DYNAMICS OF CARBON DIOXIDE, METHANE AND NITROUS OXIDE FLUXES IN PLANTED SHELTERBELTS AND ADJACENT CROPPED FIELDS

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Chukwudi Christopher Amadi

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ABSTRACT

For more than a century, over 600 million shelterbelt trees have been distributed to land owners in the Canadian Prairies mainly to protect farms from soil erosion and extreme wind events. In Saskatchewan, there exists over 60,000 km of planted shelterbelts; however, there is a lack of data quantifying the role of shelterbelts in mitigating greenhouse gas (GHG) emissions in agricultural landscapes. These limited estimates of carbon (C) sequestration and GHG mitigation potential for shelterbelts are needed for regional C budgets and GHG inventories. The objective of this research was to quantify the role of shelterbelts on the mitigation of CO_2, CH_4 and N_2O in cultivated fields. Chamber-based GHG monitoring and modeling approaches were employed. Nitrous oxide emissions were lower in shelterbelts (0.65 kg N_2O-N ha^{-1} yr^{-1}) than in cultivated soils (2.5 kg N_2O-N ha^{-1} yr^{-1}), attributed to the capability of deep rooting trees to remove excess available N and soil water. Both shelterbelt and cultivated soils were small sinks for CH_4, though the sink potential was 3.5-times greater for the shelterbelt soils. Soil-derived CO_2 emissions were greater in the shelterbelts (4.1 Mg CO_2-C ha^{-1} yr^{-1}) than in the adjacent fields (2.1 Mg CO_2-C ha^{-1} yr^{-1}). Nevertheless, cumulative emissions of non-CO_2 GHGs was reduced by 0.55 Mg CO_2e ha^{-1} yr^{-1} in the shelterbelts and soil C storage (0–30 cm soil depth) was 27% greater, representing an increase of 28 Mg ha^{-1} in the shelterbelts than in the cropped fields, attributed to long-term inputs from tree litter. Holos model simulations of GHG fluxes in a cereal-pulse rotation indicated that a shelterbelt planting occupying 5% of the farmland resulted in total farm emissions being reduced by 8.2 – 23% during a 60-year period, depending on selected tree species. Between 90 – 95% of GHG mitigation by shelterbelts was through C sequestration in tree biomass and in stable SOC pools, while the reduction in N_2O emissions and increased oxidation of soil CH_4 totalled 5.1 – 9.8% of the overall GHG mitigation by shelterbelts. Faster growing trees (e.g. hybrid poplar) were more effective in accumulating C in tree biomass and soil and in mitigating soil GHG emissions. This study provides evidence that farm shelterbelts function as net biological sinks of CO_2 and can play a role in mitigating soil-derived GHG emissions in agricultural landscapes.

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DEDICATION

This dissertation is dedicated to the people who made me the person I am today, supported me and believed in me even when I was not sure of what I was doing . . . my parents, Jonathan and Bertha Amadi; my sisters, Precious, Miracle and Chioma; and my brother, Chibuike. Thanks loads!

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1. GENERAL INTRODUCTION

Agricultural practices such as the manufacture and application of high rates of N fertilizers as well as conventional tillage are some of the main contributors to the increased concentrations of carbon dioxide (CO_2) and other greenhouse gases (GHGs), including methane (CH_4) and nitrous oxide (N_2O), in the atmosphere (Paustian et al., 2000; Robertson and Grace, 2003). With an increasing awareness of global climate change and the world population expected to exceed 9 billion by 2015 (Alexandratos and Bruinsma, 2012), agricultural producers face the challenge of reducing GHG emissions from farming while maintaining food production for a growing human population (Evers et al., 2010). Forest and agroforestry sinks have been promoted as a sustainable strategy for mitigating GHG emissions from the agricultural sector. Consequently, there has been increasing global interest in adapting agroforestry systems, including shelterbelts for long-term carbon sequestration and stabilization of GHG emissions (Kort and Turnock, 1999).

Shelterbelts (also known as windbreaks) are linear arrays of trees and shrubs planted to alter environmental conditions in agricultural systems while providing a variety of economic, social and ecological benefits valued by land owners (Mize et al., 2008)—including C sequestration (Kort and Turnock, 1999). Benefits associated with shelterbelt use have been extensively reported in various parts of the world (Kort, 1988; Nair, 1993; Cleugh and Hughes, 2002; Luis and Bloomberg, 2002; Brandle et al., 2004; Zhang et al., 2009). Particularly in Saskatchewan Canada, shelterbelts may be found in the form of farmyard and field shelterbelts, forest belts, eco-buffers and/or natural shelterbelts composed of various species of drought hardy, coniferous and deciduous trees (Amichev et al., 2012). The major benefits of shelterbelts to adjacent agricultural fields are a reduction in wind erosion and enhancement of local microclimates conducive for crop production (Kort, 1988; Brandle et al., 2004). During the last two decades however, shelterbelts have been recognized for their potential in offsetting increasing atmospheric CO_2 concentrations through the storage of C in woody biomass and soil pools (Sauer et al., 2007; Kort and Turnock, 1999; de Brauw, 2006). Tree plantings have also been reported to mitigate nitrous oxide (N_2O) emissions and enhance methane oxidation in agricultural fields (Mutuo et al., 2005; Thevathasan and Gordon 2004).

Shelterbelts exert a measurable effect on soil properties in adjacent cropped fields (Kowalchuk and de Jong, 1995) and, as such, affect the production and consumption of soil-derived GHGs. Changes in soil properties following shelterbelt establishment are a result of modification of local soil microclimate and addition of soil organic matter (SOM) through litter fall and tree root distribution in adjacent soils (Martius et al., 2004; Peichl et al., 2006; Sauer et al., 2007). In general, the leeside of a shelterbelt is characterized by increased springtime soil moisture and relative humidity, along with reduced evaporation and night-time air temperatures (Rosenberg, 1974; Wang et al., 2010; Kort et al., 2011). At the same time, tree roots continuously extract soil moisture from an area 1.5- to 2-times the height of the shelterbelt (i.e., 1.5H–2H) (Kowalchuk and de Jong, 1995).

Carbon sequestration in shelterbelts, like other agroforestry systems occur mainly as increased C storage in the above- and belowground biomass of trees and by the stabilization of SOC contained in the lignin-rich litter provided by trees (Montagnini and Nair, 2004; Peichl et al., 2006). Hence, soils under shelterbelts accumulate more C compared to those in adjacent fields. Sauer et al. (2007) reported greater soil organic C content in a 35-year-old shelterbelt in Nebraska, USA (40 Mg C ha^{-1}) compared to the adjacent cropped field (36 Mg C ha^{-1}). The tree litter in the shelterbelt system contained an additional 13 Mg C ha^{-1}. In a separate study, Bronick and Lal (2005) found that SOC in wooded soils was about twice that in the adjacent cropped field. Moreover, C sequestration in the biomass of commonly used shelterbelt trees between the ages of 33 – 54 years in Saskatchewan, Canada was estimated at 52 kg C $tree^{-1}$ to 263 kg C $tree^{-1}$ (Kort and Turnock, 1999).

Although a considerable number of studies have addressed C sequestration in shelterbelts, there are no available data detailing the role of shelterbelts on the mitigation of CO_2, CH_4 and N_2O emissions in adjacent cropped fields. Yet, this information is necessary if we are to develop accurate estimates of the C sequestration and GHG mitigation potential of shelterbelts for global C budgets and GHG inventories.

Evidence from other agroforestry trials suggest that the incorporation of trees into agricultural landscapes reduces soil N_2O emission and increases CH_4 oxidation (Evers et al., 2010; Kim et al., 2009; 2010). For example measurements of N_2O emissions from fast growing hybrid poplar tree-based intercropping (TBI) systems in Southern Quebec showed that emissions were three times greater in conventional monocropping systems (60 g N_2O-N ha^{-1} d^{-1}) than in TBI systems (17 g N_2O-N ha^{-1} d^{-1}) (Beaudette et al., 2010). The reduced N_2O emissions in the TBI system was attributed to (i) lower potential nitrification rates possibly due to better fertilizer N utilization, and (ii) reduced heterotrophic denitrification due to reduced soil moisture at various distances in the TBI plots.

Upland soils are an important sink for CH_4 contributing up to 15% of annual global CH_4 oxidation; however, agricultural practices reduce the capacity for CH_4 oxidation in upland soils (Le Mer and Roger, 2001). Agronomic practices such as tillage, use of pesticides and herbicides and the application N fertilizers have been reported to have various degrees of inhibitory effect on CH_4 uptake in arable lands (Arif et al., 1996; Mosier et al., 1997; Powlson et al., 1997; Hütsch, 2001). There is a dearth of data on the effect of shelterbelts on CH_4 oxidation in adjacent fields; however, studies have demonstrated significantly greater CH_4 oxidation in undisturbed forests and grassland soils than in similar soils converted to agriculture (Goulding et al., 1996; MacDonald et al., 1996; Prieme and Christensen, 1999; Robertson et al., 2000).

The Agriculture and Agri-Food Canada Prairie Shelterbelt Program has distributed over 600 million shelterbelt tree and shrub seedlings to thousands of landowners across the Prairies, since 1903 to reduce wind speed and protect crops (Wiseman et al., 2009). In Saskatchewan, there are an estimated >60,000 km of planted shelterbelts at various planting designs and comprised of different tree and shrub species (Amichev et al., 2014). Given the thousands of kilometers of planted shelterbelts throughout the Prairie regions (Brandle et al., 1992; Wiseman et al., 2009), the impact of shelterbelts on C-sequestration and mitigation of soil GHG may be of significant environmental importance. Data detailing the effect of shelterbelts on the dynamics of C sequestration and GHG fluxes will lead to a more accurate estimation of the environmental and economic benefits of shelterbelt establishment, which in turn will support policy and management decisions on shelterbelt systems.

The general objective of the research presented in this dissertation was to evaluate the influence of shelterbelts on C sequestration and soil-atmosphere exchange of CO_2, CH_4 and N_2O in the agricultural landscape. A combination of field experiments and modeling studies were used to achieve the following objectives:

i. Quantify soil-atmosphere exchange of CO_2, CH_4 and N_2O in shelterbelts compared to adjacent crop fields
ii. Assess the extent of the influence of shelterbelts on soil-atmosphere exchange of CO_2, CH_4 and N_2O along a gradient from a shelterbelt to the center of the adjacent agricultural field.
iii. Quantify and compare soil-atmosphere exchange of CO_2, CH_4 and N_2O from shelterbelts under elevated soil moisture and dry conditions.
iv. Conduct a whole-farm simulation of GHG mitigation by shelterbelts using the HOLOS model

This dissertation reports on the role of shelterbelts in mitigating GHG emissions in arable soils. The dissertation is organized in a manuscript-based format consisting of seven chapters. This chapter and the following one provide a general introduction and comprehensive literature review, respectively. The subsequent four chapters (3-6) report on specific research studies to address each of the objectives.

Chapter 1 is a general introduction to the dissertation and addresses the research questions, objectives and scope of the studies conducted.

Chapter 2 provides an in-depth review of the literature pertinent to the subject. It covers the background on the contribution of agriculture to atmospheric GHG emissions, and a detailed review of available studies on C sequestration and GHG mitigation by shelterbelts and other agroforestry systems.

Chapter 3 presents results of a two-year field study quantifying emissions of CO_2, CH_4 and N_2O in nine shelterbelts and their associated cropped fields at three locations within the Boreal Plains and Prairies Eco-zones of Saskatchewan Canada.

Chapter 4 covers the results of the two-year field study monitoring changes in soil CO_2, CH_4 and N_2O fluxes at increasing distances away from a 33-year-old, two-row hybrid poplar-caragana shelterbelt within the parkland region of Saskatchewan Canada.

Chapter 5 presents the whole-farm modelling studying examining the role of three selected shelterbelt tree species (i.e. hybrid poplar, white spruce and caragana) on GHG mitigation at the farm-scale using the HOLOS model within the Dark-brown soil zone.

Chapter 6 synthesizes and integrates the key findings of the individual research studies reported in this dissertation and concludes with some recommendations for future research work.

Appendix A. reports on the results of the two year field study at the Canada Saskatchewan Irrigation Diversification Center (CSIDC) quantifying fluxes of CO_2, CH_4 and N_2O from shelterbelts under irrigated and non-irrigated conditions.

2. LITERATURE REVIEW

2.1 Increasing Atmospheric GHGs – a global concern

The increasing level of atmospheric GHGs and its impact on global climate have become a topical issue that requires viable mitigation strategies (IPCC, 2013). Consequently, agricultural producers face the challenge of reducing GHG emissions from agronomic practices while maintaining food production needed for a growing human population (Evers et al., 2010). The major factor believed to be the cause of climate change is increasing global temperatures, resulting from increasing concentration of GHGs in the earth's atmosphere; a phenomenon referred to as the greenhouse effect (IPCC, 2007). The greenhouse effect is an important natural event regulating global temperatures, which in turn permits the existence of life on earth. Increasing the concentrations of GHGs in the atmosphere intensify this effect, causing increased global temperatures (Mondor and Tremblay, 2010). Average global temperatures have increased by approximately 0.75 °C over the past century; however, considering the current rates, average global temperatures may increase by 1.8-4.0 °C by the end of this century (IPCC, 2013).

The greatest contributors to the greenhouse effect, after water vapour are CO_2, CH_4 and N_2O. During the pre-industrial era, atmospheric concentrations of CO_2, CH_4 and N_2O were estimated at 280, 0.289 and 0.275 parts per million by volume (ppmv), respectively (Statistics Canada, 2000). In recent times, the concentration of these gases has increased to approximately 380, 1.8 and 0.32 ppmv for CO_2, CH_4 and N_2O, respectively and these values are predicted to reach 535-983 ppmv CO_2, 1.46-3.39 ppmv CH_4 and 0.36-0.46 ppmv N_2O by 2100 (IPCC, 2007). Long-term cycles of natural processes over thousands to millions of years have maintained a relatively constant concentration of CO_2, CH_4 and N_2O in the atmosphere (Johnson et al., 2007; Lemke et al., 2010). However, human activity has been increasing the atmospheric concentration of GHGs (Schneider, 1989; Smith et al., 2008). Since the amount of infrared radiation (heat) trapped increases with an increase in concentration of these GHGs, there is the general conclusion that the accumulation of GHGs in the atmosphere due to anthropogenic activities is having a measurable impact on global temperatures (IPCC, 2001).

