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Greenhouse mitigation strategies for agronomic and grazing lands of the US Southern Great Plains

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Abstract

Challenges to sustainable agriculture are increasing with forecasts for greater climate variability, including rising temperatures, extreme precipitation events, and prolonged droughts. One important factor that contributes to the increasing climate variability is greenhouse gas emissions, including from agro-ecosystems. The US Environment Protection Agency indicates soil management and enteric fermentation from livestock contribute ~ 80% of total greenhouse gas from agriculture sector. Management practices conducive to greenhouse gas emissions, and possible mitigation strategies for the agricultural systems of Southern Great Plains, an integral part of the US beef industry, have not been thoroughly defined. The objective of this paper is to review and synthesize the literature regarding management practices conducive to emissions [carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4)] from croplands and grazing lands of Southern Great Plains, and potential strategies that may aid in greenhouse gas mitigation in the region. The results from different published studies evaluating such strategies were analyzed to determine whether these practices have potential in mitigating greenhouse gas emissions from agronomic and grazing lands. Based on the analysis, it can be recommended that increasing the amount of cropland managed by conservation tillage, fertilizer management, crop rotation systems, grazing management, and fertilizer amendments can be potential management strategies for greenhouse gas mitigation. As agro-ecosystems are very complex and reducing emissions using strategies in one sector may stimulate higher emissions in other sectors, these strategies require testing at the systems-level before they can be implemented to advise applied policies for the Southern Great Plains region.

Keywords Carbon dioxide · Nitrous oxide · Summer fallow · Fertilizer · Cover crops · Nitrification inhibitors

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1 Background

Emission of greenhouse gasses (GHGs) from human sources is a global phenomenon related to a wide range of activities. Included are activities related to industrial production, transportation and movement of goods and people, and the production of foods for humans and animals (Fissore et al. 2010; Conant et al. 2011). Fluxes in GHGs as part of the soil-plant-animal-human interface are not uniform across the planet, and emissions in one region into the atmosphere have a global interface. The concentration of GHGs in the atmosphere has increased over the past centuries, has been correlated to an array of human activities (reference), and has also been correlated to increases in global temperatures (Signor and Cerri 2013).

While developing techniques and systems at a global scale would be a direct, more strategic, method of addressing GHG emissions, such an approach will not likely occur due to the effects of regional geopolitics, and demands for services from landscapes by human populations. However, there are increasing regional-scale concerns related to effects of GHG emissions on climate, and some desire as to how they can be addressed. One specific issue in regions with agricultural economies is an increased concern of how different types of land use and landforms may affect concentrations of the three primary GHGs (carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4)) at the landscape-atmosphere interface of grazing lands and croplands (Fissore et al., 2010; Conant et al. 2011).

According to an IPCC (2014) report, agriculture accounts for roughly 6% and 24% in the USA and globally produced GHG emissions, respectively. Although these proportions are relatively small compared to GHGs added to the atmosphere through other human activities, releases from agriculture are still significant (Cole et al. 1997; Paustian et al. 1998). The proportions of total GHG emissions contributed by different agricultural sources are presented in Fig. 1 (U.S. EPA, 2008). In agriculture, CO_2 is produced by burning of plant materials or the decomposition of plant litter and soil organic matter by microbial communities through a number of production activities (Janzen 2004). In contrast, the production of N_2O from agriculture is mostly contributed by biological processes (nitrification or denitrification), with small amounts produced by non-biological processes such as chemo-denitrification (Hénault et al. 2012). Biological processes are sources of N_2O production when available soil N exceeds the amount of N required for plant growth, and the water-filled pore space of soils are greater than 60% (Smith and Conen 2004). Denitrification is a microbial process that contributes to N_2O emissions from biomass incorporated into the soil (Li et al. 2016). Nitrate (NO_3) or nitrite (NO_2^-) are reduced to N_2 through intermediate products of nitric oxide (NO) and N_2O in denitrification.

Methane is produced during microbial decay of organic material under anaerobic conditions, particularly from stored manures and flooded conditions in rice production (Smith et al. 2007). Fermentation of consumed forages in the rumen of ruminant animals, such as cattle, sheep, and goats, is also a form of microbial consumption of plant materials within an anaerobic environment (Kebreab et al. 2006; Liu and Liu, 2018). Certain landforms in agricultural areas, such as transient wetlands, or wet puddled soils, are also short-term sources of CH_4 (Conant et al. 2011).

This paper primarily focuses on review of factors related to the production of CO_2, CH_4, and N_2O in Southern Great Plains (SGP). Carbon dioxide is the most significant contributor to climate change and variability due to its high concentration, while N_2O is the most potent GHG affecting global warming. Nitrous oxide is 265–290 times as potent as CO_2 in its effects and can remain in the atmosphere for over 114 years (Follett et al. 2005; Signor and Cerri

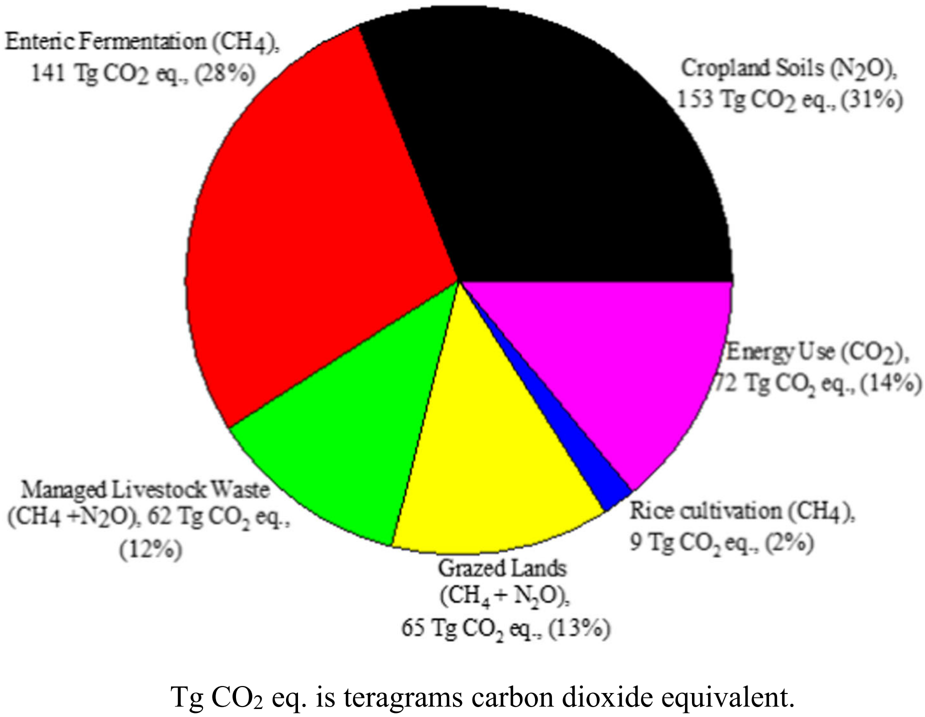

Fig. 1 Amount of greenhouse gas emissions from different agricultural sources in the United States in 2008. Tg CO_2 eq. is teragrams carbon dioxide equivalent

2013). This review presents and discusses literature on these GHG emissions from both croplands and grazing lands in the US SGP and different available mitigation strategies. Methane has 34 times greater potential than as CO_2 in its effects and can persist in atmosphere for a period of over 100 years (Smith et al. 2007). The contribution of GHG emissions from other landforms of the broad agricultural landscape that exists in the SGP, such as agroforestry or buffer strips, has not been considered in this review. However, the issues discussed related to agricultural emissions in SGP also translates to other agricultural regions and systems in semi-arid and sub humid environments.

2 Southern Great Plains

2.1 Climatic conditions

The SGP is comprised of sections of Kansas, Oklahoma, Texas, Colorado, and New Mexico and is one of the six geographic regions of the continental USA. About one third of the total area of these states is contained within the SGP (Fig. 2; Savage and Costello 1948). It is bordered by high-elevation mountainous states of Colorado and New Mexico on the western edge, and more humid states (Missouri, Arkansas, and Louisiana) on the eastern edge (Mullens et al. 2018). The boundary extends into southeastern New Mexico and adjacent areas of western Texas (Savage and Costello 1948). Due to this geographic variation, the elevation of the SGP varies from 1500 to 1800 m on the western edge of the region, to < 600 m on the eastern and southern edge.

Annual precipitation within the SGP also displays a wide range in relation to this geographic variation. As an example, precipitation in Oklahoma ranges from 380 mm in the western portions of the Panhandle to > 1200 mm along the eastern edge (Baath et al. 2018b). Roughly two thirds of the total annual rainfall is received during April through September, which is a key input for producing warm-season crops and forage grasses (Northup and Rao 2015; Savage and Costello 1948) (Fig. 3).

