59% ($0.65 \pm 0.1 \text{ Pg CO}_2$ yr⁻¹) of the net CO₂ sink in rangelands.

3.3. Spatial and regional variations in net GHGs

Although grassland ecosystems acted as a net GHG sink of $-1.20 \text{ Pg CO}_2$-eq yr⁻¹ ($-0.33 \text{ Pg CO}_2$-C yr⁻¹), model results show large variations in the spatial pattern of net GHG balance, with some areas showing a clear net GHG sink and others a net GHG source (figures 3(a) and (b)). Regional variations in GHG indicated that rangelands, in particular, were a strong GHG sink in Africa and Southern Asia. Interestingly, model results show that pasturelands were a net GHG source in Europe and GHG neutral to small sink in other regions (figure 3(b)). Europe contributed to the GHG source of $23 \pm 10 \text{ Tg CO}_2$-eq yr⁻¹ during 2001–2010, which was largely dominated by N₂O fluxes. Likewise, Southern Asia was a small sink of $49.7 \text{ Tg CO}_2$-eq yr⁻¹ largely because of a recent (since 2001) increase in N₂O flux, which offsets 75% of the CO₂ sink. Overall, the regional GHG sink/source was primarily driven by a recent increase in N₂O emissions, associated with high N inputs in the form of excreta, and manure and fertilizer N application.

4. Discussion

4.1. Net GHG balance in grassland ecosystems

Our results indicate that management intensity (i.e. excreta N deposition, manure/fertilizer N application and manure N management practices) and increased livestock density have dramatically altered the GHG balance of grassland ecosystems at the global scale. Grazing and management intensity (excreta, manure and fertilizer N) induced N₂O and CH₄ emissions were responsible for offsetting nearly 81% of the net CO₂ sink during 1961–2010 (figure 1). Extensively managed rangelands acted as a strong GHG sink ($-442 \pm 266 \text{ Tg CO}_2$-eq yr⁻¹), while intensively managed pasturelands were a small source of $1.5 \pm 143 \text{ Tg CO}_2$-eq yr⁻¹ (figure 2). These findings are consistent with recent studies of grassland GHG fluxes at site (Soussana et al 2010, Merbold et al 2014) and regional (Schulze et al 2009, Chang et al 2015) levels, which shows that the GHG sink strength is reduced following intensive grassland management. Using eddy covariance (EC) flux measurements at the ecosystem level, Merbold et al (2014) found a shift from a C sink to a significant C source following grassland restoration practices such as ploughing and N fertilization. Continental scale studies from Europe point toward mixed results with a net GHG source at the farm level due to additional losses of C from the transport of harvested biomass (Chang et al 2015), while a net GHG sink at the ecosystem level (Schulze et al 2009, Chang et al 2015). Compared to Chang et al (2015) GHG sink estimates of $-19 \pm 10 \text{ g CO}_2$-eq m⁻² yr⁻¹ using a process based model, we estimated a net sink of $-9.2 \pm 14 \text{ g CO}_2$-eq m⁻² yr⁻¹ during 1961–2010. On the other hand, Schulze et al (2009) used top-down and bottom-up approaches to estimate a grassland C sink of $-85 \pm 12 \text{ Tg C yr}^{-1}$,
which is comparable to our estimate of $-95 \pm 3$ Tg C yr$^{-1}$ during 2000–2005.