Agricultural activities and land use changes are significant contributors to the atmospheric levels of CO_2, CH_4 and N_2O, accounting for about one third of the global anthropogenic impact (Cole, et al., 1997; Werner et al., 2006; Smith et al., 2008). The decomposition of organic matter following land clearing and burning of biomass, drainage, tillage and grazing have contributed more than 30% of the anthropogenic increase in CO_2 (Lemke et al., 2010). The oxidation and mineralization of soil organic matter due to human activities accelerated during the industrial era has had a significant contribution to increasing CO_2 concentration and global warming (Lal, 2004). Similarly, the increase in fertilizer use, livestock herd size, cropped and grazed area that followed the industrial revolution have been increasing CH_4 and N_2O emissions (Ruddiman and Thompson, 2001; Ruddiman, 2003). The current global estimate of atmospheric CH_4 is 350Tg CH_4, 67% of which could be attributed to agricultural activities such as farm manure handling systems, rice cultivation and enteric fermentation in ruminants. Estimate of current anthropogenic N_2O emissions is about 7 Tg N_2O-N, approximately 90% of which represent emissions from agriculture (IPCC, 2001).

Studies have investigated the potential impacts of climate change on natural and agricultural ecosystems. Some have focused on economic impacts on crop production (e.g., Smith and Tirpak, 1990; Easterling et al., 1993; Rosenzweig and Parry, 1994; Easterling et al., 2000; Hertel and Rosch, 2010); while others have evaluated the ecological impacts (e.g., Acevedo-Whitehouse and Duffus, 2009; Huntingford et al., 2009; Huntingford et al., 2005). Typically, climate change is characterized by an increase in global temperatures and sea levels as well as extremes in hydrological cycles. Depending on the geographical region, these features can cause more frequent and severe climate events such as drought, storms and floods (IPCC, 2007). Several climate models demonstrate that high latitude regions will likely show greatest changes in temperature, while mid-continental regions, such as the North American Great Plains, will probably experience the greatest alterations in precipitation (IPCC, 1990). However, total production of food, fiber and feed will need to greatly increase to feed and clothe the ever-increasing human population. Even so, agriculture is often vulnerable to losses caused by unfavourable weather or climate conditions. These losses may occur as a result of direct linkages with severe weather events or from less direct or more complex combinations of weather conditions. Increased frequency and intensity of agricultural losses due to climate related factors could contribute to widespread malnutrition, displacement of populations and disruption of economies (Pautian et al., 1998).

In effect, the increasing anthropogenic emissions of GHGs and its effects on climate change will exacerbate the current and future challenges facing agriculture. Instant response strategies to control these challenges may intensify the impacts of climate change. For example, increased adoption of irrigation agriculture may reduce water available for non-agricultural uses (Guertin et al., 1997). As the world's population continues to increase, the pressure on farmland, both from expansion of urban areas (United Nations, 2002) and from a need to produce more food and fiber (Gardner, 1996), will increase.

Given the economic and ecological challenges facing agricultural production due to increasing concentrations of GHGs in the atmosphere, there is a need for adequate identification of best management practices that provide social, ecological and economic benefits, while reducing atmospheric GHG emissions (Udawatta and Jose, 2011). The incorporation of trees into agricultural landscapes, i.e. agroforestry in the form of windbreaks or other tree-based systems have been recognized for their role in offsetting C and N losses from conventional agricultural systems, while providing other economic benefits (Thevathasan and Gordon, 2004; Mize et al., 2008; Udawatta and Jose, 2011).

2.2 Shelterbelts in Prairie Soils

Windbreaks, also commonly referred to as shelterbelts, have been used for centuries to reduce wind speed in several wind prone eco-regions. They are a valuable agroforestry system (Fig. 2.1) believed to provide a range of ecosystem services and environmental benefits, which are valued by landowners and society (Kort, 1988; Mize et al., 2008; Kulshreshtha and Kort, 2009). Creating shelterbelts is well recognised for controlling soil erosion (Wang and Takle, 1995; Brandle et al., 2004), protecting crops from wind damage (Sudmeyer et al., 2007), improving soil moisture via snow entrapment (Kort et al., 2011), preventing pesticide drift and flood risk (Balazy, 2002; Carroll et al., 2004) and mitigating livestock odour (Tyndall and Colletti, 2007; Tyndall, 2009).

Other benefits may include enhancement of wildlife habitat, biodiversity and water quality (Johnson and Beck, 1988; Correll, 1997; Alexander et al., 1998), suppression of invertebrate pests, reduction of household energy use, farmyard diversification and landscape beautification (Akbari, 2002; Tsitsilas et al., 2006; Mize et al., 2008; Gardner, 2009). Recent recognition of shelterbelt establishment as a strategy for mitigating GHG emissions under the Kyoto protocol has increased research interest on shelterbelts as a viable strategy for carbon sequestration and bioenergy (Schroeder, 1994; Johnson and Henderson, 1995; Kort and Turnock, 1999). Shelterbelts have been used extensively in various parts of the world: Canada (Kort, 1988), Australia (Cleugh and Hughes, 2002), North China (Zhang et al., 2009), USA (Brandle et al., 2004), South America (Luis and Bloomberg, 2002) and several developing countries (Nair, 1993). But the dynamics of CO_2, CH_4 and N_2O emission and C storage in shelterbelts has been less been studied, creating less incentive for landowners to establish or maintain shelterbelts for its economic and ecological benefits (Zhang et al., 2012).

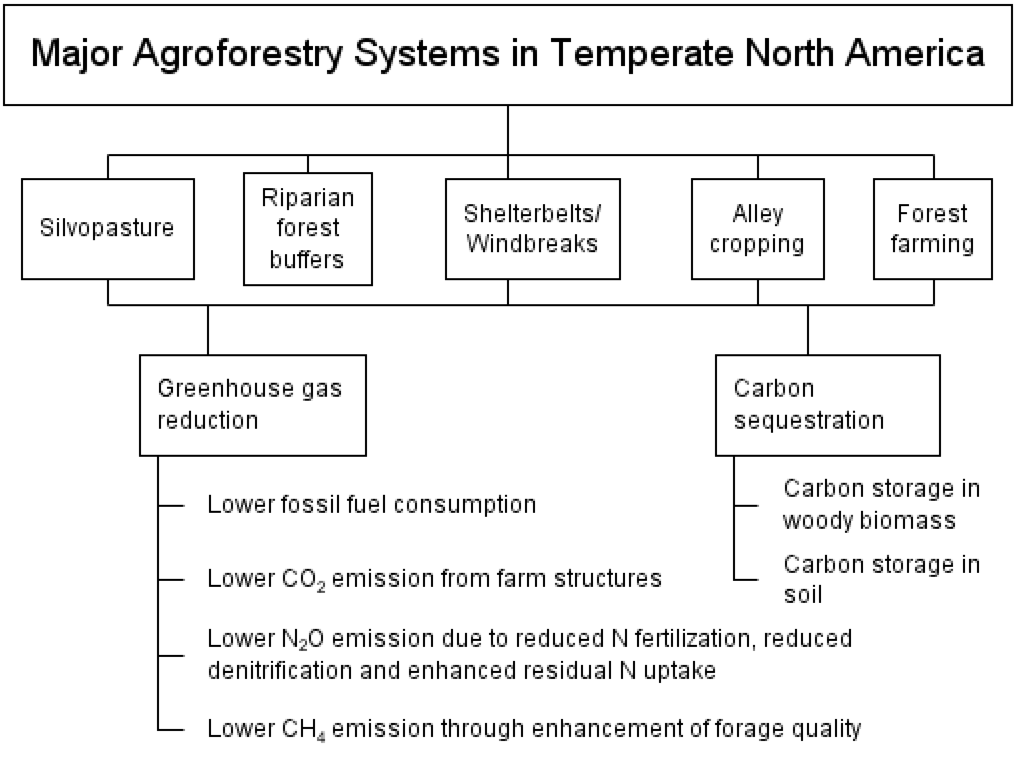

Fig. 2.1 Role of various agroforestry systems in mitigating climate change through C sequestration and reduction of agricultural GHGs

In the Canadian prairies, tree planting dates back to the 1870s, about 60 years before the expansive planting of shelterbelts to control severe soil drifting during the great depression (Kulshreshtha et al., 2010). The early tree-planting program was initiated by the Canadian government to encourage settlement in the west; as it thought that such tree plantings would enhance the landscape, increase rainfall and lead to a successful agriculture in the otherwise arid region (Watters, 2002). By 1901, the Canadian government began a co-operative tree-planting program, based on the understanding that the settlers needed the trees but did not have money to buy them. In 1902, the Forest Branch of the Department of the Interior began a program of distributing trees to farmers free of charge (Howe, 1986). At the start of the Canadian co-operative program, the adoption in 1901 was only 58,000 seedlings by 47 farmers; however, the number of tree seedlings distributed rapidly increased to 1.8 million by 1904 and 7.7 million by 1917 (Ross, 1923). These trees were planted primarily for reducing wind velocity, and to protect soils, crops and farmyards. The plantings may be found in the form of farmyard and field shelterbelts, forest belts, eco-buffers or natural shelterbelts composed of various species of drought hardy, coniferous and deciduous trees and shrubs (Howe, 1986; Kulshreshtha et al*.*, 2010).

Several tree species were introduced in the Canadian prairies during this period; most of them did not survive, mainly because they were non-hardy species (Ross 1923). This led to a continuous selection of adaptable species through massive field trials over time. Commonly used shelterbelts species within the prairie eco-zone include green ash (Fraxinus pennsylvanica), hybrid poplar (Populus spp), Manitoba maple (Acer negundo), caragana (Caragana arborescens), Siberian elm (Ulmus pumila), Scots pine (Pinus sylvestris), Colorado spruce (Picea pungens), white spruce (Picea glauca), buffalo berries (Sherphardia canadensis), choke cherry (Prunus virginiana), villosa lilac (Syringa vulgaris) and sea-buckthorn (Hippophae rhamnoids). One or more of these tree species were often selected and planted in rows to achieve a specific shelterbelt design, or for a definite purpose. For example, a combination of buffalo berry, choke cherry and Scots pine may be planted in a certain configuration for the provision of wildlife habitat, due to the additional food and warmth they provide. More so, a mixture of caragana and green ash may be preferred as field shelterbelts, for their efficient wind control, adequate snow distribution, long life span and nitrogen fixation.

Over the years, shelterbelt planting on the Canadian Prairies evolved in response to farmers' needs and concerns. Early settlers who arrived from Ukraine and Russia had a thorough understanding of the need to plant trees for crop and soil protection; and their knowledge contributed to the success of tree planting initiatives in the 1900s (Thevathasan et al., 2012). A combination of drought, soil erosion and depression events in the 1930s led to a great increase in high-density shelterbelt planting, through the Prairie Farm Rehabilitation Act (PFRA, 2000). A series of recurrent drought and severe soil erosion events in the 1960s and 1980s also led to the implementation of large-scale shelterbelt plantings programs.

In the 1990s, however, adoption of shelterbelt planting on the Prairies saw dramatic changes due to changing technical and socioeconomic conditions. Increases in farm size, use of large heavy equipment and farming on rented land modified the common erosion problem into that of climate change, biodiversity loss and reduced water quality. At the same time, there was a growing market for fruits and vegetables as well as an increasing demand for organic products. These led to the adoption of shelterbelts designed with trees that were capable of meeting these demands (Thevathasan et al., 2012).

Until now, shelterbelts are still being created or maintained on the Canadian Prairies for the same benefits they provided in the last few decades; however, the increased research on their effects, designs and limitations suggest that their use is becoming perhaps more targeted than in the past (Kulshrestha et al., 2010). There is a growing awareness that there may be opportunities for landowners to derive economic benefits from GHG reduction and carbon sequestration associated with their shelterbelt–planting activities. Although carbon credit trading markets are not well established in most parts of the world (Fischer et al., 1998), the Canadian domestic C offset system is currently in its final stages, and will include agricultural and forest-related carbon sinks (Kulshrestha et al., 2010). This could open a new opportunity for increased use of shelterbelts and shelterbelt planting activities.

Other tree-based systems that can contribute to C storage and GHG reduction include riparian buffers where linear rows of trees are grown in crop fields or around a riverbank for the purpose of reducing wind speed and preserving water quality (Fig. 2.1). Silvo-pastoral and alley cropping systems involve the integration of trees or shrubs with food crops and /or livestock on the same piece of land. Others may include forest farming, where high-value specialty crops such as mushrooms, decorative ferns or ginseng are grown under a modified forest canopy. Integration of trees in farms results in the production of food, fodder for livestock, timber for building, pollen for honeybees and wood for fuel (Nair et al., 2009). Simultaneously, tree based systems provide a broad array of ecosystem service functions including maintenance of soil fertility through organic matter inputs and nutrient cycling, biodiversity conservation, conservation of water quality and quantity through improved infiltration and reduced surface runoff, moderation of microclimate, carbon sequestration in biomass and soil, and reduction of agricultural GHGs (Stavi and Lal, 2013). The importance of incorporating trees into agricultural fields in terms of C storage and GHG mitigation is widely recognized; however, there is a scarcity of quantitative data on how tree based systems mitigates GHGs.

2.2.1 Zones in a sheltered farm

The establishment of shelterbelts in cropped fields result in the formation of distinct zones that may have an impact on crop yield and perhaps soil-derived GHGs. Wojewuda and Russel (2003) described three distinct zones occurring in sheltered fields, namely: the shelterbelt zone, transition zone (ecotone) and the cropped zone (Fig. 2.2). The shelterbelt zone is the area under the crown width of the linearly planted shelterbelts. Crown width values for mature shelterbelts in Saskatchewan could range from 4 to 15 m, depending on tree species and number of tree rows. The transition zone represents the area between the shelterbelt and the cropped field. It is the area that is indirectly influenced by shelterbelts, e.g., by shading, root activity, litter depositions and micro climatic influences. The transition zone varies with the height of the trees and extends 1.5 times the height of the shelterbelt (1.5H). The cropped field describes the area of the field that is not influenced by the shelterbelts or are under very limited influence.

Planted shelterbelts have been shown to influence variations in crop yield within the zones created be the established shelterbelts. Kort (1988) reported no crop yield within the shelterbelt area; a 50% reduction in yield due to root competition at distances from 0.5H to 1H; and a shelterbelt-induced increase in crop yield at distances from 1.5H to 15H, with the largest increase occurring between 1.5H to 3H. Soil concentrations of N and P followed a similar trend (Kowalchuk and de Jong, 1995). In addition to the influence of shelterbelts on crop yield and soil properties, Wojewuda and Russel (2003) reported a positive impact of shelterbelts on soil respiration and microbial abundance and diversity in the shelterbelt zone relative to the cropped field and transition area. However, there have been no data detailing the dynamics of GHG fluxes at various distances from a shelterbelt in order to uncover the role of shelterbelts in mitigating the emissions of CO_2, CH_4 and N_2O in cultivated soils. Results in Chapters 4 and 5 of this thesis provide data on GHG emissions within these zones created by shelterbelts.