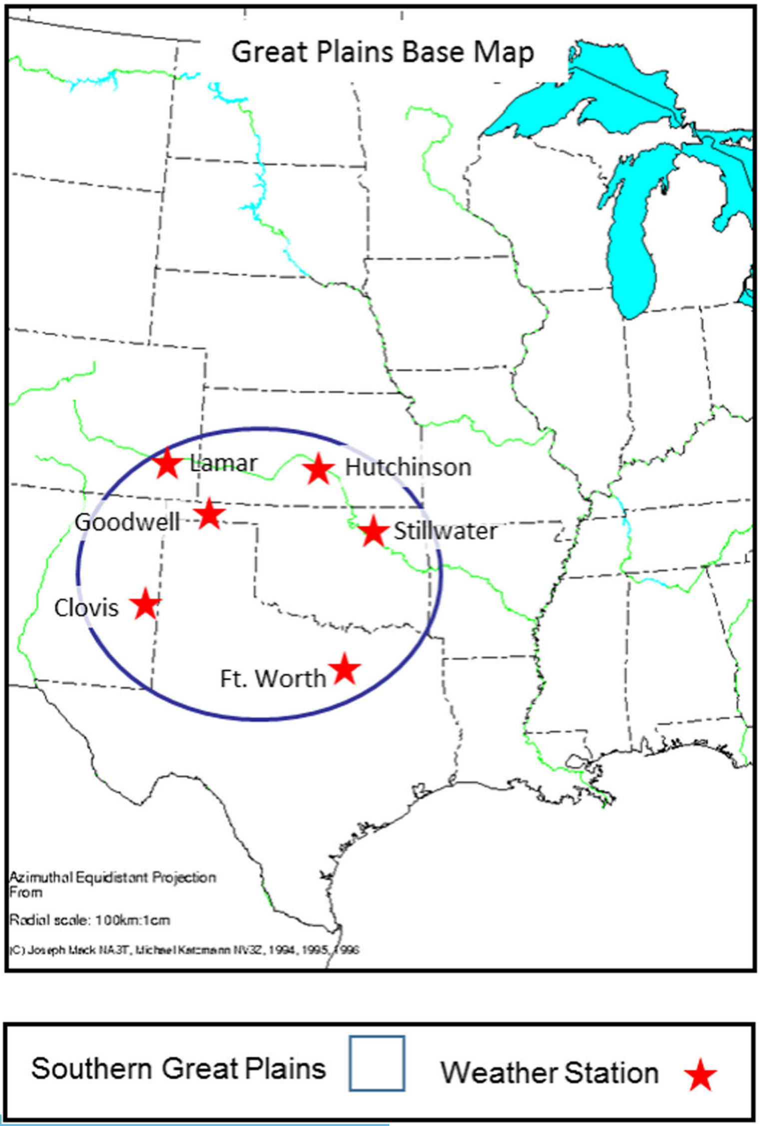

Fig. 2 The central region of the US Southern Great Plains (SGP) and six locations used to represent climate conditions

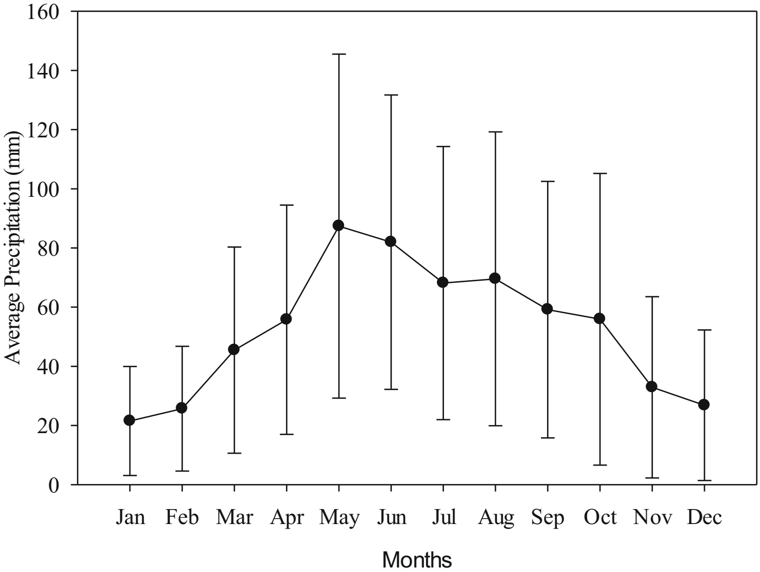

Fig. 3 Average monthly precipitation for six stations within the Southern Great Plains of the USA from 1966 to 2016. Error bars indicate the standard deviation for each month

2.2 Soil and land use

Soils of this region vary dramatically, ranging from heavy clays to dune sands in some areas, and often in close proximity (Goodman 1977; Aandahl 1982). Soils of the region include Mollisols, Alfisols, Inceptisols, Aridisols, and Vertisols (Table 1). Most of the soils in the SGP region have an ustic moisture regime and lie within thermic and mesic temperature regimes (Aandahl 1982). Many of the soils of the SGP evolved from parent materials defined as shales, siltstones or sandstones, or alluvium or colluvium derived from these materials (USDA- NRCS 2007).

The SGP covers an area of approximately 1,067,075 km^2 (412,000 mi.^{2}) and comprises 12% of the total land area of the continental USA. According to USDA-NASS (2014), about 63.3 million ha of land in the SGP are being used as grazing lands, defined as either rangeland or pastureland. These land types in the three states that constitute the majority of the SGP (Kansas, Oklahoma, and Texas) comprise ~ 30% of the total grazing lands of the USA (Peel 2003).

The median producer of cattle in the USA has small herds of cows (~ 43 heads) and limited forage resources on their farms, due to shortages of land and competing values for other uses of available land (USDA-NASS 2014). In response, weaned cattle from farms and ranches throughout the USA are sold at local livestock markets, and shipped to feedlots in the Central High Plains region for finishing, and eventual slaughter at co-located processors (Peel 2003). Most cattle weaned each year in the USA are finished in concentrated animal feedlots, in the High Plains or Midwest (Phillips and Coleman 1995). However, there is a shortage of feedlot space to simultaneously handle all cattle that are weaned annually, and large numbers spend some time grazing high-quality forage in the SGP before feedlot finishing (Peel 2003). Therefore, these lands in the SGP provide a significant contribution to the US beef production industry (Baath et al. 2018b; Peel 2003). Kansas and Texas are ranked among the top five states for numbers of total cattle on feed. Oklahoma, Kansas, and Texas are ranked among the top ten states for total cattle inventory and cattle sales (Baath et al. 2018b).

Table 1 Ecoregion provinces and portions of land resource regions, major land resource areas, and dominant soils contained within Southern Great Plains

Ecoregion	Land resource region	Major land resource	Dominant soil
331: Great Plains-Palouse Dry Steppe	Central great plains	78—Central Rolling Red Plains 77—Southern High Plains	Ustolls, Ustalfs, and Ochrepts, Orthids and Argids
	Western great plains	67—Central High Plains 69—Upper Arkansas Valley Rolling Plains 70—Pecos-Canadian Plains and Valleys	
315: Southwest Plateau and Plains Dry Steppe and Shrub	Central great plains	78—Central High Plains	Ustolls, Ustalfs, and Ochrepts
321: Chihuahuan Desert	Western great plains	67—Central High Plains 69—Upper Arkansas Valley Rolling Plains 70—Pecos-Canadian Plains and Valleys	Ustolls, Orthids, and Argids
311: Great Plains Steppe and Shrub	Central great plains	78—Central Rolling Red Plains 77—Southern High Plains	Ustolls, Ustalfs, and Ochrepts
332: American semidesert and desert	Central great plains	78—Central Rolling Red Plains 77—Southern High Plains	Ustolls, Ustalfs, and Ochrepts
251: Prairie parkland (temperate)	Central Feed grains	111—Indiana and Ohio Till Plain 112—Cherokee Prairies	Aqualfs and Udolls
255: Prairie parkland	South west Prairies	86—Texas Blackland Prairie	Usterts, Ustolls, Aqualfs, and Ustalfs
231: Southern mixed forest	East and central farming	128: Southern Appalachian ridges and valleys 129: Sand mountain	Udults, Ochrepts
M222: Ozark Broadleaf Forest—Meadow	East and central farming	128: Southern Appalachian ridges and valleys 129: Sand mountain	Udults, Ochrepts
M231: Ouachita Mixed Forest—Meadow	East and central farming	128: Southern Appalachian ridges and valleys 129: Sand mountain	Udults, Ochrepts

The primary crop grown throughout the SGP is winter wheat (*Triticum aestivum* L.). It is planted on ~ 8.3 million ha of cropland in Kansas, Oklahoma, and Texas (USDA-NASS 2014). This area represents about 30% of the total available cropland of the SGP and accounts for roughly 43% of the total wheat production in the USA. Most of the area cultivated under winter wheat in this region is utilized in a dual-purpose role, to provide fall and winter forage to beef cattle and a grain crop at the end of growing seasons (Edwards et al. 2011; Redmon et al. 1995). Roughly two thirds of all wheat acreage is used in graze-grain settings, while smaller amounts are managed in grazed-only or grain-only settings (Redmon et al. 1995). The high nitrogen content and digestibility of forage allow wheat pasture to be used to generate low-cost gains by yearling stocker cattle (Peel 2003; Fieser et al., 2006). Other major crops grown in the SGP are cotton (*Gossypium hirsutum* L.) (2.9 million ha in a semi-arid area of Texas), corn (*Zea mays* L.) (2.2 million ha), sorghum (*Sorghum bicolor* (L.) Moench) (1.5 million ha), and soybean (*Glycine max* (L.) Merr.) (1.6 million ha in total and primarily grown in eastern Kansas) (Steiner et al. 2015). The area under sorghum cultivation has been increasing due to its demand as a bioenergy crop.

The remaining area of the SGP includes commercially or naturally managed forests, which comprise small but important land areas. Among the three states that constitute the majority of the SGP, Texas has approximately 4.8 million ha of commercial forest cover, Oklahoma has roughly 4.04 million ha of forest cover mainly in central and eastern parts, and Kansas has ~ 2.10 million ha (10% of state area) of forest cover (Atchison et al. 2010, Johnson et al. 2010, Simpson et al. 2013). These forest areas provide beneficial effects to the SGP through sequestration of more permanent forms of carbon (C), biological diversity, and watershed regulating services (Steiner et al. 2015).

3 Greenhouse gas emissions from crop and grazing lands

The variable geography, climate, parent materials, and soils that define the SGP have resulted in a range of different landforms and types of management. These landforms (croplands, grazing lands, and forested areas) will act as both sources and sinks of GHGs in the SGP. Forested areas of the region provide carbon storage at the rate of −26 Tg CO_2 eq. per year (Fig. 4; Steiner et al. 2015). In comparison, CO_2 and N_2O releases occur at a rate of 43 Tg CO_2 eq.