Our results also demonstrate large spatial and grassland-specific variations in net GHG balance due to unprecedented changes in management intensity and grazing pressure (SI figures S2–S6). Spatial variations in the trend of CO$_2$, N$_2$O and CH$_4$ fluxes point toward a declining trend in C uptake and CH$_4$ release in Europe (SI figure S7). Despite a decline in C uptake and CH$_4$ release due to reduction in grazing pressure and excretion rates, N$_2$O fluxes still show a significant positive trend because of an increase in synthetic fertilizer and manure application rates (Xu et al 2019). Additionally, Southern Asia has experienced the largest magnitude of increase in GHG sink since 1961 (table 1). These changes are largely attributed to a strong C uptake particularly in eastern China and India due to N enrichment (SI figure S5), which stimulates plant growth and biomass production (Xia and Wan 2008). In the northern China, Niu et al (2008) found a significant increase in growing-season C uptake (NEE) by up to 60%, but the increase declined over time due to N-induced shift in species composition. In a similar study with different levels of N addition, Bai et al (2009) show a saturating effect of N addition on biodiversity and ecosystem function at rates of approximately 10.5 gN m$^{-2}$ yr$^{-1}$. Average N inputs based on our study is 2.4 gN m$^{-2}$ yr$^{-1}$ in Southern Asia, which is below the threshold value recognized by Bai et al (2009). We found that average annual C uptake (NEE) has increased at a rate of 22% since 2000.

### 4.2. Carbon sequestration and net GHG balance

Most studies, to date, suggest that grasslands are a net sink of CO$_2$, but the mechanism driving this sink is not well understood. While some studies show that grasslands are a natural C sink (Soussana et al 2007, Piao et al 2007, Fay et al 2015), others indicate the important role of the legacy effect of past land use and management on grassland C sink (Smith 2014). The sink strength of grassland ecosystems also depend on other environmental factors such as climate change, elevated atmospheric CO$_2$ concentration and the timing and magnitude of grassland management activities (Pepper et al 2005, Reich et al 2006, Felzer et al 2011). Our study indicates that grassland ecosystems are sequestering C at a rate of 0.88 Pg C yr$^{-1}$, contributing to 28% of the residual C sink (3.1 Pg C yr$^{-1}$) from the terrestrial biosphere (Le Quéré et al 2016). Statistical analysis further revealed that higher N inputs in the form of excreta, manure and fertilizer were the dominant drivers of net C uptake (SI figures S8a–S8d) in both pasturelands and rangelands (SI figures S8e and S8f). Carbon sequestration potential of grasslands ecosystems are increasingly limited by N input, and that soil N availability can further constrain the C...

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**Table 1.** Net GHG fluxes across different regions between the 1960s and the 2000s. The error bar represents the standard deviation of the sum of all the fluxes for that decade. The positive values means that land is a net GHG source and vice versa.

| Region    | Pasturelands | Rangelands |
|-----------|--------------|------------|
|           | 1960s (Tg CO$_2$-eq yr$^{-1}$) | 2000s (Tg CO$_2$-eq yr$^{-1}$) | 1960s (Tg CO$_2$-eq yr$^{-1}$) | 2000s (Tg CO$_2$-eq yr$^{-1}$) |
| Africa    | 38 ± 7       | -55 ± 10   | -41 ± 19   | -239 ± 13  |
| Europe    | 7 ± 16       | 48 ± 9     | -2 ± 1     | -23 ± 2    |
| N. America| 36 ± 8       | -35 ± 12   | 11 ± 5     | -2 ± 5     |
| N. Asia   | -2 ± 1       | -3 ± 1     | -1 ± 0     | -9 ± 1     |
| Oceanica  | 2 ± 0        | 2 ± 0      | -4 ± 7     | -62 ± 2    |
| S. America| 15 ± 1       | -29 ± 4    | 24 ± 5     | -49 ± 11   |
| S. Asia   | 151 ± 15     | -51 ± 15   | -85 ± 19   | -470 ± 19  |