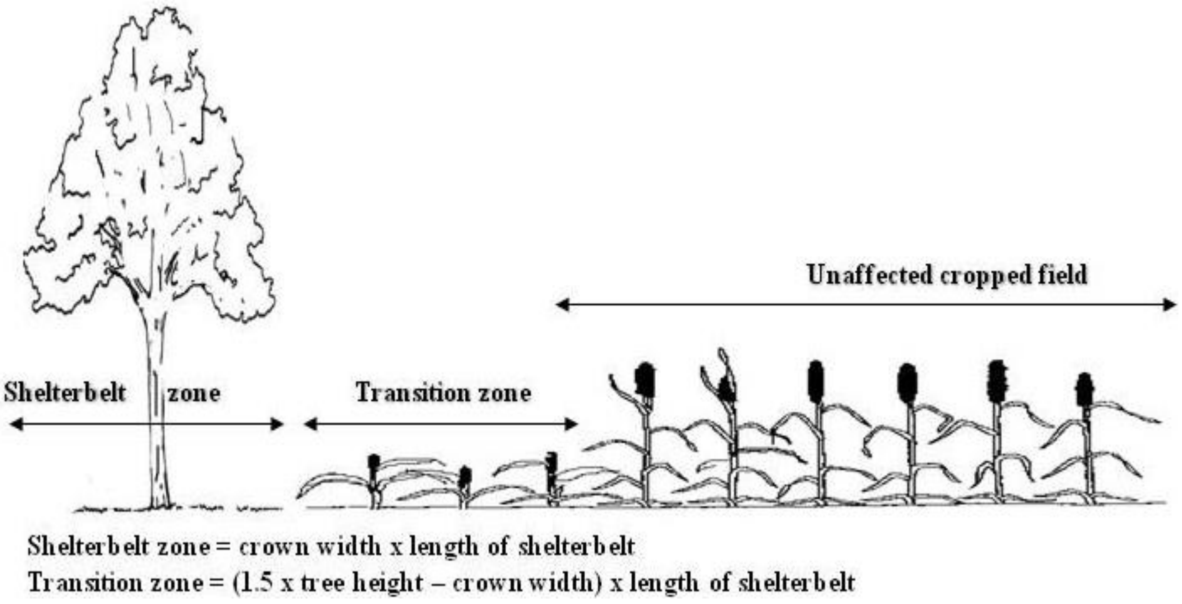

Fig. 2.2 Schematic showing the zones (i.e. shelterbelt, transition and unaffected cropped zones) within a sheltered field

2.3 Carbon Storage in Shelterbelts and other Tree Based Systems

Trees planted in agricultural landscapes are considered to be C sinks since the integration of trees into crop fields results in CO_2 sequestration from the atmosphere into tree components (Dixon, 1995; Sampson, 2001; Montagnini and Nair, 2004). The C sequestration potential of tree-based systems in the temperate latitudes has been estimated to range from 10 to 208 Mg C ha^{-1} (Schroeder, 1994; Kort and Turnock, 1999; Turnock, 2001). More specific studies have estimated carbon sequestration within shelterbelts based on a combination of information on the aboveground, time averaged carbon stocks (Table 2.1) and C storage in soil (Table 2.2). Brandle et al. (1992) estimated aboveground carbon storage of 20 year old, single row conifer, hardwood and shrub windbreaks to be 28, 26 and 1 kg C $tree^{-1}$, respectively. However, Kort and Turnock (1999) found that depending on species and age, aboveground carbon sequestration in Canadian Prairie shelterbelts ranged from 52 to 263 kg C $tree^{-1}$. The age of trees in this study ranged from 33 to 54 years, which resulted in greater biomass C sequestration compared to the study conducted on 20 year-old trees by Brandle et al. (1992).

Although agroforestry systems occupy small land areas, the potential to increase C sequestration of major agriculture dominated landscapes is greater with tree-based systems than monocropping, due mainly to the high rates of C sequestered per unit area (Nair and Nair, 2003; Nair et al., 2009; Schoeneberger, 2009; Morgan et al., 2010). For example, Thevasthasan and Gordon (2004) reported a four- to six-fold increase in the rate of C storage in Tree Based Intercropping systems (TBI) with hybrid poplar (2.4 Mg C ha^{-1} yr^{-1}) compared to a maize (*Zea mays* L.) monocrop field (0.4-0.6 Mg C ha^{-1} yr^{-1}). The greater C sequestration potential in TBI systems resulted in an approximately 1% increase in SOC at 0-15 cm depth over a period of 7 to 8 years, and the effect extended 4 m into the alley, thus amounting to 30-34% increase in SOC close to the tree rows. More recently, Udawatta and Jose (2011) estimated potential C sequestration rates for above- and belowground biomass components in North American agroforestry systems at 2.6, 3.4, 6.1, and 6.4 Mg C ha^{-1} yr^{-1} in riparian forest buffers, alley-cropping systems, silvopastures, and shelterbelts, respectively.

Carbon sequestration in tree-based systems occur mainly as increased C storage in the above- and belowground biomass of trees and by the increase and stabilization of SOC due to slower decomposition of lignin-rich litter provided by trees (Montagnini and Nair, 2004; Peichl et al., 2006). Carbon in woody biomass and soil characterize the bulk of easily observable additional C in tree-based systems (Montagnini and Nair, 2004; Schoeneberger, 2009). For example, Peichl et al. (2006) compared C storage in a 13 year old hybrid poplar and Norway spruce TBI system and in a barley monocrop conventional system. They reported that C sequestered in the woody biomass component at a planting density of 111 trees per hectare was 128 and 57 kg C tree^{-1} for the hybrid poplar and spruce trees, respectively, relative to the monocropping system where crops are harvested annually and residues are left to decompose. Furthermore, soil C sequestration was greater in the hybrid poplar TBI system (78.5Mg C ha^{-1}) compared to the conventional barley sole cropping system (65 Mg C ha^{-1}). In a separate study in eastern Nebraska, Sauer et al. (2007) reported significantly greater SOC in the 0-15 cm layer within a 35-year-old shelterbelt system (40 Mg C ha^{-1}) than in cultivated fields (36 Mg C ha^{-1}). Moreover, the tree litter in the shelterbelt system contained an additional ~13 Mg C ha^{-1}, representing a significant increase in the C sequestration potential of shelterbelts systems.

These studies suggest that incorporation of trees into cropped fields improves soil properties and can result in enhanced C storage in soil and tree biomass; however, the effects may vary greatly depending on species- and site-specific characteristics. For example, fast growing trees such as hybrid poplar sequestered more carbon in tree biomass (263 kg C tree^{-1}) after 33 years than other species with comparatively slower growth rates, such as green ash (79 kg C tree^{-1} after 53 years) or caragana (52 kg C tree^{-1} after 49 years) (Kort and Turnock, 1999).

The major reasons for an increase in C sequestration following tree planting have been discussed in Lorenz and Lal (2014). Tree-based systems may be associated with higher proportions of stabilized SOC in deeper mineral soil horizons as roots have the potential to grow extensively into deeper soil horizons (Nepstad et al., 1994; Jobbágy and Jackson, 2000). Root derived C contributes significantly to soil C pools and are more likely to be stabilized by physiochemical interactions with the soil matrix than shoot derived C (Rasse et al., 2005). The most significant contributions to soil C occur in the relatively stable fine textured soil fractions, i.e., lower than 53 μm diameter, where C is better protected through soil aggregation (Ingram and Fernandes, 2001). However, the capacity of soils to store C over time is finite and depends on soil biophysical properties (Paustian et al., 2000; Ingram and Fernandes, 2001).

Table 2.1 Carbon stocks in the biomass of trees in shelterbelts and various tree-based systems in temperate North America

Source	Location	Agroforestry system	Age (yr)	Tree species	C in tree biomass ($kg\ C\ tree^{-1}$)
Kort and Turnock (1999)	Saskatchewan, Canada	Shelterbelt	53	green ash	79
		Shelterbelt	33	Hybrid poplar	263
		Shelterbelt	54	white spruce	144
		Shelterbelt	49	caragana	52
Turnock (2001)	Western Canada	Shelterbelts	ND	Various spp	22 – 208†
Peichl et al. (2006)	Ontario, Canada	Tree Based Intercropping	13	Hybrid poplar	128
		Tree Based Intercropping	13	Norway spruce	57
Brandle et al. (1992)	USA	Shelterbelt	20	conifer	28
		Shelterbelt	20	hardwood	26
		Shelterbelt	20	shrubs	1
Schroeder (1994)	Temperate regions	Agroforestry	ND	Various spp	63†

† Data expressed as $Mg\ ha^{-1}$
ND Not determined

Table 2.2 Soil carbon stocks in shelterbelts and various tree-based systems compared to conventional cropping systems in temperate North America

Source	Location	Age (yr)	Landuse	SOC (Mg C ha^{-1})
Baah-Acheamfour et al (2014)	Alberta, Canada	20 - 50	Shelterbelts	47.7
		40 -100	Hedgerow	62.5
		41 -100	Silvo-pasture	81.3
Sauer et al. (2007)	Nebraska, USA	35	Eastern red cedar; Scotch pine (shelterbelt)	39.94
		ND	cropped field (conventional)	36.23
Peichl et al. (2006)	Ontario, Canada	13	Hybrid poplar (TBI)	78.5
		13	Norway spruce (TBI)	66
		ND	Cropped field (Barley)	65
Bambrick et al. (2010)	Quebec, Canada	8	TBI systems	77.1
		8	cropped field (conventional)	43.5
	Ontario, Canada	21	Hybrid poplar (TBI)	57
		21	Norway spruce (TBI)	51
		21	Cropped field (Barley)	51
Lee and Jose (2003)	Florida, USA	47	pecan/cotton (Alley cropping)	27.6
		47	pecan/orchard (Alley cropping)	43.01
		3	pecan/cotton (Alley cropping)	17
		3	pecan/orchard (Alley cropping)	16
		ND	cotton (conventional)	19.1

ND Not determined

Further, tree roots are more efficient in recovering nutrients such as N from below the crop rooting zone and capturing the resources available at the site for biomass growth. Through litter fall, the nutrients recovered below the crop-rooting zone are added back to crops to enhance plant biomass growth. The improvement in biomass growth in both trees and crops results in an increase in soil C pools (Allen, 2004; van Noordwijk et al., 1996). Shelterbelts are important in protecting surface soils of intensively managed agricultural lands from erosion and this reduces soil C losses (Albrecht and Kandji, 2003). Trees modify microclimatic conditions such as soil temperature and moisture levels, which could affect the quality, and quantity of soil litter C inputs (Laganière et al., 2010). Further, changes in microclimatic conditions due to the incorporation of trees into agricultural landscapes may induce changes in the biodiversity and activities of soil microbial communities, which may enhance soil C storage. However, the processes and mechanisms of SOC storage due to changes in microbial communities in tree based systems have been less studied (Lorenz and Lal, 2014)

Laganière et al. (2010) outlined major factors contributing to improvement of SOC pools following afforestation on agricultural soils, which include previous land use, tree species, soil clay content, pre-planting disturbance, and, to a lesser extent, climate zone. Thus, in a given climate zone, more soil C are likely to be sequestered in soils with more than 33 % clay, and broadleaf tree species have a greater potential of increasing SOC pool, due to their higher root biomass relative to coniferous tree species (Laganière et al., 2010). In summary, soil organic matter levels can be increased when practices that improve soil productivity are adopted. For tree based systems in particular, higher soil C pools can be achieved by increasing the amount of biomass input to soil and promoting SOM stabilization while reducing losses through the process of SOM destabilization, biomass decomposition, leaching and erosion (Lorenz and Lal, 2014; Albrecht and Kandji, 2003).

The potential of atmospheric CO_2 reduction by agroforestry systems occurs not only through carbon accumulations in tree biomass and soil, but also through various indirect benefits associated with agroforestry. Such indirect benefits include enhanced storage of C in wood products, substituting wood for fossil fuel and reducing the need for forest clearing and fertilizer production (Schroeder 1994; Dixon 1995; Montagnini and Nair 2004). Moreover, increasing SOC in agroforestry systems is associated with improved soil C and nutrient concentrations, enhanced soil structure and moisture content and improved community of soil organisms (Thevathasan et al., 2012). These indirect benefits may increase the quantity of direct soil C storage by 2- to 15-fold (Kürsten and Burschel, 1993).

2.3.1 Uncertainties and challenges for enhancing C storage in tree based systems

The incorporation of trees into agricultural landscapes has been shown to create positive interactions that lead to enhanced productivity and C sequestration, (Nair et al., 2009). However, there are also many possible negative interactions. Climate factors such as precipitation and temperature could modify the dynamics of soil C sequestration following the inclusion of trees in agricultural landscapes. For example, in a study to investigate ecosystem C changes along a precipitation gradient due to woody plant invasion into grasslands, Jackson et al. (2002) found a negative relationship between precipitation and soil C changes, with the drier sites gaining, and the wetter sites losing SOC. As a result, although there was a net positive ecosystem C balance in drier soils, losses of SOC at wetter sites were substantial enough to offset increases in biomass C. Such studies imply that current land-based assessments may have led to an over-estimation of terrestrial C sinks (Jackson et al., 2002). Monitoring and predicting changes in soil C pools over time can be challenging as such changes occur at a slow rate.

Furthermore, although soil C models of varying complexity are available, there is little confidence in SOC model predictions as long-term tree based studies are rarely replicated (Jandl et al., 2014). However, given that the inaccuracies in land-based C assessment methods are compounded by uncertainty in the long term changes in global climate and land use, long term monitoring techniques in addition to modelling approaches are required for better assessment of changes in soil C pools following tree inclusion in agricultural landscapes. On-going research at Agriculture and Agri-Food Canada is improving the HOLOS model, which estimates whole-farm GHG emissions, to enable the estimation of agroforestry-based GHG mitigation (Little et al., 2010). Similarly, research at the Centre for Northern Agroforestry and Afforestation at the University of Saskatchewan is using the 3PG model to predict C sequestration in shelterbelts in the different Canadian prairie ecodistricts (Amichev et al., 2015). Given that tree root systems are generally deeper than sampling depths in most studies, long-term whole profile studies are required for accuracy in assessing SOC pool changes following tree planting (Shi et al., 2013).

Aside from climate influences, another factor that may affect SOC storage in tree-based systems is overall soil fertility and productivity of the site. Trees perform poorly on infertile soils, especially in dry or semi-arid areas. In such soils, positive changes in SOC following tree planting are unlikely to be realized if there is insufficient biomass production (Lorenz and Lal, 2014). In addition, other biotic and abiotic factors such as tree pests and diseases, bush fires and drought may contribute to poor biomass production in tree based systems (Sileshi et al., 2007), which in turn affects soil C storage. Furthermore, many plants contain allelochemicals that are released into the soil through several mechanisms (Jose et al., 2004). Some allelochemicals can have an effect on germination, growth, development, distribution, and reproduction of a number of plant species (Inderjit and Malik, 2002); which may affect the overall productivity and soil C storage in tree-based systems.