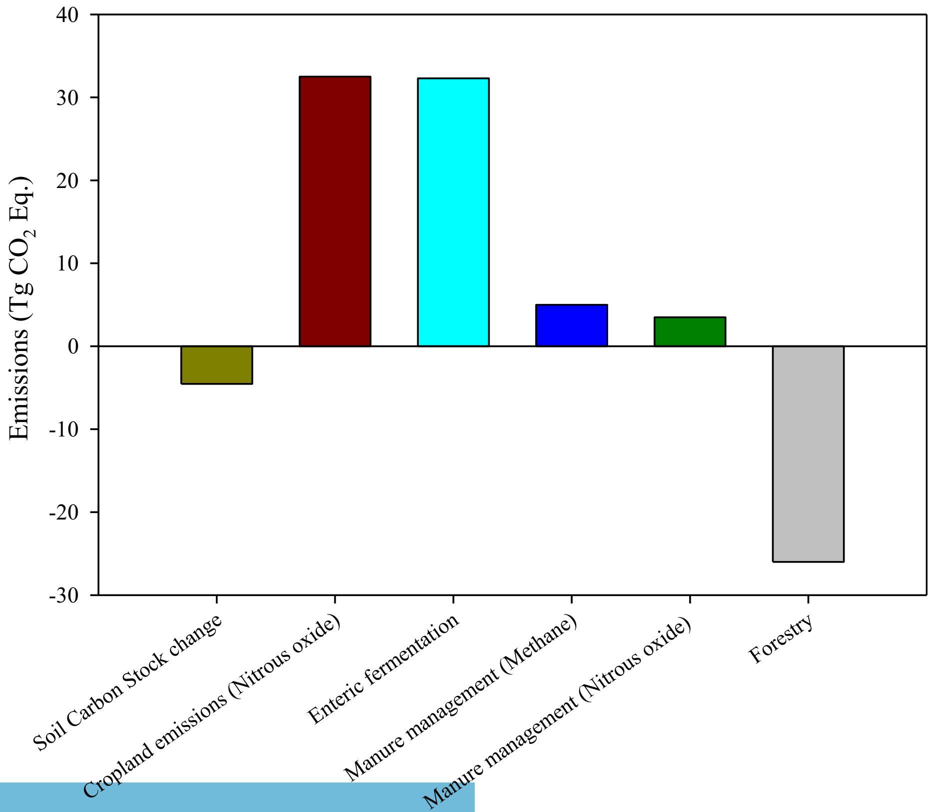

Fig. 4 Amount of GHG emissions from different sources in the SGP

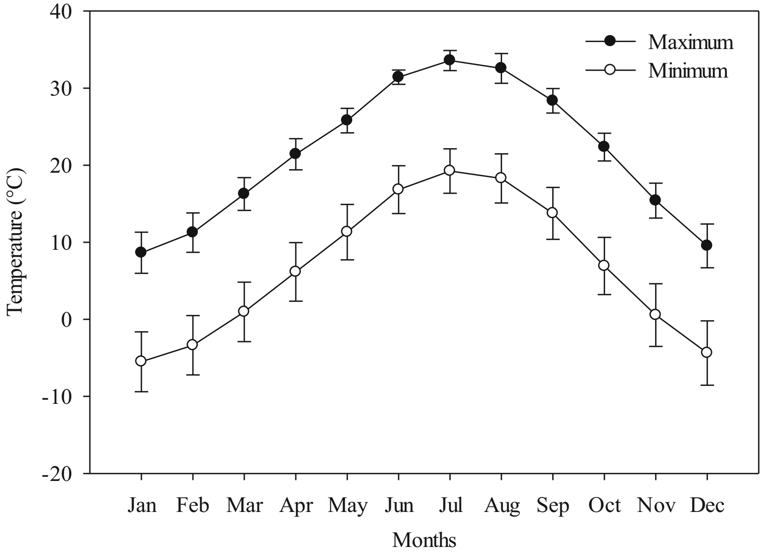

Fig. 5 Average monthly maximum and minimum air temperatures for six stations within the Southern Great Plains of the USA from 1966 to 2016. Error bars indicate the standard deviation for each month

and 33 Tg CO_2 eq., respectively, from croplands are the major GHG emissions in the region (Steiner et al. 2015).

3.1 Crop management and climate interactions

Agricultural soils can not only sequester both C and N but also contribute to GHG emissions, quite often in equal measures within individual growing seasons, depending on the type of management (Liebig et al. 2010; Liu et al. 2006; Six et al. 2004; Tribouillois et al. 2016; Wagle et al. 2018, 2019). Agricultural soils are major contributors of N_2O, which is 265–298 as potent as CO_2 as a GHG (Myhre et al. 2013; Parton et al. 2015). Application synthetic N fertilizers, livestock manures, green manures, and cover crops all have potential to produce N_2O and CO_2 emissions (Cai et al. 2017; Ciais et al. 2013; Han et al. 2017a), depending on type and amount of N and water inputs to soils, aerobic conditions within soil profiles, and soil temperatures.

Cover crops have some capacity to provide agronomic and environmental benefits such as weed control, nutrient retention, erosion control, and N supply (Bergtold et al. 2017; Tonitto et al. 2006). There is also a common perception that soil C increases and CO_2 emissions are mitigated by cover crops (Fageria et al. 2005; Lal 2004), though there is evidence to counter this premise (Basche et al. 2014; Huang et al. 2004; Kravchenko et al. 2017; Pimentel et al. 2015). However, the potential enhancement of N_2O emissions after termination of cover crops may diminish any benefits of C uptake associated with growing cover crops (Basche et al. 2014; Huang et al. 2004).

The aboveground biomass of most legumes cultivated as green N sources have low C:N ratios and high mineralization rates after incorporation, which can increase N_2O and CO_2 fluxes to the atmosphere (Basche et al. 2014). Incorporation of cover crops into the soil during wet periods may increase emissions of N_2O through rapid denitrification (Pimentel et al. 2015;

Rosecrance et al. 2000). Further, increased availability of labile C after incorporation combined with favorable soil temperatures may result in large releases of CO_2 (Kravchenko et al. 2017). While management systems that include legumes as green N sources appear to provide different positive services to society, there is a need to quantify the environmental impacts of green manures in the SGP and develop tools that allow prediction of their function in a range of environments.

3.2 Livestock-plant-soil-climate interactions: C and N dynamics

Ruminant animals derive nutrients from cellulosic (fibrous) materials, which allow utilization of millions of ha of non-grain plant materials to produce meat products for human consumption (Hristov et al. 2013; Liebig et al. 2010). Ruminants convert cellulosic material to nutrients and metabolites that are absorbed and utilized for meat, milk, and fiber production via anaerobic fermentation of consumed forage by symbiotic bacteria, protozoa, and fungi in the rumen. Both CO_2 and CH_4 are by-products of ruminant fermentation, with CH_4 production driven mainly by the quality of available forage and environmental conditions. While CO_2 and CH_4 are the primary GHGs produced by cattle in the process of rumination and metabolic activities, livestock can also drive emissions of N_2O from soils (~3.75% of all GHG emissions). Although cattle are frequently cited as major sources of GHGs released to the atmosphere by agriculture, domesticated herbivores (beef, dairy, sheep, goats, horses) account for only 1.8% of all GHGs emitted in the USA (Hristov 2012; U.S. EPA 2008).

Among the animal-related emissions in the SGP, enteric fermentation is a major CH_4 contributor (32 Tg CO_2 eq.), while manure management within confinement-based systems contributes both N_2O and CH_4 at rates ~ 8 Tg CO_2 eq. (Steiner et al. 2015) (Fig. 4). Among all livestock types, cattle are least efficient at converting the biomass (including crude protein) of consumed forages into beef; roughly 97% of the cattle in SGP are beef cattle. Most of the N in consumed crude protein is excreted through either urine or fecal matter (Cole et al. 2003; Waldrip et al. 2013). Cattle grazing grasslands generally retain < 25% N of consumed forage in body mass and excrete ~74% N as urea-N in urine (Whitehead 2000). Reports show if animal feed is high in N concentration, the N content of urine and manure are higher, and hence lead to greater amounts of N_2O production (Gupta et al. 2007). Roughly 81% of total N_2O emissions from animal excreta is contributed by beef cattle in the SGP (Steiner et al. 2015). Proper management of excreta from confinement operations, such as storage and treatment before use as fertilizer or fuel, is an important opportunity for mitigating N_2O. However, such activities are difficult to apply to the unconfined, large areas that comprise grazed paddocks of either native rangelands (Barnes et al. 2008; Augustine et al. 2013), or tame pasture (Dubeux et al. 2014). Proper management of excreta in confined systems includes storage at levels of pH, temperature, aeration, and moisture that are not conducive for N_2O production. According to Dalal et al. (2003), low pH, increased aeration, high temperature, and low moisture content during storage favor N_2O production.

Also, the implementation of GHG mitigation measure for manure-related emissions may lead to trade-off between the GHG emissions. For example- Ammonia (NH_3) mitigation strategy in solid manure storage may lower down NH_3 while it may enhance CH_4 or N_2O emissions (Szanto et al. 2007). The meta-analysis performed by Hou et al. (2015) looked up total GHG emissions budget affected due to different management techniques. It was suggested that slurry acidification lead to decrease total budget of GHG emissions. Among comparison between stockpiling and slurry pit covering, it was suggested that slurry pit covering decreased total GHG emissions while stockpiling decreased only NH_3 emissions but increased N_2O and CH_4 emissions. Therefore, accounting the trade-off between the GHG emissions before selecting the mitigation strategies for manure related emissions is very necessary. The GHG mitigation strategy should not be suggested based on some specific GHG mitigation potential while total GHG emission budget should be considered.

4 Management impacts on GHG emissions

4.1 Summer fallow

Winter wheat-summer fallow systems are the primary crop rotations used in the SGP. Prolonged drought periods of varying intensity are experienced frequently in SGP, and erratic amounts and occurrences of rainfall occur on a monthly basis (Patrignani et al. 2014; Rao and Northup 2011). Winter wheat serves as a drought avoidance crop in the region, by taking advantage of soil moisture accumulated during summer fallow (June–August) for growth during fall through spring (Baath et al. 2018b). Summer fallow minimizes risk for following wheat crop since summer crops compete for soil moisture and nutrients (Rao and Northup 2009a). Double-cropped wheat-soybean rotations in western Kansas resulted in 18% reductions of wheat forage and 31% reductions of grain yields as compared to wheat-summer fallow rotation (Aiken et al. 2013). However, wheat-summer fallow rotations are reported to have sustainability issues, particularly greater water and wind erosion, decreased amounts of soil organic C and N, and less efficient use of precipitation received during the summer period (Farahani et al. 1998; Kelley and Sweeney 2010).