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![Figure 1](image_url). Net greenhouse gas balance in grassland ecosystems (a) and their decadal changes (b) during 1961–2010. The error bars represent the standard deviation of fluxes during the study period (a) and each decade (b). The solid black symbols in (b) represent the net GHG balance for each decade calculated as a sum of individual fluxes.
sink under elevated atmospheric CO₂ concentration (Reich et al 2006). A recent global meta-analysis has shown an increase in grassland productivity by up to 79% following N addition (You et al 2017), indicating an increased sensitivity of grassland ecosystems to N availability (SI figure S8). However, prolonged inputs of N lead to N saturation (Tian et al 2016a) and C sequestration potential may level off (Pepper et al 2005) resulting in no additional C gains (Fisher et al 2007). To examine the relative effects of N addition on NEE, we analyzed nitrogen use efficiency (NUE) measured as a ratio of NEE to N input for both pasturelands and rangelands. Our analysis indicates that NUE has increased significantly in grassland ecosystems (SI figure S9a), with a higher rate of increase in rangelands (0.13 gC gN⁻¹ yr⁻¹) compared to pasturelands (0.06 gC Tg N⁻¹ yr⁻¹). The strong N limitation of rangelands agrees with other studies (Fay et al 2015, You et al 2017), and that stronger response of NEE to N addition can be attributed to the interaction of non-N limiting conditions (higher excreta N) and elevated atmospheric CO₂ concentration (Reich et al 2006). While a strong NEE response was also prevalent in pasturelands, our results show...
that the sustained increase in NEE levels off after 2000, indicating a diminishing effect of N enrichment on plant productivity (Tian et al 2016a).

The potential of grassland ecosystems to sequester C also depends on climate and its interaction with other environmental factors (Parton et al 1995, Heimann and Reichstein 2008). As expected, wetter and warmer climatic conditions and its interaction with management intensity led to a significant increase in NEE at the rate of 6.6 Tg C yr$^{-2}$ ($R^2 = 0.79$; $p < 0.05$) in pasturelands, largely because higher water availability and non-N limiting conditions alleviated both soil moisture and nutrient constraints to plant growth and productivity (Niu et al 2008). However, despite the climatic constraints (drier conditions) that should lead to a decline in CO$_2$ sequestration (Liu et al 2009, Rajan et al 2013), model simulations show a significant increase in NEE at the rate of 6.8 Tg C yr$^{-2}$ ($R^2 = 0.20$; $p < 0.05$) in rangelands. Rangelands cover approximately 77% of the global grassland area resulting in higher total C uptake (NEE) compared to pasturelands. Analysis of the cumulative C uptake per unit area revealed a sequestration rate of $−2.18$ g C m$^{-2}$ and $−0.60$ g C m$^{-2}$ for pasturelands and rangelands, respectively during 1961–2010. To further examine the effect of moisture limitations on NEE between pasturelands and rangelands, we estimated precipitation use efficiency (PUE) as a ratio of NEE to annual precipitation totals (SI figure S9b). Our analysis shows that PUE increased significantly in both pasturelands and rangelands. Although the rate of increase in PUE was similar for both grasslands, the magnitude of PUE was 72% higher in pasturelands (0.042 gC/kgH$_2$O) compared to rangelands (0.024 gC/kgH$_2$O). This is because PUE is strongly dependent on the interaction of soil moisture availability with N limiting conditions. For example, higher N inputs per unit area in pasturelands alleviated N limiting conditions allowing vegetation to utilize water more efficiently (Bai et al 2008) compared to rangelands that are still experiencing N limitation. Although PUE and NUE explains the inherent differences in the magnitude and strength of NEE between rangelands and pasturelands, other factors such as the size, timing and lag effect of precipitation (Huxman et al 2004, Schwinning et al 2004, Heisler-White et al 2008, Thomey et al 2011) can alter grassland productivity, and NEE (Huxman et al 2004). Here, we demonstrate that the effects of climate on NEE is modulated by its interaction with management factors (grazing and/or N enrichment) and the intensity of grassland management.

4.3. Nitrous oxide fluxes and net GHG balance

Our results also suggest that N$_2$O emission was the major driver of the GHG balance in grassland ecosystems, with large differences in the magnitude of emissions between rangelands and pasturelands (figure 2). The differences in N$_2$O fluxes between pasturelands and rangelands can be explained by two possible reasons: (a) higher management intensity in pasturelands due to additional N inputs in the form of manure/fertilizer N; and (b) warmer and wetter climatic conditions in pasturelands indirectly modify soil water availability, thereby increasing N$_2$O fluxes. Annual N$_2$O emission was several orders of magnitude higher in pasturelands compared to rangelands due to higher management intensity, lower N limitation and smaller plant growth responses, which led to the retention of larger proportion of applied N in soils intensifying nitrification and denitrification rates (Hartmann and Niklaus 2012). Sensitivity analysis of the effect of N inputs show that N$_2$O fluxes from soils increased significantly due to an increase in excreta deposition, and manure and fertilizer application (SI figure S10).