Socioeconomic factors such as increasing farm size followed by acquisition and use of larger farm machines and equipment may discourage landowners from adopting tree-based systems such as shelterbelt systems as such trees are seen as disruptions to easy movement of farm machinery and other agronomic activities. Moreover, the tree–crop competition for water usually results in low crop yields, especially in dry or semi-arid environments, which further makes tree planting in croplands unattractive for dryland farmers.

2.4 Greenhouse Gas Emissions in Shelterbelts and other Treed Systems

If C sequestration in shelterbelts and other tree-based systems are well documented, this is not the case for major GHGs such as CO_2, N_2O and CH_4. Despite the enhanced shelterbelt plantings resulting from tree distribution programs in temperate North America, there are very few studies focusing on GHG emissions from this ecosystem. Some environmental and climatic conditions are unique to shelterbelts. Shelterbelts reduce wind velocity, which has a direct impact on soil-surface water evaporation during the summer and on snow management during the winter (Kort, 1988). Furthermore, the microclimate on the leeside is modified; there is increased soil moisture, relative humidity as well as reduced evaporation and night-time air temperatures (Rosenberg, 1974; Wang et al., 2010). Reduced night-time air temperature was shown as the most important of these factors as it reduced soil respiration (Rosenberg, 1974).

Campi et al. (2009) reported reduced evapo-transpiration in the Mediterranean region, although Grace (1977) reported no changes in total evapo-transpiration in the sheltered zone.

Stomatal resistance was reduced in the sheltered area, implying increased photosynthesis through an increased CO_2 diffusion rate (Rosenberg, 1974). Moreover, well-developed canopies protect soil microfauna from temperature and moisture stress (Martius et al., 2004), while the addition of litter and interception of organic matter-rich windblown sediments contribute to increased soil organic carbon (SOC) content in shelterbelts (Sauer et al., 2007). As well, trees are deep rooting and can inhibit the denitrification process in the soil profile by taking up residual N and excess soil water that would otherwise be available for N_2O emission or NO_3 leaching (Allen et al., 2004). These factors can affect net GHG fluxes in shelterbelt systems through their influence on soil microbial communities, root activity, and soil organic carbon (SOC) inputs and decomposition (Zhang et al., 2012). However, the key factors of CO_2, CH_4 and N_2O emission in shelterbelts have less been studied.

2.4.1 Nitrous oxide

Nitrous oxide emissions from agricultural fields represent a substantial contribution to the global anthropogenic GHG emissions. One major feature of conventional crop production systems is the widespread use of synthetic N fertilizers, which increases the amount of available mineral N and consequently N_2O emissions in such systems. Thus increased N_2O emissions have been reported in conventional crop production systems compared to native forests or grasslands (Corre et al., 1996; Izurralde et al., 2004). Agroforestry practices offer many opportunities to mitigate N_2O emissions in crop production systems in addition to maintaining soil productivity.

Firstly, incorporating trees into cropped fields can reduce N_2O emissions by excluding fertilizer N application on areas occupied by trees. Secondly, tree are deep rooted and can take up residual or excess N that would otherwise be available for N_2O emission and leaching on- or off-site, and eventually return them back to the soil through litter fall. This process is recognized as the safety-net role of tree roots (Allen et al., 2004), and the result is a more efficient N cycling, decreased fertilizer N demand by surrounding soils and, by implication, reduced N_2O emissions from N fertilization (Thevathasan et al., 2012). Thirdly, integrating trees into crop fields may reduce N_2O emissions from denitrification by taking up excess soil water. Trees have more demand for soil water than annual crops, which may cause increased soil oxidation potential and reduced denitrification rates around tree roots.

Measurements of N_2O emissions from fast growing hybrid poplar TBI systems in Southern Quebec showed significantly reduced soil moisture content in TBI systems than in monocropping systems (Table 2.3). Correspondingly, N_2O emissions were three times greater in monocropping systems (60 g N_2O-N ha^{-1} d^{-1}) than in TBI systems (17 g N_2O-N ha^{-1} d^{-1}) (Beaudette et al., 2010). The reduced N_2O emissions in the TBI system was attributed to (1) lower potential nitrification rates possibly due to better fertilizer N utilization, and (2) reduced heterotrophic denitrification due to reduced soil moisture at various distances in the TBI plots. In a separate study comparing N_2O emissions in 7 - 17 year-old re-established riparian forest buffers relative to cropped soils in Central Iowa, Kim et al. (2009) reported lower N_2O emissions in the forest buffers (2.8 - 11.0 kg N_2O-N ha^{-1} yr^{-1}) than in the cropped field (39.4 kg N_2O-N ha^{-1} yr^{-1}), attributed to reduced N-inputs to soils occupied by the forest buffer. Clearly, the incorporation of trees into cropping systems plays a defined role in decreasing soil N_2O emissions, and may be important in reducing the overall GHG emission footprint of agriculture. For example, several years of study on crop/tree intercropping systems with fast growing poplar in Southern Ontario showed that due to reduced fertilizer use and more efficient N-cycling, N_2O emission in the crop field was reduced by 0.7 kg ha^{-1} yr^{-1} (Thevathasan and Gordon, 2004).

2.4.2 Methane

Upland soils are an important sink for CH_4 contributing up to 15% of annual global CH_4 oxidation, which helps to balance global atmospheric methane concentrations (Powlson et al., 1997; Suwanwaree and Robertson, 2005). However, the size or strength of the sink is affected by land management (Table 2.4). For example, studies have demonstrated substantially greater methane consumption in undisturbed forests and grassland soils than in similar soils converted to agriculture (Goulding et al., 1996; MacDonald et al., 1996; Prieme and Christensen, 1999; Robertson et al., 2000). Indeed, the application N fertilization has been reported to significantly reduce methane oxidation in agricultural fields, especially when N is applied in the form of ammonium (Mosier et al., 1991; Hansen et al., 1993; Hütsch et al., 1993; Bronson and Mosier, 1994; Hütsch et al., 1994; Castro et al., 1994; Le Mer and Roger, 2001). Similarly, other agronomic practices such as tillage, use of pesticides and herbicides and fertilization have been shown to have various degrees of inhibitory effect on CH_4 uptake in arable lands (Arif et al., 1996; Mosier et al., 1997; Powlson et al., 1997; Hütsch, 2001).

The production of methane has been positively correlated with elevated soil moisture (Smith et al., 2003). However, the ability of tree roots to take up excess moisture and N in surrounding soils (Beaudette et al., 2010) creates favourable conditions for methane oxidation, which in turn, increases the size of methane sink in soils under treed systems. For example, Kim et al. (2010) measured methane fluxes in 7 to 17 year-old re-established riparian forest buffers relative to crop field soils in Central Iowa, where they reported greater CH_4 oxidation in riparian buffers compare to the cropped field. In addition they observed significant increments in soil C, total N, pH and a decrease in soil bulk density, all of which support CH_4 oxidation. Studies on methane fluxes in tree-based systems relative to conventional agricultural fields are limited. However, the limited data suggests that the integration of trees into cropped fields progressively modify soil properties to enhance CH_4 oxidation.

Table 2.3 Nitrous oxide exchange in various tree-based systems and cropped fields in temperate North America

Source	Location	Land use	N_2O flux ($g\ N_2O\text{-}N\ ha^{-1}\ d^{-1}$)
Kim et al. (2009)	Iowa, USA	Forest buffer	2.8 - 11.0
		cropped field	39.4†
Evers et al. (2010)	Ontario, Canada	Tree Based Iintercropping	7.5
		Cropped field	10.7
Beaudette et al. (2010)	Quebec, Canada	Tree Based Iintercropping	17
		cropped field	60
Peichl et al. (2010)	Southern Ontario, Canada	Temperate pine forest	-8 to 3.5
Mosier et al. (2006)	Northeastern Colorado, USA	Irrigated cropping system	-1.2 to 132
Matson et al. (2009)	Saskatchewan, Canada	Boreal forest	-5 to 7
Ellert and Janzen (2008)	Alberta, Canada	Irrigated cropping system	-1 to 72

Table 2.4 Soil methane exchange from various tree-based systems compared to conventional cropping systems

Source	Location	Land use	CH_4 (g CH_4-C ha^{-1} d-1)
Kim et al. (2009)	Iowa, USA	Forest buffer	-5.0 to 9.0
		grass filter	-2.0 to 10.0
		cropped field	-2.0
Goulding et al. (1996)	Rothamsted, UK.	Forest	0 to -2.4
		Grassland	0 to -1.9
		Cropped field	0 to -1.3
Dobbie and Smith (1994)	U.K	Forest	-8.0 to -2.4
		cropped field	-3.0 to -13.0
	Denmark	Forest	-4.0 to -7.0
		cropped field	-0.8 to -2.0
Mosier et al. (1991)	USA	Grassland	-3.5 to -8.4
		Cropped field	-0.3 to -2.0
Peichl et al. (2010)	Southern Ontario, Canada	temperate pine forest	-28 to 44
Mosier et al. (2006)	Northeastern Colorado, USA	Irrigated cropping system	-4 to 21.6
Matson et al. (2009)	Saskatchewan, Canada	Boreal forest	-28 to 15
Ellert and Janzen (2008)	Alberta, Canada	Irrigated cropping system	-3 to 2

Table 2.5 Soil respiration from various tree-based systems and conventional cropping systems across temperate North America

Source	Location	Age (yr)	Land use	Soil respiration (Mg C ha^{-1} yr-1)
Peichl et al. (2006)	Ontario, Canada	13	Hybrid poplar (TBI)	3.7
		13	Norway spruce (TBI)	4.5
		ND	Cropped field (Barley)	2.8
Lee and Jose (2003)	Florida, USA	47	pecan/cotton (Alley cropping)	5.1
		47	pecan/orchard (Alley cropping)	4
		3	pecan/cotton (Alley cropping)	3.5
		3	pecan/orchard (Alley cropping)	3.5
		ND	cotton (conventional)	4.2
Peichl et al. (2010)	Southern Ontario, Canada	4 - 67	temperate pine forest	2 to 50
Mosier et al. (2006)	Northeastern Colorado, USA	ND	Irrigated cropping sytem	0.3 to 77
Ellert and Janzen (2008)	Alberta, Canada	ND	Irrigated cropping sytem	0.2 to 48

ND Not determined

2.4.3 Carbon dioxide

Carbon dioxide fluxes resulting from soil respiration is an important component of the global C cycle and is an indicator of root respiration and soil biological activity (Raich and Schlesinger, 1992; Ewel et al., 1987; Tufekcioglu et al., 2001). Trees have the potential to improve soil fertility and quality by storing C and enhancing soil microbial activity, which in turn results in greater soil respiration. In a study of soil respiration and microbial biomass in a pecan – cotton alley cropping system in Southern US, Lee and Jose (2003) reported that microbial biomass C and soil respiration were approximately two-fold greater in old alley cropping systems than the monoculture system (Table 2.5). In a related study Peichl et al. (2006) reported greater soil respiration in 13 year Norway spruce and hybrid poplar intercropping systems (4.5 and 3.7 Mg C ha^{-1} yr^{-1}, respectively) than in a barley sole cropping system (2.8 Mg C ha^{-1} yr^{-1}), suggesting greater SOC content and greater root and microbial biomass C in the tree-based systems compared to the barley monocropping system.

Soil respiration in forested soils could be up to three times higher than in conventional agricultural lands (Swisher, 1991; Raich and Schlesinger, 1992; Adesina et al., 1999); and this highlights the substantial effect trees have on processes of soil respiration. But, emissions of CO_2 in tree-based systems are counter-balanced by the continual consumption and storage of atmospheric CO_2 by growing trees and plants. Whether the ecosystem is a source or sink of CO_2 will depend on the net difference between the amount of CO_2 taken up by plants through photosynthesis and the amount of CO_2 emitted from the soil surface back to the atmosphere (Paustian et al., 2000). Any disproportions between CO_2 uptake and CO_2 emissions is usually reflected in the total amount of soil organic carbon stored in the soil, highlighting the importance of size of organic matter pool in carbon cycling and CO_2 emission (Malhi et al., 2010).

2.5 Nitrogen-fixing Trees

Nitrogen-fixing trees are useful in improving the C and nutrient status of N-depleted soils. Leguminous trees used in agroforestry such as Caragana (*Caragana arborescens*) may derive up to 80% of their plant tissue N through fixation (Moukoumi et al., 2013). In a review of the value of leguminous trees such as *Faidherbia albida* incorporated into crop fields, Garrity et al. (2010) reported overall improvement in soil fertility, increased crop yield and increased carbon storage in tree biomass. Inclusion of leguminous trees in conventional cropping systems may facilitate biological N fixation, making N available for main crops, thereby reducing the need for N-fertilizers and reducing N_2O emissions associated with N-fertilizer application. Additional N supplement may be achieved through the incorporation of litterfall and prunnings (Thevathasan and Gordon, 2004).

Studies have shown that N inputs derived from leguminous trees can exceed crop N requirements, which may lead to N_2O emissions (Choudhary et al., 2002). Kandji et al. (2006) showed that compared to non N-fertilized fields, increased use of leguminous plants may result in net N_2O emission. Hence, although such N-fixing trees may be beneficial in terms of C sequestration, they may be significant sources of atmospheric N_2O emissions, which may constitute a greater environmental hazard. Kang et al. (1999) argued that N losses in tree-based systems are reduced since deep-rooted trees capture most of the residual or excess available N, making them unavailable for losses through denitrification or leaching; thus, mitigating N losses from N-fixing trees. This idea that tree roots play the role of a safety net to capture excess N and water that may otherwise result in N_2O emissions was further demonstrated by Allen (2004) who reported that tree roots could lower N losses in cropped fields by up to 53%. Clearly, the success of agroforestry in tackling climate change will depend on adequate understanding of trade-offs between C sequestration and the emission of trace gases such as CH_4 and N_2O. More research is therefore needed to elucidate the role of N-fixing tree species on GHG dynamics in tree-based systems.

2.6 Conclusion

Compared to intensive cropping systems, well-managed agroforestry practices in temperate soils can reduce N_2O and enhance C storage while maintaining a strong CH_4 sink. The potential for higher soil C pools in tree-based system can be achieved by increasing the amount of biomass input to soil and promoting SOM stabilization while reducing losses through the process of SOM destabilization, biomass decomposition, leaching and erosion. Studies on GHG emissions in shelterbelt systems are scant and only a few have focused on N_2O and CH_4 fluxes. For better understanding of the impact of shelterbelt trees on C and N fluxes, changes in C sequestration within tree-based systems should be considered alongside soil respiration, N_2O and CH_4 emissions. Although N-fixing trees have the potential of increasing soil C storage through enhanced biomass production, the observation that the inclusion of N-fixing trees may induce greater N_2O emission requires further study. The C sequestered by N-fixing trees in shelterbelt systems may not be enough to offset potential N_2O emissions that may result from extra mineral N produced by such trees.