It is important to account for soil organic carbon (SOC), both spatially and temporally, to understand both production and mitigation of GHG emissions from production systems applied to winter wheat and other crops (Tan et al. 2005). Soils in the SGP possess the lowest amounts of SOC among different regions in the USA, averaging 96.39 Mg ha^{-1} in 0.7 m soil profile (Bronson et al. 2004). Further, 94% of croplands in the SGP have shown reductions in amounts of SOC over time, with > 6.72 Mg ha^{-1} lost over 30 years (Parton et al. 1987). One of the suggested reasons explaining this decrease was the widespread use of winter wheat-summer fallow rotations (Aulakh et al. 1982).

An increase in CO_2 sequestration from ~738 Tg CO_2 equivalence in 1990 to ~884 Tg CO_2 equivalence in 2006 was observed within the SGP, which was related to reductions in croplands under summer fallow in semi-arid areas, changes in land use, and adoption of conservation tillage (Follett 2010). Such results indicate continuous soil cover may help in sequestering C by reducing CO_2 emissions from croplands under summer fallow (Desjardins et al. 2001). Ten different peer-reviewed studies were selected which compared the SOC in two different crop rotations (with and without summer fallow). The results from these studies were compared using the paired *t* test. Synthesis of 10 different studies suggested that C sequestration would be significantly greater ($p < 0.001$) with crop rotations that eliminate summer fallow (22 Mg ha^{-1}), compared to rotations including summer fallow (19.96 Mg ha^{-1}; Table 2).

Summer fallow has also been identified as a practice that can result in high losses of N from soils. Summer rainfalls and temperatures at the end of fallow periods in the SGP are conducive for N loss as N_2O from the soils (Wilson et al. 2016). The average temperature during May through September varies dramatically across the region, ranging from lows of 10°C and 18°C

Table 2 Estimates of soil C (Mg ha^{-1}) sequestration potential for eliminating summer fallow

Location	Soil taxonomy	Soil depth	SOC (summer fallow)	SOC (eliminated summer fallow)	Source
Semi-arid regions of the USA			20.2	23.2	Sperow et al. (2001)
Akron, CO, USA	Weld loam	0–15 cm	15.0	17.0	Bowman et al. (1999)
Bushland, TX, USA	Clay loam	0–20 cm	28.2	32.6	Potter et al. (1997)
Bushland, TX, USA	Clay loam	0–20 cm	14.7	16.1	Peterson et al. (1998)
Havre, MT, USA	Scobey clay loam	0–20 cm	19.4	20.6	Sainju et al. (2006)
Mandan, ND, USA	Temvik-Wilton silt loam	0–15 cm	18.2	21.4	Halvorson et al. (2002)
Sterling, CO, USA	Aridic Paleustolls	0–15 cm	24.9	23.5	Sherrod et al. (2003)
Saskatchewan, Canada	Swinton silt loam	0–15 cm	27.1	29.6	McConkey et al. (2003)
Saskatchewan, Canada	Hatton fine sandy loam	0–15 cm	17.2	19.2	McConkey et al. (2003)
Saskatchewan, Canada	Swinton silt loam	0–7.5 cm	14.7	16.8	Curtin et al. (2000)
Mean			19.96	22	

in the northern and southern regions, respectively, with summer means usually above 21°C (Savage and Costello 1948) (Fig. 5). As there is no crop for N uptake during this period, N losses can be high, and thus affects soil fertility of wheat-summer fallow rotations (Aulakh et al. 1982). An additional factor causing high N losses during periods of summer fallow is the large fractions of annual rainfall received during this period, which creates conditions conducive for nitrification and denitrification (Savage and Costello 1948) (Fig. 3). Therefore, N losses can occur as either N_2 or N_2O. Overall, the practice of summer fallow within cropping systems applied to winter wheat is conducive to the production of both CO_2 and N_2O from croplands in the SGP.

4.2 Tillage

The 1930's drought that occurred in the SGP, combined with wind erosion, made crop production difficult in the region. In response, conservation practices such as windbreaks, and tillage systems like reduced tillage and no-tillage (defined as conservation tillage) which leave crop residues on the soil surface to control soil erosion, were introduced (Unger and Baumhardt 2001). The main difference between conservation and conventional systems of tillage is that the former limits soil disturbance and leaves the soil surface covered with crop residues, while the latter applies different forms of tillage and leaves few or no residues on the soil surface (Aulakh et al. 1982). Though the application of conservation tillage has merits related to soil conservation, the adoption of no-tillage or reduced tillage systems is currently limited in the SGP (Unger and Baumhardt 2001). For example, a survey conducted in Oklahoma found that only 8% of the total area under continuous wheat-fallow rotations is managed by no-till, while 36% and 56% of the area is managed under reduced tillage and conventional tillage, respectively (Hossian et al. 2004).

Table 3 Comparison of soil C sequestration between different tillage practices

Location	Soil taxonomy	Soil depth	No till	Conventional till	Source
College Station, TX, USA	Fluventic Ustochrept	20	25.3	23.9	Franzluebbers et al. (1994)
College Station TX, USA	Fluventic Ustochrept	20	25.4	23.0	Franzluebbers et al. (1995)
Columbus, OH, USA	Typic Argiaquolls	15	50.0	47.0	Puget and Lal (2005)
Corpus Christi, TX, USA	Typic Ochraqualf	20	23.2	18.4	Potter et al. (1998)
Corpus Christi, TX, USA	Typic Ochraqualf	20	19.0	17.6	Potter et al. (1998)
Corpus Christi TX, USA	Typic Ochraqualf	20	21.6	18.7	Salinas-Garcia et al. (1997)
Merchouch plateau, Morocco	Vertisols	20	10.0	9.7	Moussadek et al. (2014)
Shaanxi Province, China	Middle loam	20	7.0	2.5	Wang et al. (2018)
Temple, TX, USA	Udic Pellustert	30	56.9	53.0	Potter and Chichester (1993)
Temple, TX, USA	Udic Pellustert	30	63.1	61.2	Reicosky et al. (1997)
Temple, TX, USA	Udic Pellustert	20	47.4	46.0	Potter et al. (1998)
Tribune, KS, USA	Richfield silt loam	20	19.3	17.5	Stone and Schlegel (2010)
Mean			30.69	28.21	

Emissions of GHGs vary with the type of tillage system, with higher emissions reported for conventional tillage than no-till. Emissions of CO_2 increases by 62–118% from conventionally tilled barley compared to no-till (Sainju et al. 2008). One explanation for decreased CO_2 emissions under no-till is the slow oxidation of organic C relative to conventional tillage (Abdalla et al. 2013). Conventional tillage breaks down soil organic matter (SOM), produces CO_2, and results in reduced total soil C content and C sequestration, which explains the decreased amounts of SOC in 94% of the cropland area of the SGP (Abdalla et al. 2013).

Twelve different peer-reviewed studies were selected which compared SOC in two different tillage systems (conventional tillage and no-tillage). The results from these studies were compared using the paired t test. From a compilation of these literature citations, SOC was 28.21 Mg ha^{-1} under conventional tillage, which was significantly lower ($p < 0.01$) than that of 30.69 Mg ha^{-1} under no-till (Table 3). The ratio of SOC under no-till to conventional till was 1.19 ± 0.14 kg kg^{-1}, which was significantly different from 1 ($p < 0.01$). These values indicate SOC should be an average of 19% greater under no-till than conventional till. Some contrasting results showed no difference in CO_2 emissions between no-till and conventional till, or higher emissions under no-till treatments (Kainiemi et al. 2015), which can be explained by the effects of soil texture and climatic conditions on GHG emissions under different tillage systems (Abdalla et al. 2013).

Emissions of N_2O are higher under conventional tillage than conservation tillage (Chatskikh and Olesen 2007; Gregorich et al. 2006). However, contradictory results have been reported, which show higher N_2O emissions from conservation than conventional tillage (Arah et al. 1991), or no effect of tillage systems (Liu et al. 2006). Factors that cause higher N_2O emissions from conservation tillage could be greater bulk density, the presence of more soil moisture, or increased activity of microorganisms, which increase the rates of nitrification and denitrification (Palma et al. 1997). Further, N_2O emissions are dependent on temperature, soil properties (Flechard et al. 2007), and the length of time cropland has been managed under conservation or conventional tillage (Six et al. 2004). Therefore, suitable tillage operations such as reduced tillage should be encouraged to mitigate GHG emissions in SGP.

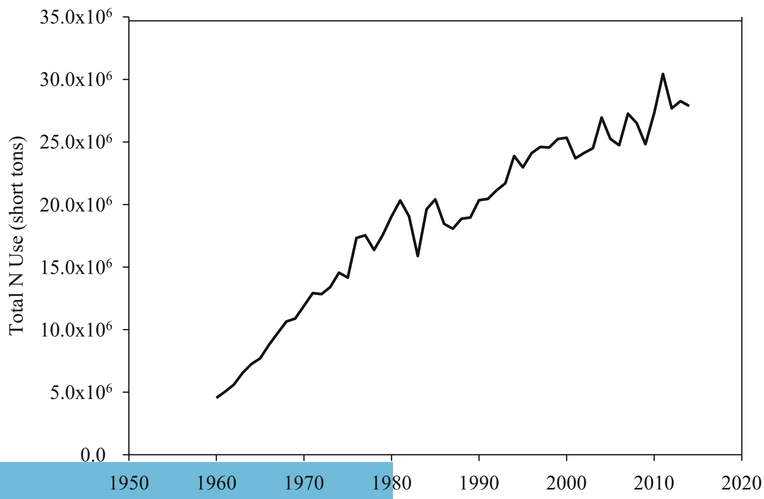

Fig. 6 Increase in total nitrogen use in the USA from 1960 to 2014

4.3 Nitrogen fertilizer use

The use of N fertilizers (synthetic, manure, and fixed N) has increased exponentially at the global scale since 1960, with synthetic fertilizers being the most significant contributor (Fig. 6) (Lassaletta et al. 2014). The use of N fertilizers has also increased in the SGP since the 1960s. According to USDA-NASS (2014), the use of N fertilizer has risen from 434,309 to 525,073 Mg in Oklahoma from 1985 to 2014. In Texas, the consumption of N fertilizer for corn production increased from 50 kg ha^{-1} in 1964 to 139 kg ha^{-1} in 2016, while use of N fertilizers in Kansas increased from 67 to 160 kg ha^{-1} (USDA-ERS 2018).