In addition to management intensity, air temperature and precipitation changes can affect N$_2$O emissions by indirectly modifying nitrification and denitrification rates, often in a non-linear way (Davidson 2009, Shcherbak et al 2014). To understand the mechanism of higher N$_2$O emissions in pasturelands compared to rangelands, we further examined the relative role of air temperature and precipitation changes on N$_2$O emissions. Our results show that both pasturelands and rangelands did not experience substantial changes in precipitation during the study period, despite higher magnitude of annual precipitation totals in pasturelands (1043 mm yr$^{-1}$) compared to rangelands (490 mm yr$^{-1}$). In contrast, air temperature in pasturelands increased significantly at the rate of 0.04 °C yr$^{-1}$, contributing to larger magnitude of N$_2$O fluxes compared to rangelands. Given that pasturelands are less water limited compared to rangelands (Klein Goldewijk et al 2017), an increase in temperature enhances microbial activity (Scanlon and Kiely 2003), and possibly intensifies denitrification rates due to anaerobic condition as a result of high precipitation. Our findings are consistent with other studies indicating that N$_2$O flux are higher when air temperature is between 5 °C and 18 °C (Saggar et al 2004, Flechard et al 2007, Rafique et al 2011).

Likewise, precipitation can influence N$_2$O fluxes in two possible ways: (a) high rainfall events can increase water filled pore space (WFPS), thereby increasing N$_2$O fluxes (Dobie and Smith 2001, Bateman and Baggs 2005); and (b) higher rainfall also increases nitrate leaching, which ultimately leads to a reduction in the concentration of nitrates necessary for denitrification (Saggar et al 2007). A previous study show that simulated drought resulted in a larger and sustained reduction in N$_2$O emissions compared to non-drought conditions (Hartmann and Niklaus 2012), but such effects were counterbalanced by high N$_2$O emissions at moderate to high soil moisture. This implies that despite high N inputs in the form of excreta in rangelands, warmer temperatures and low mean annual precipitation possibly triggered drought...
conditions resulting in low N₂O fluxes. In contrast, warmer and wetter climatic conditions resulted in continuous and sustained increase in N₂O fluxes in pasturelands.

The effects of management intensity on N₂O emissions are well evident across site to regional scale studies (Soussana et al 2007, Schulze et al 2009, Chang et al 2016), but spatio-temporal variations in management intensity and their impact on N₂O emissions have not been examined in previous studies (Tian et al 2019). Our results show that N₂O emission was the major regulator of GHG balance in Europe, North America and Southern Asia (figure 3(c), SI figure S7), where increased N inputs resulted in an emission of 302 Tg CO₂-eq yr⁻¹, 190 Tg CO₂-eq yr⁻¹ and 499 Tg CO₂-eq yr⁻¹, offsetting 59%, 46% and 41% of the gains in C uptake (NEE), respectively. Traditionally, Asia and other developing regions were dominated by extensive production units, which largely relied on local fodder, crop residues, unconsumed human food and natural pastures to meet the demand of milk and meat products (Steinfeld et al 2006). But, rapid population growth, urbanization, rising incomes and dietary changes led to a dramatic increase in animal numbers and per capita meat production (Tilman et al 2002), particularly in Asia. Cattle and buffalo population have increased by 1.5 fold from 200 million in 1961 to 428 million in 2014 in Southern Asia, while carcass weight (weight of an animal after being partially butchered) of cattle has increased by 1.7 times since 1961 (FAO 2019). Similarly, China that shares the largest area in Southern Asia has experienced 5.3 fold increase in pig population and 1.8 fold increase in carcass weight (FAO 2019). Changes in livestock density (SI figures S2–S4) has resulted in an increase in excreta N deposition and manure N application (SI figures S3–S6), contributing to spatial and regional variations in N₂O emissions (figure 3; SI figure S7).