Whereas the inclusion of trees into agricultural landscape has been long perceived as leading to an automatic increase of C stock, it is becoming increasingly clear that this does not happen in all conditions. Any event (e.g. extreme climate and soil conditions or anthropogenic influences) that affects biomass production or soil microbial activity could impact the rate of C sequestration in the stable SOC pool. Shelterbelt systems involve complex interaction of trees with crops; however, research into the dynamics of GHG emissions within tree-crop systems may provide relevant information needed for achieving a climate-friendly agriculture. Essentially, research is required in standardizing methodologies for estimating above- and belowground C in Agroforestry systems, to enhance reliability of data. This is will inform the development of long term monitoring and predictive models to assess the impact of tree based systems on C and N fluxes under changing climate situations.

3. SOIL – ATMOSPHERE EXCHANGE OF CARBON DIOXIDE, METHANE AND NITROUS OXIDE IN SHELTERBELTS COMPARED WITH ADJACENT CROPPED FIELDS[1]

3.1 Preface

The need to balance food production with environmental sustainability—including achieving real, sustainable reductions in greenhouse gas emissions—is a major issue facing humankind. Planted shelterbelts and other agroforestry systems have been recognized as viable management practices for mitigating the impact of agriculture on climate change. Until now, our knowledge on the role of planted shelterbelts in mitigating climate change was based solely on C sequestration in tree biomass and soils underneath shelterbelts. Indeed, there have been no studies providing relevant data on the ability of shelterbelts to mitigate GHG emissions in agricultural landscapes. Shelterbelts have unique characteristics that could affect the dynamics of soil-atmosphere GHG exchange including modified microclimate conditions, large tree root networks, and entrapment of weed blown sediments. However, the contribution of shelterbelts to soil GHG (N_2O, CO_2 and CH_4) is not well understood. The objective of this study was to quantify and compare GHG emissions from shelterbelts to those from adjacent cropped fields. The field study was carried out at three locations within the Boreal Plains and Prairie Eco zones of Saskatchewan, through chamber-based measurements of soil GHG flux and the monitoring of soil conditions.

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[1] This chapter was published, with minor formatting differences, as Amadi, C.C., K.C.J. Van Rees, R.E. Farrell. 2016. Soil – atmosphere exchange of carbon dioxide, methane and nitrous oxide in shelterbelts compared with adjacent cropped fields. Agric., Ecosyst. Environ. 223:123-134. Dr. Farrell contributed critical equipment, assited with establishing the schedule for data collection and analysis, funding and fruitful discussions. Prof. Van Rees' operating grants supported the field campaign and allowed the purchase of necessary supplies. In addition, Prof. Van Rees contributed field work expertise and expansion of the major ideas in this chapter in numerous discussions. I wrote the majority of the manuscript, prepared the figures and tables, completed data analysis and interpretation, and made the final editorial decisions regarding all text and graphics.

3.2 Abstract

Farm shelterbelts are used as a management tool to reduce erosion, conserve moisture, protect crops and buildings, and sequester carbon. Although carbon storage in shelterbelts has been well researched, there have been no measurements of soil GHG exchange in shelterbelts relative to cropped fields. The objective was to quantify, for the first time, soil CO_2, CH_4 and N_2O fluxes from shelterbelts and compare them to emissions from adjacent cropped fields to assess their potential for GHG mitigation. During 2013 and 2014, non-steady state vented chambers were used to monitor soil GHG fluxes from nine shelterbelts and their associated cropped fields at three locations within the Boreal plains and Prairies Eco-zones of Saskatchewan Canada. Mean cumulative CO_2 emissions from shelterbelt soils (4.1 Mg CO_2-C ha^{-1} yr^{-1}) were significantly greater than those from cropped fields (2.1 Mg CO_2-C ha^{-1} yr^{-1}). However, SOC storage (0–30 cm) was 27% greater—representing an increase of 28 Mg ha^{-1}—in the shelterbelts than in the cropped fields. Cumulative CH_4 oxidation was greater in shelterbelts (-0.66 kg CH_4-C ha^{-1} yr^{-1}) than adjacent cropped fields (-0.19 kg CH_4-C ha^{-1} yr^{-1}) and cropped soils emitted significantly greater quantities of N_2O (2.5 kg N_2O-N ha^{-1} yr^{-1}) than the shelterbelts (0.65 kg N_2O-N ha^{-1} yr^{-1}). Total seasonal exchange of non-CO_2 GHGs was reduced by 0.55 Mg CO_2e ha^{-1} yr^{-1} in shelterbelts as compared with cropped fields, 98% of which was soil-derived N_2O. Patterns of soil temperature, moisture and organic matter distribution beneath shelterbelts suggest a modification in soil micro-environment due to shelterbelt establishment and root activity that, in turn, may be responsible for the observed increase in soil CO_2 emissions and CH_4 oxidation. The results demonstrate that shelterbelts have substantial potential to mitigate GHGs by enhancing C storage and reducing N_2O emissions, while maintaining a strong CH_4 sink.

3.3 Introduction

Agricultural lands are being pressed to provide more environmental and economic services as a consequence of the increased global demand for food and other agricultural products. However, as a result of increases in fertilizer use, livestock herd size and tillage that followed the industrial revolution atmospheric concentrations of CO_2, CH_4 and N_2O have increased dramatically (Paustian et al., 2000; Ruddiman, 2003). Average global temperatures have increased by approximately 0.75 °C over the past century and it is likely that further increases in atmospheric GHG concentrations will result in additional increases in global temperatures (IPCC, 2013).

Promoting agroforestry systems has been recognized as a viable land-use alternative for mitigating the impact of agriculture on climate change (IPCC, 1995); and shelterbelts have been targeted as a strategy for biological C sequestration (Kort and Turnock, 1999). Agricultural lands therefore, present an opportunity for removing large amounts of atmospheric GHGs if trees are incorporated into farm systems (Nair, 2011). Shelterbelts (also known as windbreaks) are linear arrays of trees and shrubs planted to alter environmental conditions in agricultural systems while providing a variety of economic, social and ecological benefits valued by land owners (Mize et al., 2008)—including C sequestration (Kort and Turnock, 1999).

Despite occupying only a small land area in agricultural landscapes, shelterbelts can sequester large amounts of C on a per unit area basis. For example, above- and belowground components in shelterbelts can potentially sequester as much as 6.4 Mg C ha^{-1} yr^{-1} (Udawatta and Jose, 2011), though this rate may vary with the age and species of tree and with climate factors. Brandle et al. (1992) estimated C stored in aboveground biomass of 20-year old, single row conifers, hardwoods and shrubs at 9.14, 5.41 and 0.68 t km^{-1}, respectively. Shelterbelt establishment also has been demonstrated to increase soil C stocks compared to conventional cropping systems. Sauer et al. (2007) reported that soil C stocks were significantly greater (by 371 g m^{-2}) in a 35-year-old red cedar-Scots pine shelterbelt compared to cultivated fields. An additional 1,300 g C m^{-2} was contained in the shelterbelt tree litter layer. Therefore, in addition to reducing wind erosion and providing other environmental services (Wang and Takle, 1995; Brandle et al., 2004; Sudmeyer et al., 2007; Kort et al., 2011), shelterbelt systems offer a viable option for enhancing C quantities on marginal parts of agricultural landscapes (Nair, 2011).

While C sequestration in woody biomass (Kort and turnock, 1999) and soils beneath shelterbelts (Sauer et al., 2007) is synonymous to CO_2 mitigation (Pautian et al., 2000), little else is known about the influence of shelterbelts on agricultural GHGs. Yet this information is necessary for an accurate assessment of the potential mitigating effects of shelterbelts on agricultural GHG emissions, and in developing future GHG estimates and inventories from shelterbelt systems (Davis et al., 2012).

Biotic and abiotic factors unique to shelterbelts may affect GHG fluxes from soils. The microclimate in the woodland and on the lee side of shelterbelts is modified; e.g., air temperature and evapotranspiration are reduced, while humidity is increased (Campi et al., 2009; Zhang et al., 2012). Moreover, well-developed canopies protect soil microfauna from temperature and moisture stress (Martius et al., 2004), while the addition of litter and interception of organic matter-rich windblown sediments contribute to increased soil organic carbon (SOC) content in shelterbelts (Sauer et al., 2007). As well, trees are deep rooting and can inhibit the denitrification process in the soil profile by taking up residual N and excess soil water that would otherwise be available for N_2O emission or NO_3 leaching (Allen et al., 2004). These factors can affect net GHG fluxes in shelterbelt systems through their influence on soil microbial communities, root activity, and soil organic carbon (SOC) inputs and decomposition (Zhang et al., 2012).

The Agriculture and Agri-Food Canada Prairie Shelterbelt Program has distributed over 600 million shelterbelt trees across the Prairie Provinces since 1903 to reduce wind erosion and provide other environmental benefits (Wiseman et al., 2009). Similarly, shelterbelts have been planted extensively in the Great Plains of the US. However, there have been no studies quantifying GHG exchange in shelterbelts relative to cultivated fields to determine the full capacity of shelterbelts in mitigating agricultural GHG emissions. The objective of this study, therefore, was to quantify for the first time, soil-atmosphere exchange of CO_2, CH_4 and N_2O in shelterbelts compared to adjacent crop fields.

3.4 Materials and Methods

3.4.1 Study Area

The study area consisted of nine paired shelterbelts and adjacent cropped fields located at three sites in the Boreal Plain (Boreal Transition Ecoregion; Prince Albert, SK) and Prairie (Mixed Moist Grassland Ecoregion; Saskatoon and Outlook, SK) Ecozones of Saskatchewan, Canada (Table 3.1; Appendix B). To avoid root interactions and shading effects, the cropped field adjacent to each shelterbelt was sampled along a transect running perpendicular to the shelterbelt, with the first plot located at least 50 m from the shelterbelt. Basic site and management characteristics of the shelterbelt and cropped sites are presented in Tables 3.1–3.3.

Table 3.1 Site information for the three study sites located in Saskatchewan, Canada.

Parameter	**Sites**		
	Outlook	**Saskatoon**	**Prince Albert**
Ecozone	Prairie	Prairie	Boreal Plain
Ecoregion	Moist Mixed Grassland	Moist Mixed Grassland	Boreal Transition
Soil classification	Dark Brown Chernozem	Dark Brown Chernozem	Black Chernozem
Soil Texture	Sandy loam	Clay	Sandy loam
Latitude (ºN)	51.29°N	52.12°N	53.22°N
Longitude (ºW)	107.03°W	106.62°W	105.68°W
Elevation (m)	541.0	504.1	428.2
Mean annual air temp (ºC) – 2013†	15.5	15.3	14.5
Cumulative annual PPT (mm) – 2013†	181.0	200.5	332.6
Total irrigation (mm) – 2013	87.5	NA	NA
Mean annual air temp (ºC) – 2014†	12.9	12.5	11.4
Cumulative annual PPT (mm) – 2014†	328.2	310.2	466.6
Total irrigation (mm) – 2014	112.5	NA	NA

† Data from Environment Canada (2014). Note: average air temperature and total precipitation were calculated for the sampling period in each year (April – October).

NA = not applicable.

PPT = precipitation.

Outlook

The Outlook site is located at the Canada Saskatchewan Irrigation Diversification Centre (CSIDC). Average temperature and cumulative precipitation during the sampling period (April – October) were 12.5°C and 278 mm, respectively (based on 1981-2010 climate norms; Environment Canada, 2015). The soils are classified as Orthic Dark Brown Chernozems, a mix of Asquith and Bradwell Association, with moderately sandy loam textures. The site features well drained soils formed mainly in aeolian sands and loamy lacustrine materials on a slightly undulating topography.

The shelterbelt plots consisted of a one row shelterbelt consisting of Scots pine (*Pinus sylvestris* L.) (O-SB1); a one row mixed species shelterbelt consisting of green ash (*Fraxinus pennsylvanica* Marsh.) and caragana (*Caragana arborescens*) (O-SB2); and a single row of caragana (O-SB3). In 2013, the field sites were cropped to wheat (O-CF1) and soybean (*Glycine max* L. Merr.; O-CF2 and O-CF3). In 2014, the O-CF1 site was cropped to soybean while the O-CF2 and O-CF3 sites were cropped to wheat. Tillage operations occurred on May 27, 2013 and

May 16, 2014. The cropped field sites received a total of 87.5 and 112.5 mm of irrigation water during the 2013 and 2014 growing seasons, respectively.

Saskatoon

The Saskatoon site is located at the University of Saskatchewan Horticulture Field Research Station. Mean temperature and cumulative precipitation during the sampling period (April – October) were 12.4°C and 277 mm, respectively (based on 1981-2010 climate norms; Environment Canada, 2015). The soils are classified as Orthic to slightly Solonetzic Dark Brown Chernozems of the Sutherland Association, with moderately fine clay texture.

The shelterbelt plots included a two-row shelterbelt consisting of jack pine (*Pinus banksiana*) and caragana (S-SB1); a single row shelterbelt consisting of a mixture of jack pine and Chokecherry (*Prunus Virginiana*) (S-SB2); and a single row of caragana (S-SB3). In 2013, the adjacent fields (S-CF1, S-CF2, and S-CF3) were cropped to haskap (*Lonicera caerulea*)—also known as blue honeysuckle. There were no tillage operations in the cropped fields in 2013 as haskap is a perennial plant. However, gas chambers were transferred to an adjacent asparagus (*Asparagus officinalis*) field in 2014 due to a planned change in land use and interference with the gas chambers from the machinery used to weed the haskap. The asparagus field was established in 2002 and has since received yearly organic amendments plus 70 kg N ha^{-1} yr^{-1} inorganic nitrogen fertilizer. In 2014, tillage operations (site preparation) in the asparagus fields commenced on May 22nd. There were no irrigation activities at this site.

Prince Albert

The Prince Albert site is located at the Conservation Learning Centre (CLC) research farm—approximately 18 km south of the city of Prince Albert, SK, Canada. Mean temperature and cumulative precipitation during the sampling period (April–October) was 11.0°C and 348 mm, respectively (based on 1981-2010 climate norms; Environment Canada, 2015). The site is characterized by an undulating topography with remnant native upland areas and wetlands. The soils are classified as Orthic Black Chernozems, a mix of Hamlin and Blaine Lake association (Udic Boroll) of fine sandy loam texture, on a gently sloping topography (Soil classification working group, 1998). The soil and landscape features are characteristic of the northern agricultural region of Saskatchewan.

Table 3.2 Characteristics of shelterbelts at the three study sites in Saskatchewan, Canada.