Nitrogen use efficiency (NUE) is defined as the ratio of total crop production to total N inputs and indicates that N is generally supplied in excess of plant use. Consequently, the excessive amounts of applied N are often lost as nitrates through leaching, or emitted as different forms of nitrogen (N_2, N_2O), thus leading to N-related environmental contaminations (Galloway et al. 2003; Lassaletta et al. 2014). Results from different meta-analyses concluded that exponential increases in N_2O emissions from croplands occur due to N fertilizer applied in excess of crop needs (Basche et al. 2014; Han et al. 2017b; Shcherbak et al. 2014). However, a regression model showed that only 2.5% of the N fertilizer applied between 1860 and 2005 was converted to N_2O (Davidson 2009).

The N_2O emissions from N fertilizer depend on several factors, and the type of N fertilizer is an important factor. Emissions of N_2O among different N fertilizer applications decreased in the order of anhydrous ammonia (1.57 kg N ha^{-1}) > organic fertilizers (1.49 kg N ha^{-1}) > urea (0.31 kg N ha^{-1}) > ammonium nitrate (0.30 kg N ha^{-1}) > nitrate salts (0.18 kg N ha^{-1}) > ammonium salts (0.12 kg N ha^{-1}) (Bouwman 1994). According to the USDA-NASS (2014)

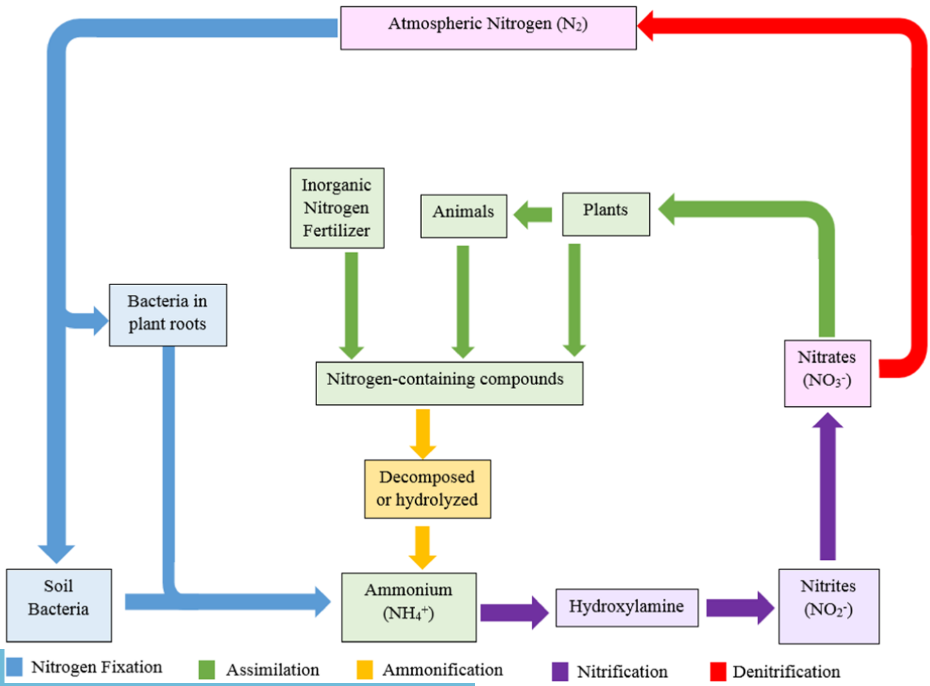

Fig. 7 Nitrogen cycle

report, the use of urea and anhydrous ammonia are more heavily utilized in the SGP than nitrate and ammonium salts, which may be attributed to lower prices and transport costs of urea, and its high N content. Therefore, the large-scale use of these fertilizers can also be a possible reason for high N_2O emissions from agricultural lands in the SGP. High N_2O emissions from urea occur as it hydrolyzes on contact with soil, resulting in rapid increases in NH_3 production (Van Der Weerden et al. 2016) (Fig. 7). Further nitrification of NH_3 to nitrate NO_3 leads to the release of some N_2O, and if conditions are favorable, this NO_3 is further denitrified, and large amounts of N_2O are produced through denitrification (Dobbie and Smith 2003).

Another factor affecting NUE is the mode of application of N fertilizer, particularly the placement of fertilizer into the soil. Urea is generally surface-broadcast in dryland systems of the SGP, where incorporation depends on unpredictable precipitation, and air temperatures are generally high at planting (Adams et al. 2018). The high temperature and surface broadcasting of urea provide favorable conditions for hydrolysis of urea. Once the urea is hydrolyzed to Ammonium (NH_4), it can be lost directly by volatilization or indirectly as N_2O (Engel et al. 2011). Therefore, loss of N either by volatilization or as N_2O via nitrification and denitrification can occur in response to warm temperatures and rainfall.

4.4 Grazing practices

Since a large number of animals spend time grazing as stocker cattle in the SGP, massive amounts of GHG emissions occur due to N_2O production from urine patches or CH_4 production from enteric fermentation in livestock (Hristov et al. 2013). Nitrous oxide is the predominant GHG emitted from soils of grazing lands, accounting for ~96% of all GHG emissions from grazing lands; the remaining 4% are mainly CH_4 (U.S.-EPA 2008). Reports on the sources of GHG emissions from the grazing lands of the SGP differ from other regions in the USA and other countries in the world. Overall, GHG emissions from cow-calf production in SGP were reported to be higher than elsewhere in the USA (Wang et al. 2015). Further, the large numbers of yearling stockers that graze wheat pasture annually (Edwards et al. 2011; Redmon et al. 1995) have significant potential to contribute GHGs (Kandel et al. 2018), due to high N concentrations and digestibility of wheat forage, and low amounts of N retention in cattle (MacKown and Northup 2010).

Soils of grazing lands can emit N_2O due to enhanced nitrogen cycling, in addition to small amounts of CH_4 emissions from manure deposits, or from grazing cattle. Manure deposited on grazing lands (i.e., unmanaged manure) produces little CH_4 due to predominantly aerobic conditions after deposition. In comparison, direct and indirect N_2O emissions are associated with increased N from forage legumes and waste from grazing animals, respectively (U.S.-EPA 2008). One primary driver of significant GHG emissions from perennial grasslands grazed by cattle is deposition of N (and C) in excreta within paddocks. Cattle grazing rangelands or tame pasture consume biomass from the entire area of paddocks and redeposit both C and N in areas of paddocks in urine and feces. This re-deposition can result in uneven distributions of N and C inputs to soils that can act as localized sources of GHG emissions or within smaller zones of entire paddocks. Research in shortgrass prairie in northeastern Colorado noted that cattle wearing global positioning system collars spent ~27% of their total time on paddocks in locations (water sources and corners) that represented just 2.5% of the total area of 65 to 130 ha paddocks (Augustine et al. 2013). Cattle redistributed 49% of all N in consumed biomass to these areas, which resulted in potential point-source N pollution. Other studies in Florida noted higher amounts of NO_3 in areas close to water or shade in smaller paddocks of tame warm-season grasses (Dubeux et al. 2014; Mathews et al. 1994). Pineiro et al. (2010) concluded that there was a diversion of N, from the conversion of NO_2 to ammonia gas, through a reducing release of N_2O from (compacted) soils to the atmosphere.

There is a general expectation of the occurrence of specific hotspots of mineral N, or higher overall amounts of mineral N, in soils within grazed paddocks, or portions of grazed paddocks, under continuous stocking. This premise is based on the greater opportunity (more time) for cattle to congregate in local areas (e.g., water sources) of paddocks, relative to the paddock as a whole, resulting in less-even N distributions (Bailey et al. 1996). However, the response of mineral N in soils to the form of stocking can be variable, and application of a stocking method purported to achieve uniform distributions of grazing and pasture use may not achieve such usage (Northup et al. 2019). A study comparing effects of 2 and 12-day rotations with continuous stocking on bermudagrass (*Cynodon dactylon* (L.) Pers.) paddocks reported no differences in paddock-scale amounts of NO_3-N, a precursor for N_2O emissions, but higher accumulations in the one third of paddock areas closer to water and shade across all forms of stocking (Mathews et al. 1994). Similar results were noted in bahiagrass (*Paspalum notatum* Flueggé) paddocks managed using similar stocking methods (Dubeux et al. 2014).

One reason used to explain increased N_2O emissions from grazed paddocks is increased compaction of soils. Sharrow (2007) reported 13% higher bulk density and 7% lower porosity in compacted soil due to grazing livestock compared to soils in non-grazed paddocks. Studies have reported higher N_2O emissions from compacted soils compared to non-compacted soils due to increases in soil bulk density from foot traffic, water-filled pore space, and penetrometer resistance in compacted soils (Bhandral et al. 2007; Hamza and Anderson 2005).

The second major GHG from grazing lands is CH_4, which is generated via rumination by grazing animals, and in soils in response to flooded, anaerobic conditions. Production of CH_4 by cattle is an unproductive loss of dietary energy induced by enteric fermentation in livestock. In ruminants, carbohydrates are broken down and fermented by microbes in the rumen. Methane is a by-product of this process, which aids in maintaining favorable pH in the rumen by acting as a sink for hydrogen ions (Kebreab et al. 2006; Liu and Liu, 2018). The amount of CH_4 production by enteric fermentation is governed by various factors, including the age and weight of the animal, and the quality and quantity of feed (Liu and Liu, 2018). Therefore, providing high-quality forage with more digestible nutrients to grazing animals can reduce emissions of enteric CH_4 (Huhtanen and Hetta 2012). Forage quality can be maintained by avoiding over-grazing, which may be attained by applying stocking rates at levels that are below carrying capacity of the grassland, or application of stocking methods, such as rotational grazing (Wang et al. 2015). But at same time, the high digestibility of forage may also lead to higher N excretion as manure leading to higher N_2O emissions. Therefore, while recommending GHG mitigation strategy for grazing lands, the trade-off between GHGs should be considered as discussed earlier.

5 Potential management strategies for mitigating GHG emissions

5.1 Replacing summer fallow with forage crops

Considering the issues of GHG emissions and sustainability associated with summer fallow, replacing this period with a growing crop could be an effective mitigation strategy.