4.4. Methane fluxes and net GHG balance

Our results also demonstrate that CH₄ emissions associated with fermentation of feed reserves and manure management practices had a significant effect on GHG balance offsetting 37% of the net CO₂ sink. This increase is primarily attributed to CH₄ release from enteric fermentation driven by an increase in livestock density (Dangal et al 2019; FAO 2019) and anaerobic digestion of animal manure following manure management (Petersen et al 2013). Previous studies have shown that enteric fermentation is the largest source of CH₄ emission from the livestock sector (Wolf et al 2017, Chang et al 2019) driven primarily by increases in ruminant livestock density and modified by feed quantity and quality, body weight, feeding level and the activity and health of livestock type (O’Mara 2011). Here, using the IPCC Tier II guidelines that accounts for livestock density, body weight, diet quality and energy demand for maintenance, activity, lactation and work, we found that CH₄ emissions due to fermentation of feed reserves and manure management contributed to 94% of total CH₄ emissions from grasslands. However, there were large differences in the magnitude of emissions between the grassland types, with rangelands contributing to 60% of CH₄ emissions. Both the total livestock numbers and livestock densities were higher in pasturelands compared to rangelands (SI figure S2; SI table S9). Despite the higher livestock numbers and densities, we found that pasturelands contributed to 40% of the net CH₄ emissions from livestock production. This is because rangelands have low diet quality and milk production compared to pasturelands for the same livestock type.

Our results also show large spatial and regional variations in CH₄ emissions due to ideological and economic changes (Caro et al 2014) and the transition toward intensive livestock production systems (Steinfeld et al 2006). Regionally, CH₄ emissions from livestock and manure management had a larger impact on the GHG balance in Africa and South America (SI figure S7), which are dominated by rangelands. In Africa and South America, enteric fermentation and manure management resulted in a total CH₄ emissions of 263 Tg CO₂-eq yr⁻¹ and 320 Tg CO₂-eq yr⁻¹, offsetting 37% and 75% of the positive gains in C uptake, respectively. The increase in CH₄ emissions due to enteric fermentation and manure management is attributable to higher demand of meat and milk products associated with population growth, urbanization, rising incomes and dietary changes in developing regions (Caro et al 2014). In addition, inefficient livestock production units are prevalent in these developing regions (Herrero et al 2013) likely resulting in an increase in CH₄ fluxes on both per unit area and per unit livestock basis (Dangal et al 2017). For example, the beginning of the 1990s was characterized by ideological and economic changes, which transitioned developing regions toward export oriented policies (Narula and Dunning 2000, Caro et al 2014). The export oriented market policy altered the livestock production systems in developing regions, with a large increase in livestock density (Moran and Wall 2011), particularly ruminants causing an increase in CH₄ emissions. But, in developed regions such as Europe, Oceania and North America, our results show a levelling off or even decline in CH₄ emissions after 1990 due to decline in livestock numbers (in LU units) by 49%, 18% and 9%, respectively (SI figures S3, S4).