Location	Site ID	Main species	Age (yr)	Tree rows	Length (m)	Shelterbelt orientation†	Planting space (m)	Mean DBH (cm)	Mean height (m)
Outlook	O-SB1	Scots pine	19	1	435	E - W	2.5	27.7	10.5
	O-SB2	Green ash and caragana	41	1	160	N - S	2.7	53.1	15.8
	O-SB3	Caragana	34	1	750	N - S	1.0	6.8	5.0
Saskatoon	S-SB1	Jack pine and caragana	40	2	90	N - S	3.5	29.2	9.0
	S-SB2	Mixed spp.	38	1	90	N - S	2.0	17.5	15.0
	S-SB3	Caragana	38	1	84	N - S	1.0	5.3	5.5
Prince Albert	P-SB1	Poplar and caragana	27	2	176	E - W	1.6	52.7	23.8
	P-SB2	White spruce	41	4	70	N - S	2.0	15.0	7.0
	P-SB3	Caragana	19	1	150	N – S	2.0	5.6	4.0

† E-W (East-West); N-S (North-South)

Table 3.3 Characteristics of cropped fields at the three study sites in Saskatchewan, Canada.

Location	Site ID	Crop		Fertilizer N (kg ha^{-1})	
		2013	2014	2013	2014
Outlook	O-CF1	Wheat	Soybean	50	0
	O-CF2	Soybean	Wheat	0	70
	O-CF3	Soybean	Wheat	0	70
Saskatoon	S-CF1	Haskap	Asparagus	0	70
	S-CF2	Haskap	Asparagus	0	70
	S-CF3	Haskap	Asparagus	0	70
Prince Albert	P-CF1	Corn	Wheat	50	100
	P-CF2	Barley	Common bean	40	100
	P-CF3	Barley	Common bean	40	100

The shelterbelts consisted of a two-row shelterbelt consisting of hybrid poplar (*Populus spp.*) and caragana (P-SB1); a three-row white spruce (*Picea glauca*) belt (P-SB2); and a single row of caragana (P-SB3). In 2013, the fields adjacent to the shelterbelts were cropped to corn (*Zea mays*; P-CF1) and barley (*Hordeum vulgare*; P-CF2 and P-CF3). In 2014, the fields were cropped to wheat (*Triticum aestivum*) in (P-CF1) and common bean (*Phaseolus vulgaris*; P-CF2 and P-CF3). Tillage in the cropped fields occurred on May 21, 2013 and June 1, 2014. There were no irrigation activities at this site.

3.4.2 Greenhouse gas sampling

At each site, the bases for four rectangular gas chambers were placed along a transect at the center of each of the three shelterbelts during the fall of 2012; chambers were spaced approximately 20-m apart. At the same time, four additional bases were placed in the field adjacent to each shelterbelt, at a spacing of approximately 20-m along a transect running perpendicular to the shelterbelt—with the first chamber located approximately 50-m from the center of the shelterbelt. The chamber bases were installed to a depth of 5-cm and were used to anchor the flux chambers when GHG samples were being collected. Gas samples were collected by attaching a flux chamber (22-cm wide × 45.5-cm long × 15-cm tall) to the base and withdrawing 20-mL gas samples as soon as the chamber was in place (t_0) and thereafter at 10-min intervals for a total of

30 min (t_{10}, t_{20}, t_{30}). Gas samples were collected using a 20-mL polypropylene syringe (Monoject^{TM}, Luer lock fitting) fitted with a 25-gauge needle; injected into pre-evacuated 12-mL Exetainer® vials (LabCo Inc., High Wycombe, UK) fitted with butyl rubber stoppers (Rochette and Bertrand, 2008); and returned to the Department of Soil Science at the University of Saskatchewan in Saskatoon for analysis. Exetainer vials were prepared by alternate evacuation and flushing of the vial headspace for air removal.

The gas chambers were made of 0.6-cm thick poly methyl methacrylate (PMMA) and had a headspace volume of 10 L and a surface area of 1000 cm^2. Upon deployment, close-cell polyolefin foam gaskets (1-cm thick × 1.2-cm wide) were secured to the underside of the chamber lids to seal against the top edge of chamber bases. Gas chambers were vented with a clear flexible vinyl tube (4.8 mm i.d.) attached through an elbow fitting to the cover (Hutchinson and Livingston, 2001). The sampling port consisted of a silicone septum (9.5-mm o.d.) secured by a nylon bolt with a lengthwise opening, which served as a syringe guide. Efforts were made to minimize disturbance of the soil and litter deposits both in the crop fields and under shelterbelts during chamber installation. Green vegetation within the chambers was removed before sampling and all chambers remained in place throughout the two-year sampling period except during tillage and seeding operations in the cropped fields.

Gas sampling started at spring thaw and continued through to soil freeze-up in the late fall (29/04/2013 to 09/10/2013 and 08/04/2014 to 15/10/2014). Gas sampling occurred twice per week from the start of the spring thaw till about 4 weeks after seeding; gas sampling continued once per week throughout the summer and then once every two weeks during the fall. For logistical reasons, gas sampling took place over a two day period, with the Outlook and Saskatoon sites sampled on one day and the Prince Albert site sampled the next day.

Immediately following collection of the gas samples, soil moisture and temperature measurements were taken from the area directly adjacent to the chambers and at a depth of 10 cm. Soil temperature was measured using a stem-style digital thermometer (Reed PS100, Brampton, ON) and soil moisture was measured using a digital soil moisture meter (HydroSense, Campbell Scientific Inc., Logan, UT). Daily precipitation and average temperatures were obtained from the Environment Canada meteorological station nearest to each site.

3.4.3 Gas and soil analyses

Concentrations of CO_2, CH_4 and N_2O were determined using a gas chromatograph (Bruker 450 GC, Bruker Biosciences Corporation, USA) equipped with a thermal conductivity detector (TCD) for CO_2, flame ionizer detector (FID) for CH_4 and an electron capture detector (ECD) for N_2O. Samples were introduced into the chromatograph using a CombiPAL auto-sampler (CTC Analytics AG, Switzerland). Data processing was performed using the Varian Star Chromatography Workstation (ver. 6.2) software. Daily fluxes were estimated by fitting linear or exponential regression equations to the concentration *vs.* time data using a modified Hutchinson-Mosier method (Hutchinson and Mosier, 1981; Pedersen et al., 2010), implemented as an add-on package for R (R Development Core Team, 2011). For each chamber the daily flux was calculated as the slope of the fitted regression at t_0. On days when the GHG flux was low (e.g., during the summer when soil moisture was limiting) it was often noted that the change in concentration during chamber deployment did not exceed the minimum detectable concentration difference (MDCD) (Yates et al., 2006); nevertheless, fluxes were taken as the slope of the linear regression at t_0 to minimize “left censoring” of the data (Ens, 2012). Cumulative fluxes were calculated using a trapezoidal area-under-the-curve (AUC) analysis of the gas production *vs.* time curves (Hintze, 2009; Engel et al., 2010).

In June 2014, four 3.2-cm (i.d.) soil cores were collected from within each shelterbelt and crop field, divided into 0–30 and 30–100 cm depth increments, and composited by depth. Additional soil sampling occurred in July 2013 and October 2014 to monitor changes in available soil N (i.e., NO_3-N plus NH_4-N) concentration. Separate samples were collected using a hand-held core sampler (5.4-cm i.d. × 3-cm long), weighed and dried at 105°C for 24 h to determine soil water content and bulk density. All remaining field-moist samples were air dried and ground with a rolling pin to break aggregates; visible roots were removed and a subsample of soil (~150 g) passed through a 2-mm sieve. A 20-g subsample of the air dry soil was then placed on ball grinder for 5 minutes to create a fine powder (< 250 µm) for Total N and organic C analysis.

Soil organic C was determined using a LECO C632 Carbon analyzer (Wang and Anderson, 1998), following a 12N HCl pretreatment to remove all inorganic C. Total N was determined by dry combustion using LECO TruMac CNS analyzer (Figueiredo, 2008). Total inorganic N (NO_3-N and NH_4-N were determined using 2.0*M* KCl extraction (Maynard et al., 2008) and analyzed colorimetrically (Technicon Autoanalyzer; Technicon Industrial Syatems, Tarrytown, NY, USA)

Soil pH in water (1:1 paste; Hendershot et al., 2008) was measured using a Beckman 50 pH Meter (Beckman Coulter, Fullerton, CA, USA). Soil particle size was determined using a modified pipette method (Indorante et al., 1990).

3.4.4 Statistical analysis

Differences in gas fluxes and soil properties from shelterbelts and crop fields were analyzed using PROC MIXED of SAS 9.4 (SAS Institute Inc, 2013). The ANOVA assumption of normality was assessed with a Shapiro-Wilk test using PROC UNIVARIATE. A log transformation was applied to data that did not conform to a normal distribution. The statistical model used to analyze measured parameters was:

$$Y_{ij} = \mu + T_i + R_j + \varepsilon_{ij};$$

where Y_{ij} is the dependent variable, μ is the overall mean, T_i is the fixed effect of treatment (i = 2; shelterbelt and cropped fields), R_j is the random effect of study sites (j = 3; Outlook, Saskatoon and Prince Albert), and ε_{ij} is the residual error. Within site differences in gas fluxes and soil properties in shelterbelts and cropped fields across 2013 and 2014 were analyzed by a two sample comparisons t-test using PROC TTEST. The Fisher Protected Least Significance Difference (LSD) test was used for treatment comparisons. Given the spatial variability associated with GHG measurements in cropped fields and shelterbelts, the risk of a type II error in the analysis was considered to be high, although the sample size was relatively large. Consequently, unless stated otherwise, a *P* value of 0.1 was used to assess significance.

3.5 Results

3.5.1 Environmental conditions during the study period

The 2013 and 2014 growing seasons were characterized by cold, wet conditions during the spring and early summer; in 2013, this was followed by hot, dry conditions in mid to late summer, and in 2014, by hot, wet periods in the late summer (Fig. 3.1–3.3). At all sites, mean air temperatures (April-October) was greater (19-24%) than the long term average in 2013, but were similar to the long-term (site-specific) averages in 2014. Cumulative seasonal (April-October) precipitation in 2013 was lower (4–35%) than the long-term average at all sites. Conversely, in 2014, cumulative seasonal precipitation was greater (11–25%) than the long-term average at all sites (Table 3.1). Prince Albert experienced the lowest temperatures and the highest precipitation in 2013 and 2014, whereas Outlook and Saskatoon had warmer conditions and lower precipitation in both years (Table 3.1).

Soil moisture levels in both shelterbelt and cropped sites were greatest during the spring snow melt period, typically from April to May and then gradually decreased from June to October in each year. At all sites, and in both years, soil moisture was significantly ($P < 0.05$) lower within the shelterbelts than in the cropped fields (Fig. 3.1-3.3). Soil temperature (0–10 cm) ranged from -1ºC to 25ºC during the study period and was greatest between June and August in both years (Fig. 3.1-3.3). Across sites, soil temperature was significantly ($P < 0.05$) higher in the cropped fields than in the shelterbelts. However, trends in the soil temperature data show that whereas soils in the cropped fields were warmer than those in the shelterbelts during the spring and early summer months, the opposite was true during the late summer/early fall (i.e., from late August/early September to the end of the growing season).

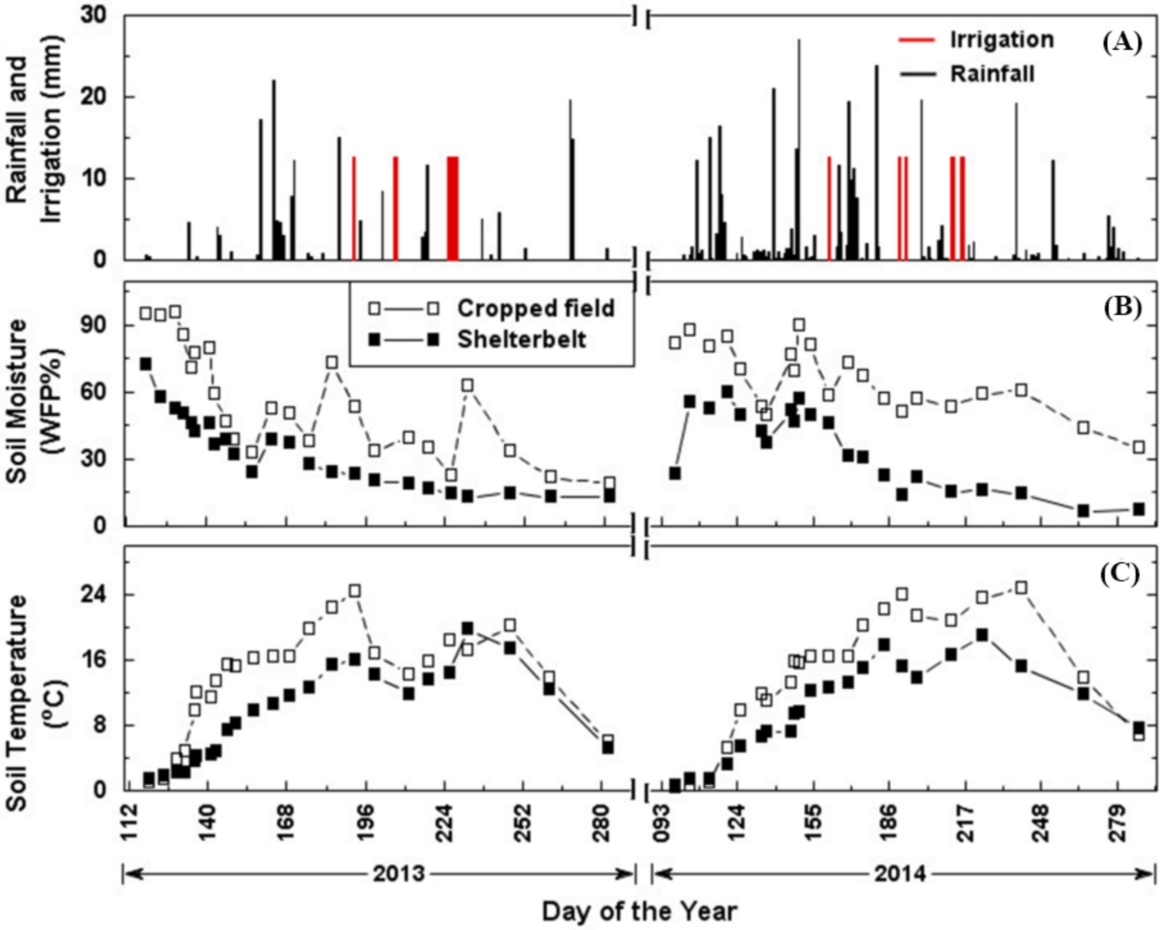

Fig. 3.1 Rainfall plus irrigation (A), soil water content (B) and soil temperature (C) measured at the Outlook during the 2013 and 2014 sampling seasons. Water filled pore space (WFPS; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both the shelterbelts and cropped fields.