However, crop selection to replace summer fallow is critical, given the prevailing agro-climatic conditions that exist in the SGP. Agriculture in the region is mostly rainfed, and the availability of soil moisture for following crop of winter wheat needs to be considered when selecting a crop species to replace summer-fallow (Northup and Rao 2015). Further, the region experiences highly erratic rainfall in terms of timing and amounts, and both short-term and prolonged droughts are a common feature of the SGP throughout the calendar year (Schneider and Garbrecht 2003). Therefore, the selection of crops to replace summer fallow should account for their capacity to perform under the high temperatures and variable amounts of soil moisture that occur during summers of the SGP. The selected crop must also have minimal effects on soil resources required by the following wheat crop (Rao and Northup 2008; Rao and Northup 2009a; Rao and Northup 2009b). Additionally, it would be beneficial if summer fallow is replaced by legume crops to provide a source of N for the subsequent wheat crop.

Another important factor to consider for the selection of summer crops would be their capability to produce quality forage that can contribute to the beef cattle industry in the SGP (Northup and Rao 2015). The region requires high-quality forage year-round to meet growth and maintenance requirements for yearling stocker cattle (Duckett et al. 2009). Most of the commonly used forages for summer grazing in the SGP are perennial warm-season grasses, such as bermudagrass, old world bluestems (*Bothriochloa* spp.), or native prairie (Coleman and Forbes 1998; Phillips and Coleman 1995; Phillips et al. 2003), which decline in quality with maturation as the growing season advances and temperatures increase. These perennial grasses become a limiting factor to growth by stocker cattle during the latter portions of summer in the SGP. Therefore, there is a need to evaluate other species for their capacity to provide nutritious forage that can fill the forage quality slump during late-summer without negatively affecting the soil resources important for growth and development of winter wheat (Rao and Northup 2009b).

Over the last two decades, various annual grain legumes (pulses) were evaluated for their potential to serve as grazed pasture (Rao and Northup 2012), forage, or green manure in the SGP (Northup and Rao 2015; Rao and Northup 2009b; Rao and Northup 2011). Some of these tested pulses can produce large amounts of high N biomass in the region (Baath et al. 2018a). These studies have focused mainly on the yield of legumes and winter wheat, or the capability of legumes to supply N for subsequent wheat crops. However, data regarding the year-round emissions of GHGs during the growing period of legumes and after soil incorporation (during the growth phase of wheat) is scarce. Therefore, it is essential to quantify the N dynamics associated with the use of summer legumes for grazing or green manure, and GHG emissions on a year-round basis in winter wheat-summer crop rotations. Such studies would help to define the capability of summer legumes to serve as a strategy to mitigate CO_2 and N_2O emissions during their growth period and the potential risks of N_2O emissions after their incorporation (Han et al. 2017a).

A recent study in the SGP reported significant emissions of N_2O during and after high rainfall events following soil incorporation of hairy vetch (*Vicia villosa* Roth) in late-spring (Kandel et al. 2018). These emissions were likely due to the low C:N ratio of hairy vetch, which is conducive for rapid mineralization of crop biomass after incorporation into the soil (Singh et al. 2019). However, the N_2O emissions approximated zero during the active growth phase of the subsequent summer crop. There are various methods which can help to reduce N_2O emissions after incorporation of cover crops and thus provide strategic reductions in N application rates (Han et al. 2017b). Some of the possible strategies to reduce N rate application using cover crops are to: grow mixtures of legume and non-legume cover crops (Han et al. 2017b), or remove the aboveground biomass of cover crops before termination and incorporation (Basche et al. 2014). The practice of harvesting aboveground biomass could provide multiple benefits, by generating forage for stocker cattle, mitigating GHG emissions during the summer, and providing N sources to following wheat crops from root biomass, assuming the cover crops accumulate sufficient biomass and N in roots (Kandel et al. 2019). Thus, there is a need to evaluate the effects of adopting such practices on year-round GHGs emissions from the croplands in SGP.

5.2 Growing crops with properties that inhibit biological nitrification

Planting forage or grain species that inhibit biological nitrification in soils is one potential strategy to reduce N_2O emissions from croplands. Nitrification is a key component in the soil N cycle is nitrification, a microbial-mediated process which converts immobile NH_4 into highly mobile NO_3. The end-product of nitrification (NO_3) is highly susceptible to losses from the root zone either through leaching or denitrification, thereby leading to substantial economic losses and GHG emissions. About 15 billion US dollars of fertilizer are lost annually, in addition to environmental degradation that occurs through groundwater pollution, increased GHG emissions, and eutrophication of surface water (Giles 2005; Raun and Johnson 1999; Subbarao et al. 2006). Thus, maintaining N fertilizers in reduced form (NH_4) by suppression of nitrification is important to minimize the loss of N fertilizer as N_2O (Sun et al. 2016). However, the only way to suppress nitrification is to impair the function of the responsible microbial species.

Nitrification of the ammonical form of N in soil takes place in two steps, which may be targeted to reduce N_2O emissions. First, the ammonia-oxidizing bacteria (AMO) such as *Nitrosomonas* sp. oxidizes ammonia to hydroxylamine using ammonia monooxygenase (Fig.7). The hydroxylamine is then oxidized to nitrite by *hydroxylamine oxidoreductase* (HAO). Therefore, one or both steps can be targeted to inhibit nitrification by impairing the activity of bacteria (*Nitrosomonas* sp.) responsible for these processes (Sun et al. 2016). There are some synthetic nitrification inhibitors used in agriculture such as nitrapyrin, dicyandiamide (DCD), and 3,4-dimethylpyrazole phosphate (DMPP). However, there are also some plant species that possess the ability to suppress nitrification, including koroniva grass (*Brachiaria humidicola*), signal grass (*Brachiaria decumbens*), and sorghum (*Sorghum bicolor*), by inhibiting the AMO bacteria *N. europaea* (Subbarao et al. 2006; Sun et al. 2016).

The roots of these plant species are known to release exudates which have chemicals that inhibit AMO (*Nitrosomonas* sp.), resulting in biological nitrification inhibition (BNI) (Subbarao et al. 2006). However, the nitrification inhibition properties of these crops have only been evaluated in vitro by growing in hydroponic cultures and collecting their root exudates for trials. In contrast, in vivo studies are scarce. Therefore, growth chamber or field studies are required to analyze the ability of such crops to inhibit biological nitrification in soils, so that areas under such crops can serve as potential mitigation strategies for GHGs.

The amount of area under sorghum is increasing in SGP due to its drought tolerance and is giving competition to corn acreage (Tolk and Howell 2008). However, there are still some factors, such as yields and price uncertainty that affect the acceptability of sorghum as a crop in the SGP compared to corn (Taylor and Brix 2013). The ability of sorghum to inhibit nitrification would decrease losses of N fertilizers and aid in mitigating GHG emissions. Therefore, field studies involving crops like sorghum or *Brachiaria* spp. should be conducted to evaluate their ability to mitigate GHGs by inhibiting nitrification.

5.3 Nitrogen fertilizer management

Increasing demands for food around the world would not allow reductions in the usage of N fertilizers to decrease N_2O emissions. Therefore, the only solution to reduce N_2O emissions from croplands without jeopardizing global food production is to enhance NUE. Strategies available to improve NUE and reduce N losses as N_2O include: improving fertilizer management, such as banding or subsurface placement of N fertilizer; use of more stable forms of N fertilizer than urea; using in-season N applications or foliar applied N; and using precision agriculture practices (Raun and Johnson 1999). There are four management factors that are a definable strategy, known as 4 R's, for reducing N_2O emissions from applied N fertilizer (Millar et al. 2014). The 4 R's stand for:

- Right N application rate
- Right formulation (fertilizer type)
- Right timing of application
- Right placement

Amount of inorganic N in soils is the single best predictor of N_2O emissions from croplands (Bouwman et al. 2002). Application of more N fertilizer than is required for crops leads to increased amounts of inorganic N in soils and may thereby result in N_2O emissions. Therefore, a proper rate of N application, where N application meets crop requirements, is important as N_2O emissions increase exponentially with increasing amounts of applied N fertilizer (Millar et al. 2010). A study evaluating the effect of rates of N fertilizer application on N_2O emissions from switchgrass (*Panicum virgatum* L.) in Kansas showed reductions in grain yields, but rises in N_2O emission factor of 2.1 to 2.6%, when amounts of applied N increased from 100 to 150 kg N ha^{-1} (McGowan et al. 2018). The possible explanation for this response was that N availability increased above crop requirements at the higher rate of fertilization. Also, if the N application exceeds optimum rates for production of crops such as corn and wheat, similar increases in N_2O emissions will likely occur (Millar et al. 2018; Song et al. 2018). Different peer-reviewed studies evaluating N_2O emissions from adequate and high N treatments were

Table 4 Comparison of N_2O (kg N_2O-N ha^{-1}) emissions between different rates of N fertilizer

Location	Soil taxonomy	Higher N rate	Lower N rate	Source
Ames, IA, USA	Typic Calciaquolls	0.50	0.61	Breitenbeck et al. (1980)
Bennekom, Netherland	Poorly drained sand	4.70	1.50	Velthof et al. (1996)
Bozeman, MT, USA	Frigid Typic Haplustolls	0.78	0.61	Dusenbury et al. (2008)
Carlow, Ireland	Sandy loam	0.63	0.42	Abdalla et al. (2010)
New Brunswick, Canada	Orthic Humo-Ferric Podzols	3.60	1.70	Zebarth et al. (2008)
Northeastern Colorado, USA	Mesic Aridic Haplustalfs	3.00	1.80	Liu et al. (2005)
Quebec, Canada	Humic Gleysol	1.80	0.78	MacKenzie et al. (1997)
Quebec, Canada	Humic Gleysol	2.62	2.06	MacKenzie et al. (1998)
St. Paul, MN, USA	Hapludolls	0.84	0.78	Venterea et al. (2016)
Wageningen, Netherland	Typic endoaquoll	0.25	0.17	Van Groenigen et al. (2004)
Mean		2.01	1.04	

compiled, and results from these studies were compared using the paired *t* test. Treatments using N rates that exceeded crop requirements showed average N_2O emissions of 2.01 kg N_2O-N ha^{-1} which were significantly greater ($p < 0.05$) than average N_2O emissions (1.04 kg N_2O-N ha^{-1}) produced by the treatments using N rates that met crop requirements (Table 4). As defined by Ribaudo et al. (2011), the best quantity of applied N for mitigating N_2O emissions is to apply no more than 40% of N being removed at crop harvest. This amount includes N supplied by both commercial and manure-based sources, carryover amounts from the previous crop, irrigation, and atmospheric deposits. Therefore, application of N fertilizer according to crop requirements is an important tool to mitigate N_2O emissions.