4.5. Study limitations

While we have made efforts to include all major processes that drive variations in CO₂, N₂O and CH₄ fluxes in grassland ecosystems, several study limitations need to be addressed in our future work.
First, the largest source of uncertainty comes from the use of excreta N deposition and manure/fertilizer N application as small changes in these input data could lead to large variations in the fluxes of CO₂, N₂O and CH₄. We used spatial datasets of excreta, manure and fertilizer N inputs developed by Xu et al (2019), which has known uncertainties, particularly related to the disaggregation of FAO datasets to grid level. Second, model parameters related to the responses of C assimilation/allocation and stomatal conductance, and nitrification and denitrification rates are calibrated based on existing field observations (Pan et al 2014, Dangal et al 2019). Using field observations to parameterize the model can introduce uncertainties due to errors in measurements and mismatch in the scale when point measurements are upscaled to the resolution of model simulations (Tang and Zhuang 2008). To account for this shortcoming, we have taken additional steps to parameterize the model by calibrating to target values at several sites and evaluating the model performance using the meta-analysis of GHG fluxes from field data collected across the globe (SI tables S4–S7). Likewise, we have not included the effect of biomass burning on the fluxes of CO₂, N₂O and CH₄ from global grasslands. Fire may lead to a decline in C uptake due to combustion of aboveground biomass (Knicker 2007), while N₂O and CH₄ fluxes are generally enhanced due to wetting-drying cycles (Davidson et al 1993). However, the duration and magnitude of post fire CO₂, N₂O and CH₄ fluxes vary as a function of the frequency and intensity of fire occurrence, plant cover and soil nutrient status (Pinto et al 2002, Williams et al 2009). Experimental studies from Australian savannas and Central Africa grasslands have shown no significant effect of fire on the exchange of N₂O and CH₄ fluxes in the long term (Castaldi et al 2010, Livesley et al 2011), although there was a large but short-lived pulse of CH₄ in the 24 h after fire events (Livesley et al 2011). Castaldi et al (2010) also found no significant effect of fire on CO₂ fluxes after 8 months following fire, while Knapp et al (1998) show an increase in CO₂ fluxes by 20%–55% following fire in the US tallgrass prairie. If fires have no significant effect on N₂O and CH₄ fluxes, it is still unclear how CO₂ fluxes following fires are compensated by CO₂ uptake during the growing season (Valentini et al 2013), which ultimately determines the GHG source/sink strength of grassland ecosystems. Finally, we have not included other processes such as emissions from wild animals excreta (Smith et al 2016) and grassland management practices such as mowing/cutting frequency (Vuichard et al 2007), which could likely add some uncertainties in the estimation of GHG fluxes. While these processes can lead to additional uncertainties, the effect of cutting/mowing frequency on GHG fluxes is uncertain (Kammann et al 1998, Neftel et al 2000, Rafique et al 2012), and would possibly have a minor effect on the GHG fluxes as these practices are localized to specific regions. Likewise, the population of wild herbivores is declining at a dramatic rate (Ripple et al 2015). As a result, wild herbivores have a minor effect on GHG fluxes compared to the rapidly increasing domestic livestock populations (Crutzen et al 1986).

5. Conclusions

This study uses the data-model framework combining process-based model with spatially explicit datasets of multiple environmental factors and management activities to quantify the magnitude, spatio-temporal patterns and trends of three major GHGs (CO₂, CH₄, N₂O) in global pasturelands and rangelands during the past 50 years (1961–2010). Our study demonstrates that grassland acted as a net carbon sink and the strength of this sink increased over time. However, non-CO₂ GHGs (CH₄ and N₂O) offset 81% of the C sink during the study period. This offset was primarily driven by an increase in management intensity (grazing pressure, excreta deposition, and manure/fertilizer application) and grassland specific (pasturelands vs rangelands) variations in CO₂, N₂O and CH₄ fluxes over time. In pasturelands and rangelands, net GHG balance was dominated by an increasing contribution from N₂O and CH₄ fluxes, respectively. Our results also indicate large spatial and regional variations in GHG balance, with Europe acting as a small GHG source and other regions (i.e. Africa and Southern Asia) a strong GHG sink.

Future increases in the demand of livestock products will put enormous stress on grasslands with significant impact on the global climate system and GHG budget. Although we found that the CO₂ sink strength increased over time due to the combined effect of N addition and CO₂ fertilization, it is likely that the sink strength will reduce or even level off with continuous N accumulation when N inputs exceed N saturation threshold in the future (Bai et al 2009). In this study, we have shown the response of grassland GHGs to current management intensity. However, more experimental studies beyond current levels of N accumulation are necessary to improve the understanding of N saturation threshold, and identify when they might occur to effectively manage grasslands in a changing global environment.

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Data availability statement

The data that support the findings of this study are openly available at the following URL/DOI: https://wp.auburn.edu/cgc/.

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