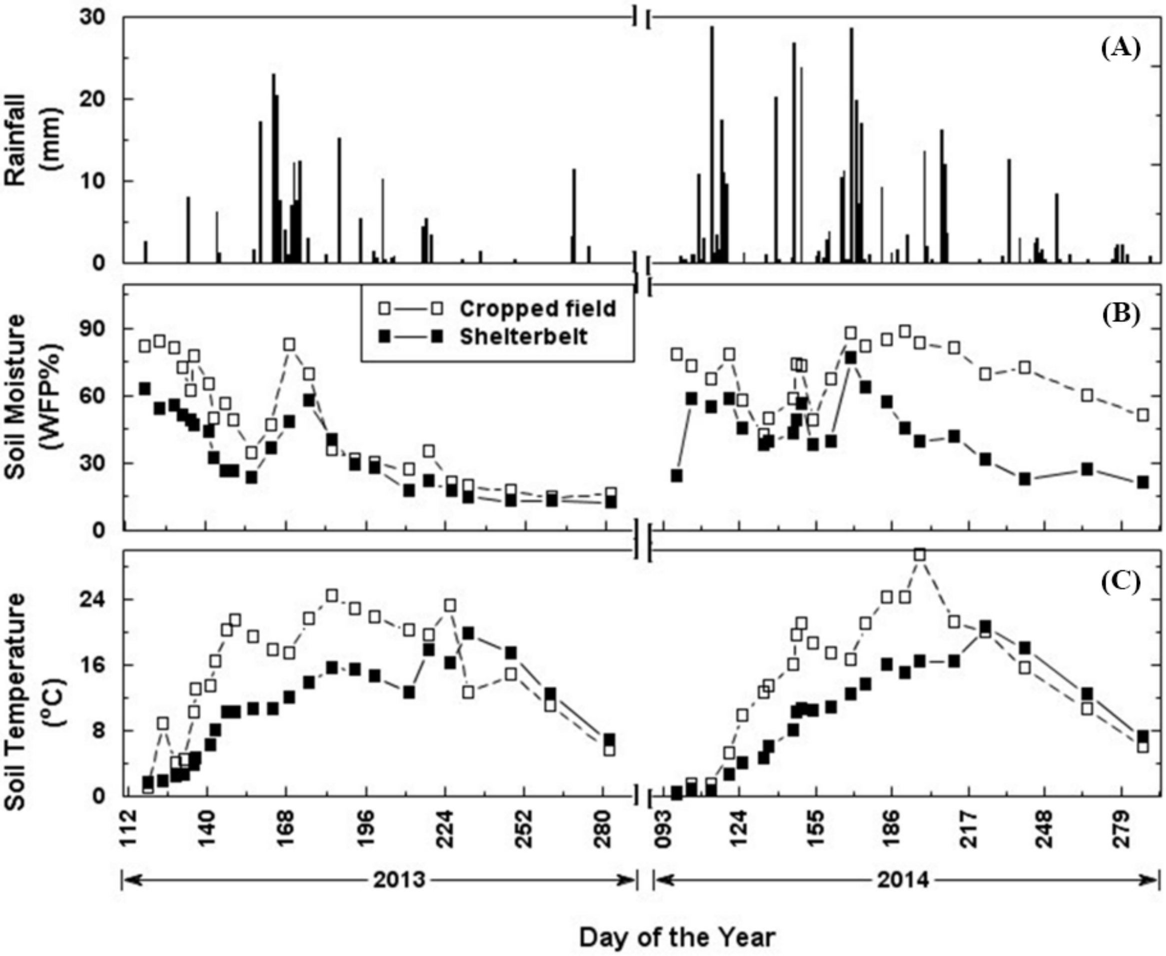

Fig. 3.2 Rainfall (A), soil water content (B) and soil temperature (C) measured at the Saskatoon during the 2013 and 2014 sampling seasons. Water filled pore space (WFPS; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both the shelterbelts and cropped fields.

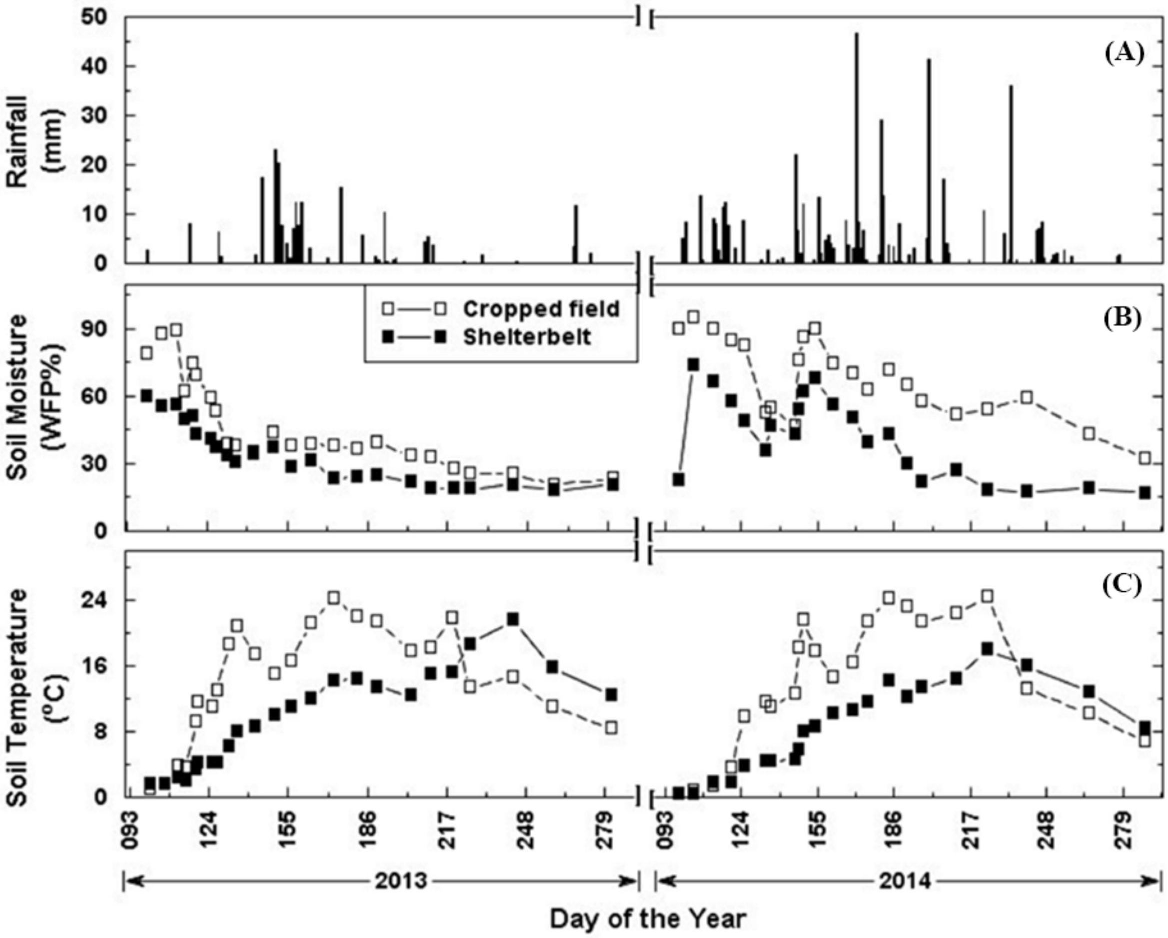

Fig. 3.3 Rainfall (A), soil water content (B) and soil temperature (C) measured at the Prince Albert during the 2013 and 2014 sampling seasons. Water filled pore space (WFPS; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both the shelterbelts and cropped fields.

3.5.2 Soil physical and chemical properties

For all sites, the near-surface (0–30 cm) soils in the shelterbelts had significantly ($P < 0.05$) greater mean SOC, total N and sand content, and significantly ($P < 0.05$) lower bulk density and clay content than the adjacent cropped fields (Table 3.4). Soil pH and C:N ratio did not differ between the two management practices. At lower soil depth (30–100 cm), SOC and total N were significantly ($P < 0.05$) greater in shelterbelts than in the cropped fields. Conversely, soil pH at lower soil depth (30–100 cm) was greater in the cropped fields than the shelterbelts; but there were no differences in soil C:N ratio and bulk density. Mean soil NO_3-N concentration in the upper 30-cm of the soil profile was greater ($P < 0.05$) in the cropped fields than the shelterbelts during July 2013 and June 2014, but not in October 2014 (Table 3.5). Extractable NH_4-N concentration was significantly ($P < 0.05$) greater in June 2013 within the shelterbelts than the cropped fields, but did not differ between the two management practices in June and October 2014.

3.5.3 Soil-atmosphere exchange of CO_2, CH_4 and N_2O

Daily CO_2 fluxes at all shelterbelt sites followed the same seasonal trends as soil temperature; i.e., daily fluxes generally increased as soil temperature increased, with peak emissions occurring when temperatures were greatest (June–August) and basal emissions occurring when soil temperatures were coldest (early spring and late fall). Patterns of CO_2 emissions in the cropped field also followed soil temperature trends, but were more irregular with major emission peaks typically associated with soil disturbance during seeding operations in May or June (Fig. 3.4). Averaged across sites, daily CO_2 fluxes during the 2013 season (Table 3.6) were significantly ($P < 0.01$) greater in the shelterbelts than the cropped fields. A similar, though weaker ($P = 0.11$) trend also was observed in 2014.

Table 3.4 Chemical and physical properties of soils under shelterbelts and cropped fields across study sites

		Sites								
		Outlook		Saskatoon		Prince Albert		Overall Mean		
Soil property	Depth (cm)	Shelterbelt	Cropped field	Shelterbelt	Cropped field	Shelterbelt	Cropped field	Shelterbelt	Cropped field	*P*-values‡
SOC (Mg ha^{-1})	0–30	77 (28.6)	52 (5.3)	108 (38.9)	93 (23.1)	132 (33.5)	90 (28.2)	106 (33.7)	78 (18.9)	0.0001
	30–100	117 (39.2)	95 (13.0)	157 (40)	118 (33.9)	149 (124)	135 (44.3)	141 (67.7)	116 (30.4)	0.0352
TN (Mg ha^{-1})	0–30	6.5 (3.0)	4.4 (0.9)	9.2 (3.2)	7.7 (2.4)	11.1 (3.5)	7.2 (2.4)	8.9 (3.2)	6.4 (1.9)	0.0001
	30–100	9.3 (3.7)	7.3 (1.6)	11.2 (3.6)	9.1 (2.7)	11.2 (10.3)	9.6 (3.9)	10.6 (5.9)	8.7 (2.7)	0.0420
C:N ratio	0–30	11.9 (2.4)	11.7 (3.0)	11.7 (2.6)	12.0 (3.2)	11.9 (1.2)	12.5 (1.6)	11.8 (2.1)	12.1 (2.4)	0.2430
	30–100	12.6 (4.1)	13.0 (3.0)	14.0 (3.8)	13.1 (3.5)	13.3 (3.3)	14.1 (3.7)	13.3 (3.7)	13.4 (3.4)	0.3270
Bulk density (Mg m^{-3})	0–30	1.2 (0.09)	1.4 (0.07)	1.1 (0.15)	1.2 (0.17)	1.3 (0.11)	1.4 (0.14)	1.20 (0.11)	1.33 (0.10)	0.0010
	30–100	1.3 (0.12)	1.3 (0.02)	1.1 (0.01)	1.1 (0.09)	1.4 (0.02)	1.4 (0.21)	1.27 (0.05)	1.27 (0.11)	0.7440
pH	0–30	7.0 (0.33)	7.2 (0.77)	7.1 (0.27)	7.1 (0.14)	6.4 (0.64)	6.2 (0.62)	6.6 (0.41)	6.9 (0.51)	0.6331
	30–100	7.3 (0.44)	7.4 (0.59)	7.2 (0.56)	7.3 (0.17)	6.5 (0.67)	7.4 (0.35)	7.0 (0.56)	7.4 (0.37)	0.0003
PSA - % sand	0–30	58.9 (5.3)	54.9 (4.4)	22.5 (4.9)	15.8 (3.8)	65.5 (7.1)	52.2 (5.2)	49.1 (6.8)	41 (4.5)	0.0365
- % silt	0–30	24.4 (10.2)	27.1 (7.1)	30.6 (12.8)	36.4 (14.5)	16.7 (11.2)	21.7 (15.4)	23.9 (11.4)	28.6 (12.3)	0.1431
- % clay	0–30	16.6 (2.8)	17.3 (2.9)	46.9 (2.7)	47.8 (3.4)	17.9 (1.8)	26.1 (2.3)	27.1 (2.43)	30.4 (2.9)	0.0195

† Numbers in parentheses are the standard deviations.

‡ Probability of a significant difference between the shelterbelts and cropped fields.

Table 3.5 Available soil nitrate (NO_3-N) and ammonium (NH_4-N) within 0-30 cm soil depth from shelterbelts and cropped fields at th study locations in July 2013, June 2014 and October 2014.

	Sites								
	Outlook		**Saskatoon**		**Prince Albert**		**Mean**		
Soil property	**Shelterbelts**	**Fields**	**Shelterbelts**	**Fields**	**Shelterbelts**	**Fields**	**Shelterbelts**	**Cropped field**	***P*-values‡**
July 2013									
NO_3-N (µg N g $soil^{-1}$)	6.6 (3.6)	10.6 (5.1)	6.0 (3.5)	12.0 (1.4)	3.1 (2.5)	14.5 (8.1)	5.2 (3.2)	12.3 (4.5)	0.0240
NH_4-N (µg N g $soil^{-1}$)	3.2 (0.9)	3.0 (0.3)	4.7 (0.2)	3.6 (0.8)	4.0 (1.0)	2.9 (0.2)	4.0 (0.75)	3.2 (0.38)	0.0450
June 2014									
NO_3-N (µg N g $soil^{-1}$)	6.0 (3.6)	9.1 (6.1)	7.8 (3.6)	10.2 (6.2)	8.7 (5.8)	12.9 (7.6)	7.5 (4.3)	10.7 (6.5)	0.0030
NH_4-N (µg N g $soil^{-1}$)	9.7 (4.5)	7.4 (4.4)	10.8 (4.9)	7.1 (1.8)	11.3 (5.7)	20.1 (11.2)	10.6 (5.0)	11.5 (5.8)	0.4190
October 2014									
NO_3-N (µg N g $soil^{-1}$)	3.9 (4.4)	8.0 (4.6)	4.3 (3.7)	3.2 (1.2)	6.7 (8.8)	7.4 (8.5)	5.0 (1.5)	6.2 (4.8)	0.4360
NH_4-N (µg N g $soil^{-1}$)	5.7 (2.0)	5.2 (1.9)	6.9 (2.5)	5.1 (1.9)	11.1 (9.7)	10.4 (6.2)	7.9 (4.7)	6.9 (3.0)	0.1290

† Numbers in parentheses are the standard deviations.

‡ Probability of a significant difference between the shelterbelts and cropped fields.