The second factor that can alter N_2O emissions from croplands is the type of N fertilizer used. According to the USDA-NASS (2014) report, the use of urea and anhydrous ammonia as N fertilizer is higher compared to nitrate and ammonium salts in the SGP. However, the emission factors for urea (0.19) and anhydrous ammonia (0.50) are reported to be higher than nitrate (0.04) or ammonium salts (0.15) (Bolle et al. 1986). The difference in N_2O emissions from ammonium salts and urea are still debatable because the reported effects correspond to different seasons. It is reported that replacing urea with ammonium salts during spring may lead to increased emissions due to warm and wet conditions, while N_2O emissions during dry summer seasons would be lower (Harrison and Webb 2001).

Nitrogen fertilizers capable of lowering N_2O emissions from croplands also consist of formulations modified with various inhibitors, such as nitrification or urease inhibitors, or both, which are also known as enhanced efficiency fertilizers (Dobbie and Smith 2003; Harrison and Webb 2001). According to Halvorson et al. (2014), enhanced efficiency fertilizers are products prepared by using some additives or coatings to increase nitrogen use efficiency through controlled release or modified soil-fertilizer reactions. Examples of products marketed as enhanced efficiency N fertilizers include a controlled-release, polymer-coated urea (PCU), ESN; a stabilized urea containing urease and nitrification inhibitors, SuperU; S-coated urea, a coated slow-release urea; anhydrous NH_3 containing nitrapyrin, a nitrification inhibitor, making it a stabilized N source; and UAN + AgrotainPlus, a stabilized UAN solution containing urease and nitrification inhibitors. However, there has been only limited use of such fertilizers by wheat producers in the SGP (Adams et al. 2018).

Different peer-reviewed studies evaluating N_2O emissions from normal urea compared to enhanced efficiency N fertilizers were selected, and the results from these studies were compiled and compared using the paired *t* test. Analysis of the studies using such forms of N fertilizer showed average N_2O emissions of 1.02 N_2O-N ha^{-1} which were significantly lower ($p < 0.05$) than average N_2O emissions produced from normal urea fertilizer treatment (1.91 kg N_2O-N ha^{-1}; Table 5). Therefore, the use of nitrate or ammonium salt-based fertilizers, or fertilizers modified with inhibitors, could help reduce N_2O emissions from croplands in the SGP. However, the cost of these fertilizers is usually higher than standard N fertilizers, which discourage their adoption by farmers to some extent. Unless the economic value of reducing loss of N in fertilizers to the atmosphere and deep percolation is soil water, along with their costs of environmental degradation to producers, can be quantified, use of such fertilizers in the wheat-based systems of the SGP will likely be limited.

Another management tool to reduce N_2O emissions is to apply fertilizers at times when the crop needs N, so that there is synchronization between the supply and uptake of N by the growing crop (Hodge et al. 2000; Robertson and Vitousek 2009). Increase in NUE could be achieved by delaying N application at planting to early vegetative stages of growth, just before the rapid growth phase, which was reported to decrease N_2O emissions and enhance N uptake

Table 5 Comparison of N_2O (kg N_2O-N ha^{-1}) emissions between different N fertilizer types

Location	Soil taxonomy	Duration of emission measurement	Urea	Other efficient forms	Source
Ames, IA, USA	Typic Calciaquolls	140 days	0.66	0.44	Breitenbeck and Bremner (1986)
Ames, IA, USA	Typic Calciaquolls	96 days	0.61	0.35	Breitenbeck et al. (1980)
Bowling Green, KY, USA	Typic Paleudalfs	Corn-growing season	2.50	2.34	Sistani et al. (2011)
Fort Collins, CO, USA	Mesic Aridic Haplustalfs	Corn-growing season	0.83	0.39	Halvorson et al. (2010a)
Fort Collins, CO, USA	Mesic Aridic Haplustalfs	Barley-growing season	0.80	0.64	Halvorson et al. (2010b)
Fort Collins, CO, USA	Mesic Aridic Haplustalfs	Corn-growing season	1.63	0.77	Halvorson and Del Grosso (2012)
Fort Collins, CO, USA	Mesic Aridic Haplustalfs	Corn-growing season	0.90	0.63	Halvorson and Del Grosso (2013)
Fort Collins, CO, USA	Mesic Aridic Haplustalfs	Corn-growing season	1.71	1.14	Halvorson et al. (2011)
Glencorse Mains, Penicuik	Gleysol with a clay loam texture	Year round	3.01	0.69	Clayton et al. (1997)
Ontario, Canada	Mesic Typic Argiaquoll	Corn-growing season	6.49	2.40	Drury et al. (2012)
Winnipeg, Canada	Osborne Orthic Black	Wheat-growing season	1.9	1.48	Burton et al. (2008a)
Mean			1.91	1.02	

(Burton et al. 2008b). Split N applications to crops result in reduced concentrations of soil mineral N in the early growth stage of crops. Application of the second portion of N during the active growth phase, when N uptake is at maximum, also reduces the potential for N_2O emissions to occur (Van Groenigen et al. 2010). Split application of N was reported as an effective strategy to reduce N_2O emissions from potato cultivation (Burton et al. 2008b). In corn production, a single application of N was reported to emit 35% more N_2O compared to split applications (Fernández et al. 2016). Split application of N fertilizer has also been suggested to reduce N_2O emissions for maize cultivation under normal rainfall patterns (Yan et al. 2001). In wheat and canola, the split application of N fertilizer with a second application in spring instead of a single application in fall could also be effective at reducing N_2O emissions (Hao et al. 2001). Therefore, split applications could be a potential mitigation strategy of N_2O emissions from croplands in the SGP. However, the production of winter wheat in the SGP is characterized as a low-input dryland system. Splitting N applications into two events requires additional use of machinery and labor, and can add to soil compaction and crop damage through additional field operations (Adams et al. 2018). Therefore, farmers in this region generally use single applications of N at planting.

Urea is the most common form of N fertilizer applied to winter wheat in the SGP. It is broadcast on the soil surface, and its incorporation into soil depends on precipitation, which is highly variable (Adams et al. 2018). Therefore, the risk of N loss from surface broadcasting as N_2O is higher compared to placing N fertilizer in the soil profile. A recent meta-analysis reported that placement of N fertilizer below 5 cm is an effective strategy for reducing N_2O emissions in no-till agroecosystems (Van Kessel et al. 2013) and would also be useful in systems of conventional tillage. The explanation for these results is that the potential for nitrification and denitrification decreased rapidly with depth (Venterea and Stanenas 2008). Therefore, the supply of inorganic N to communities of nitrifying and denitrifying microbes near the surface is decreased with the deeper placement of N fertilizer, resulting in lower N_2O emissions (Van Kessel et al. 2013). Several studies have reported reduced N_2O emissions with the subsurface application of N fertilizer (Omonode et al. 2011; Tenuta and Beauchamp 2000; Ussiri et al. 2009; Venterea et al. 2005). Therefore, subsurface placement of N fertilizer into the soil can be an effective approach to reduce N_2O emissions from SGP croplands.

5.4 Cover crops

Growing cover crops during fallow periods between cash crops could serve as a strategy to reduce GHG emissions and provide other ecosystem services that benefit the environment. Included are reducing wind and water erosion, reducing nitrate leaching, fixing atmosphere N, and improving sequestration of C (Blanco-Canqui et al. 2015; Tonitto et al. 2006). Some studies have cited the value of cover crops in mitigating climate change. Cover crops are capable of reducing GHG emissions, especially CO_2 and N_2O, by affecting C and N cycling (Kaye and Quemada 2017). The C cycle is impacted as root and shoot biomass produced by cover crops sequester C, which is stored as soil organic matter after the incorporation of crop residues into the soil. The reduced soil erosion by cover crops also reduces decomposition of soil C caused by the water transport (Berhe et al. 2007). A meta-analysis using data from 37 different sites reported sequestration rates of 32 ± 8 g C m^{-2} $year^{-1}$ with cover crops compared to control, which is equivalent to mitigating 117 ± 29 g CO_2 m^{-2} $year^{-1}$ (Poeplau and Don 2015).

The effect of cover crops on mitigating N_2O, the most-potent GHG, is still debatable. Emissions of N_2O are dependent on available soil mineral N, soil water content, available electron donors (C), and the physical properties of soil (Basche et al. 2014). Fluxes in agricultural N_2O generally result from denitrification of nitrate, which occurs under saturated soil conditions. It is assumed the conditions for N_2O production would be less conducive as cover crops take up nitrate and soil water when growing (Tribouillois et al. 2016). However, incorporation of legume-based cover crops at maturity would lead to higher C (electron donor) inputs, and mulching effects of cover crops may stimulate saturated conditions, thus enhancing denitrification and N_2O production (Mitchell et al. 2013).

A meta-analysis investigating the impact of cover crops on N_2O emissions reported that environmental and management factors, involving fertilizer N rate, soil incorporation, rainfall, and type of cover, (legume or non-legume) altered the impact of cover crops on N_2O emissions (Basche et al. 2014). The meta-analysis reported that the use of non-legumes with high C:N ratios as cover crops, and not incorporating the biomass into the soil, would have the greatest potential to mitigate N_2O emissions. This approach might have some potential in the SGP if the cover crop is used for other services than strictly as a cover. The aboveground biomass produced by a cover crop could be used as forage for beef production. Haying would reduce the amount of electron donors (C) input to the soil at the termination of the cover crop, and reduce N_2O emissions after incorporation. Although other studies reported slight increases in N_2O emissions after incorporation of cover crops, this increase could be compensated through increased C sequestration. An improvement in GHG balance of 315 kg CO_2 ha^{-1} year^{-1} was reported with cover crops compared to the bare soil (Basche et al. 2014).

5.5 Grazing management

Changing form of grazing management, and intensity of grazing pressure are among the few strategies available to reduce GHG emissions from native and tame perennial pastures. As discussed earlier, higher stocking rate applied to pastures leads to greater N_2O emissions from grazing lands, due to the effects on soil compaction and other physical, chemical, and biological properties of soils (Hamza et al. 2005). Such high rates also result in ruminants consuming greater amounts of low-quality forage, which affects both animal performance and GHG emissions (Liebig et al. 2010; Wang et al. 2015). Therefore, the management of stocking density (animal numbers ha^{-1} year^{-1}) applied to graze paddocks is an essential practice for mitigating N_2O emissions (Hamza et al. 2005). More-intensive forms of stocking are one of the possible reasons for increased N_2O emissions due to increased deposition of manure and urine. The N_2O emissions from these deposits are further supported under the anaerobic conditions caused by increased soil compaction in grazing paddocks (Núñez et al. 2007). A long-term evaluation of the effects of stocking methods on GHG emissions revealed that moderate stocking rates were most effective for net reductions in GHG emissions (Liebig et al. 2010). Another possible way to reduce N_2O emissions from grazing lands is to reduce dietary N and increase the mineral content of biomass available for grazing, as N excretion in urine is decreased upon reduced dietary intake of N (Dijkstra et al. 2013). Thus, regulating stocking rates and nutrient contents in grazed pastures could be useful recommendations for reducing N_2O emissions from grazing lands in the SGP.

Providing high-quality forage to grazing animals is a possible solution to mitigate CH_4 emissions by enteric fermentation from grazing lands. One of the methods for providing high-quality feed for grazing animal might be rotational stocking, in which one sub-paddock of a

Table 6 Comparison of N_2O (kg N_2O-N ha^{-1}) emissions with and without use of nitrification inhibitors (NI)

Location	Soil taxonomy or texture	Duration of emission measurement	NI Type (fertilizer used)	With NI	Without NI	Source
Christchurch, New Zealand	Udic Haplustepts	Year long	DCD (urine)	1.74	3.61	Dai et al. (2013)
Gleadthorpe, Central England	Sandy loam	Year long	DCD (urea)	0.27	0.52	Misselbrook et al. (2014)
Hamilton, New Zealand	Typic Hapludand	70 days growing season	Agrotain + DCD (urea)	0.50	1.10	Zaman et al. (2008)
Henan Province, China	Aquic inceptisol	Whole growing season	NBPT + DCD (urea)	0.41	0.77	Ding et al. (2011)
Madrid, Central Spain	Calcic Haploxerepts	Corn-growing season	NBPT (urea)	0.72	1.59	Sanz-Cobena et al. (2012)
Newark, North England	Eutric Gleysols	Year long	DCD (ammonium nitrate)	0.82	1.47	Misselbrook et al. (2014)
New Delhi, India	Typic Ustochrept	95 days	DCD (urea)	1.09	1.43	Majumdar et al. (2002)
New Delhi, India	Typic Ustochrept	120 days	Thiosulphate (urea)	0.50	0.76	Malla et al. (2005)
Palmerston North New Zealand	Typic Fragiaqualf	Spring season	Agrotain + DCD (cow urine)	1.50	1.10	Zaman et al. (2009)
Waikato, New Zealand	Typic Udivitrand	60 days	DCD (urine)	0.31	1.01	Di et al. (2007)
Mean				0.78	1.33	

larger group of paddocks is grazed at a time, and the remaining paddocks are allowed to recover and produce improved quality forage (Teague et al. 2013). A study evaluating different grazing strategies for mitigating GHG emissions in the SGP suggested rotational grazing as a viable option, while continuous, growing season-long grazing with lighter stocking rates could also be useful (Wang et al. 2015). Managing the stocking rate applied to pastures is a more important strategy for mitigating enteric CH_4 emissions from grazing lands, as it translates to CH_4 reductions across all stocking methods. A study evaluating the effect of stocking rates on GHG mitigation concluded that enteric CH_4 emissions were three times higher in heavily grazed pastures than moderately grazed (Liebig et al. 2010). The response was likely due to the consumption of lower quality feed on heavily grazed pastures, which limited the capacity of cattle to select higher quality forage when grazing. Therefore, adopting management that increases the availability of high-quality forage with moderate stocking rates are important GHG mitigation strategies for grazing lands of the SGP. One hypothesized effect for systems of rotational stocking has been a more uniform distribution of the paddock use by cattle, and hence the distribution of excreta (Barnes et al. 2008; Briske et al. 2008), which would prevent hotspots in N deposition. However, the inherent behavior and preferences of cattle for certain features of landscapes (for example, shade, water, even topography) may prevent the achievement of uniform distribution of paddock use, regardless of the stocking method (Arnold and Dudzinski 1978; Northup et al. 2019).

5.6 Use of nitrification inhibitors in N_2O hotspots

Inhibiting nitrification from N hotspots in grazing lands is a potentially useful strategy to mitigate N_2O emissions, though they have not been tested to any degree within the SGP. The largest proportion of total N_2O emissions from grazing lands is contributed by N hotspots which include urine patches, dung pats, shaded areas, and areas near to water troughs (Chadwick et al. 2018). A study determining spatial variability and N_2O hotspots in grazing lands revealed that these areas constitute roughly 1.1 % of the total area of pastures can contribute 55% of the total daily N_2O emissions from paddocks (Cowan et al. 2015). The primary reason for significant emissions from these hotspots is their enrichment with nutrients, especially N, and soil moisture by cow urine and dung, which provides conditions favorable for N_2O and CH_4 emissions (Flessa et al. 1996).

Numerous other mitigation strategies for reducing N_2O emissions have also been recommended, including restricted grazing during wet periods that favor denitrification (Bhandral et al. 2007); feeding cattle low-N diets, using stand-off pads (Luo et al. 2008), application of soil amendments (i.e., lime) to increase soil pH to shift the balance between N_2O and non-greenhouse N_2 (Šimek and Cooper 2002); or use of zeolite to capture soil NH_4 (Zaman and Nguyen 2010). The peer-reviewed studies were selected from different regions, which quantified N_2O emissions from the treatments using nitrification inhibitors in N_2O hotspots or with synthetic fertilizers in grazing lands as compared to treatments with no nitrification inhibitor use. Synthesis of 10 studies showed that N_2O emissions from treatments including nitrification inhibitors were significantly ($p < 0.01$) lower (0.78 kg N_2O-N ha^{-1}) than from treatments without inhibitors (1.33 kg N_2O-N ha^{-1}) (Table 6).

Among all the abovementioned strategies, the blanket application of nitrification inhibitors like dicyandiamide (DCD) in combination with urease inhibitors like nBTPT has been recommended as the best approach to reduce N losses from grazing lands (Zaman and Nguyen 2012). Another study reported that DCD was most effective in reducing N_2O

emissions from cattle urine, with 70% reductions in emissions recorded (Misselbrook et al. 2014). However, there are limitations of using DCD as nitrification inhibitors on hotspots in grazing lands. Some important drawbacks are short-term effectiveness at temperatures above 20 °C (Singh et al. 2008), a common feature of climate in the SGP during late-spring through summer and leaching into waterways due to water solubility (Zaman and Nguyen 2012). It was also reported that the form of DCD applied and the time of application also impact its effectiveness. Application of granular DCD 5–7 days before grazing was more effective in reducing N_2O emissions than spraying liquid DCD after urine deposition, as sprayed DCD is deposited on leaves, while the granular is deposited on soil (Zaman and Nguyen 2012). Therefore, future investigations on timing and type of nitrification inhibitor applied are required in the environmental conditions of SGP to determine if GHG emissions from hot spots can be controlled. An examination of the costs required to apply inhibitors to production-scale paddocks will also need to be addressed.

6 Conclusions and recommendations for future research

This review focused on the current management practices applied to agronomic and grazing lands by producers in the SGP and their capacity to influence GHG emissions, and management practices that might help mitigate GHG emissions without negatively impacting agroecosystem productivity and soil condition. As such, this review of GHG emissions and potential mitigation techniques for agricultural lands of the SGP represents one variable-and complex-segment of the global issue of human-generated GHG emissions. Many of the factors discussed here are parts of diverse agro-ecosystems characterized by ranges of interactions between land types, applied management, soil conditions, and climate. While the variability that exists in SGP agroecosystems presents challenges for identifying sources of GHGs and mitigation techniques, there is some potential for the discussed issues and concepts to translate to other regions around the globe that encounter similar environmental conditions.

Among the mitigation techniques considered, forms of management applied to croplands planted to wheat that reduce the amount of area summer fallowed could be effective, as would the use of conservation tillage. Combining conservation tillage with nitrate or ammonium salts as fertilizers that are incorporated into the soil, rather than use of urea broadcast on the soil surface, could be an effective combined strategy to reduce N_2O and CO_2 emissions, particularly if the fertilizers were treated with nitrification inhibitors. Modifying stocking methods applied to annual or perennial grasslands, or reducing stocking density, to increase the opportunity for grazing animals to consume higher quality forage could reduce CH_4 emissions. Research is required to examine how such mitigation techniques affect emissions of GHGs from agricultural lands.

As with mitigation techniques, there is limited information related to GHG emissions for agricultural lands in the SGP. Questions related to GHG emissions from crop and grazing lands under different forms of management must be addressed, to help identify mitigation techniques that are efficient at reducing emissions from agroecosystems. For instance, do management practices like growing green sources of N to support cash crops reduce GHG emissions compared to inorganic fertilizers? Further, can legume-based green N be managed to synchronize the N provided with the needs and uptake by the following crop? Also, for effective regional-scale mitigation of GHGs, integration of emissions and mitigation techniques from other sectors (forestry, municipal, energy enterprises, manufacturing) will be required. One particular issue related to the effects of land management in the SGP is the time period required for applied techniques to have an effect on the soil-plant-atmosphere interface. Longer-term experiments (> 10 years) are required to quantify the effects of climate on such factors as C sequestration, and must be combined with shorter-term studies on time-sensitive, process-oriented responses related to nutrient cycling and GHG emissions. Underscores the need for research to answer such process-level questions to make agriculture in the SGP more productive, environmentally sustainable, and profitable in meeting the growing demands of humanity for foods.

Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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