Cumulative seasonal CO_2 fluxes (averaged across study sites) also were significantly ($P < 0.01$) greater in the shelterbelts than the cropped fields in both 2013 and 2014 (Fig. 3.7A). Within study locations, cumulative CO_2 emissions in 2013 were significantly greater in the shelterbelts than the cropped fields at Outlook ($P < 0.01$), Saskatoon ($P < 0.05$), and Prince Albert ($P < 0.01$). The same trend was observed at Outlook ($P < 0.01$) and Prince Albert ($P < 0.01$) in 2014. However, at the Saskatoon site there was no significant difference in mean cumulative CO_2 emissions from the shelterbelts and cropped fields in 2014. Nevertheless, total cumulative CO_2 emissions (i.e., summed over the 2013 and 2014 seasons) were significantly greater ($P < 0.01$) in the shelterbelts than the cropped fields at all locations. Inter-annual comparisons indicate that mean cumulative CO_2 emissions in 2013 were not significantly ($P > 0.1$) different from 2014 for the shelterbelts. For the cropped field plots, however cumulative CO_2 emissions tended to be greater ($P = 0.01$) in 2014 (2.3 ± 0.19 Mg ha^{-1} yr^{-1}) than in 2013 (2.0 ± 0.09 Mg ha^{-1} yr^{-1}).

Both uptake and emission of CH_4 were observed in the shelterbelts and cropped fields, with daily fluxes ranging from -19.5 to 5.52 g CH_4-C ha^{-1} d^{-1} in the shelterbelts and from -11.0 to 17.0 g CH_4-C ha^{-1} d^{-1} in the cropped fields (Fig. 3.5). In general, daily CH_4 fluxes in the shelterbelts tended to follow soil moisture trends, with small emissions occurring during the snowmelt period followed by increased CH_4 uptake as the soil water content decreased throughout the remainder of the season. Daily CH_4 fluxes in the shelterbelts at the Saskatoon site were somewhat different in that they were (i) generally quite small and (ii) did not follow the general trend. In the cropped fields, CH_4 fluxes did not follow clear patterns, but did appear to be sensitive to soil moisture conditions. Across all sites, soil CH_4 flux rates were significantly ($P < 0.01$) lower in the shelterbelts than the cropped fields in both 2013 and 2014 (Table 3.6).

Cumulative seasonal CH_4 fluxes indicate that soils in both the shelterbelts and cropped fields were sinks for CH_4. However, cumulative CH_4 uptake was 58% and 81% greater in the shelterbelts than in the cropped fields in 2013 ($P < 0.01$), 2014 ($P < 0.01$), respectively (Fig. 3.7C). Within sites, cumulative seasonal CH_4 uptake in 2013 was two- to six-times greater in the shelterbelts than the cropped fields at Outlook ($P < 0.01$), Saskatoon ($P < 0.05$), and Prince Albert ($P < 0.01$). The same trend was observed at Outlook ($P < 0.01$) and Prince Albert ($P < 0.01$) in 2014. However, in 2014 there was no significant difference in cumulative CH_4 uptake in the shelterbelts and cropped fields at the Saskatoon site. Total cumulative CH_4 uptake (summed over the 2013 and 2014 seasons) was significantly greater ($P < 0.01$) in the shelterbelts than the cropped fields. Mean cumulative CH_4 uptake was significantly greater in 2013 than in 2014, for cropped fields ($P < 0.01$), whereas no inter-annual differences were observed in shelterbelt plots.

Despite considerable temporal and spatial variability, daily N_2O fluxes in the cropped fields generally followed an event-based/background pattern (Yates et al., 2006), with the largest emission events coinciding with the spring snow melt and precipitation/irrigation following the spring application of fertilizer N (Fig. 3.6). Conversely, there were no clear patterns in daily N_2O emissions from the shelterbelts in either year. The mean daily N_2O flux from the shelterbelts (averaged across all sites) was significantly lower than that from the cropped fields (Table 3.6) in both 2013 ($P < 0.05$) and 2014 ($P < 0.01$). Across all three study locations, seasonal cumulative N_2O fluxes were 73 % and 75 % lower in the shelterbelts than in the cropped fields in 2013 ($P < 0.01$) and in 2014 ($P < 0.01$), respectively (Fig. 3.7C). In 2013, cumulative seasonal N_2O emissions in the shelterbelts were two- to five-times lower than those in the cropped fields at Outlook ($P < 0.01$), Saskatoon ($P < 0.10$), and Prince Albert ($P < 0.10$). The same trend was observed in 2014 at Saskatoon and Prince Albert; however, differences between the shelterbelts and cropped fields were not significant at Outlook.

Table 3.6 Mean daily fluxes of CO_2, CH_4 and N_2O from shelterbelts and cropped fields across study sites

	Sites								
GHG	Outlook		Saskatoon		Prince Albert		Mean		
flux	**Shelterbelts**	**Fields**	**Shelterbelts**	**Fields**	**Shelterbelts**	**Fields**	**Shelterbelts**	**Fields**	***P*-**
CO_2									
2013	19.6 (4.2)	14.3	18.3 (5.1)	11.9	26.1 (5.8)	9.7	21.3 (5.1)	11.9	0.0004
2014	19.7 (4.4)	13.9	17.1 (3.8)	19.5	28.8 (6.5)	17.8	21.9 (4.9)	17.1	0.1014
2-yr	19.7 (4.3)	14.1	17.7 (4.4)	15.7	27.4 (6.1)	13.7	21.6 (5.0)	14.5	0.0309
CH_4									
2013	-3.7 (1.2) †	-2.4	-1.6 (1.2)	-0.5	-3.9 (0.9)	-0.05	-3.1 (1.1)	-1.0	0.0004
2014	-3.9 (0.9)	-0.8	-0.5 (0.2)	-0.6	-3.7 (1.0)	-0.3	-2.7 (0.7)	-0.6	0.0001
2-yr	-3.8 (1.0)	-1.6	-1.1 (0.7)	-0.6	-3.8 (0.9)	-0.2	-2.9 (0.9)	-0.8	0.0001
N_2O									
2013	5.2 (1.4)	13.9	3.1 (0.9)	6.7	1.6 (0.7)	9.1	3.3 (1.0)	9.9	0.0193
2014	4.9 (1.3)	6.4	4.9 (1.5)	28.2	1.5 (0.4)	25.8	3.7 (1.1)	20.1	0.0001
2-yr	5.0 (1.4)	10.1	4.0 (1.2)	17.5	1.5 (0.5)	17.4	3.5 (1.0)	15.0	0.0037

† Negative sign indicates uptake; Numbers in parenthesis represent standard error.

‡ Probability of a significant difference between the shelterbelts and cropped fields.

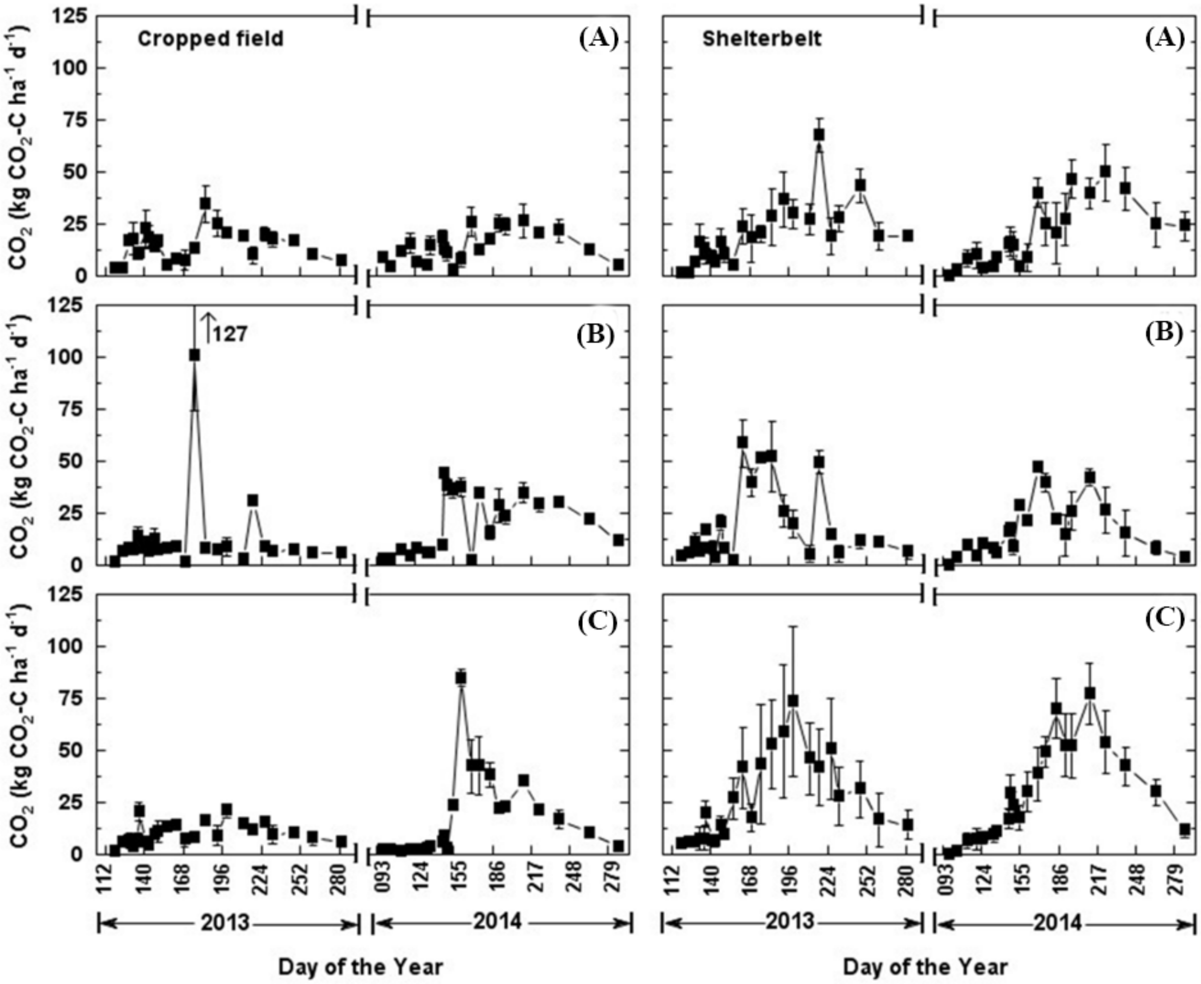

Fig. 3.4 Daily soil CO$_2$ fluxes (kg CO$_2$-C ha^{-1} d^{-1}) from shelterbelts (right) and cropped fields (left) across (A) Outlook, (B) Saskatoon and (C) Prince Albert for the periods of April to October 2013 and 2014. Error bars represent standard deviation.

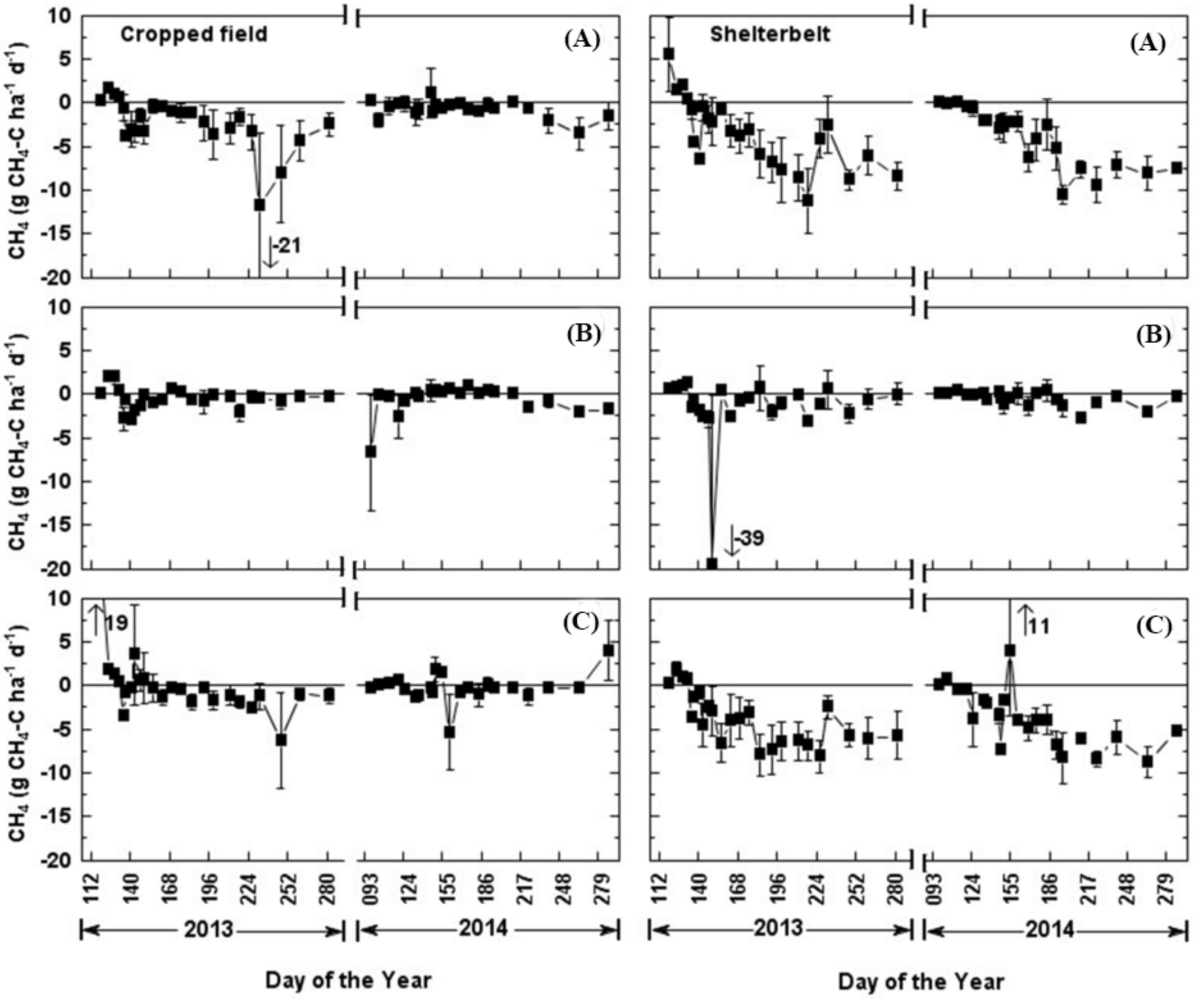

Fig. 3.5 Daily soil CH$_4$ fluxes (g CH$_4$-C ha^{-1} d^{-1}) from shelterbelts (right) and cropped fields (left) across (A) Outlook, (B) Saskatoon and (C) Prince Albert for the periods of April to October 2013 and 2014. Error bars represent standard deviation. Negative values indicate uptake.

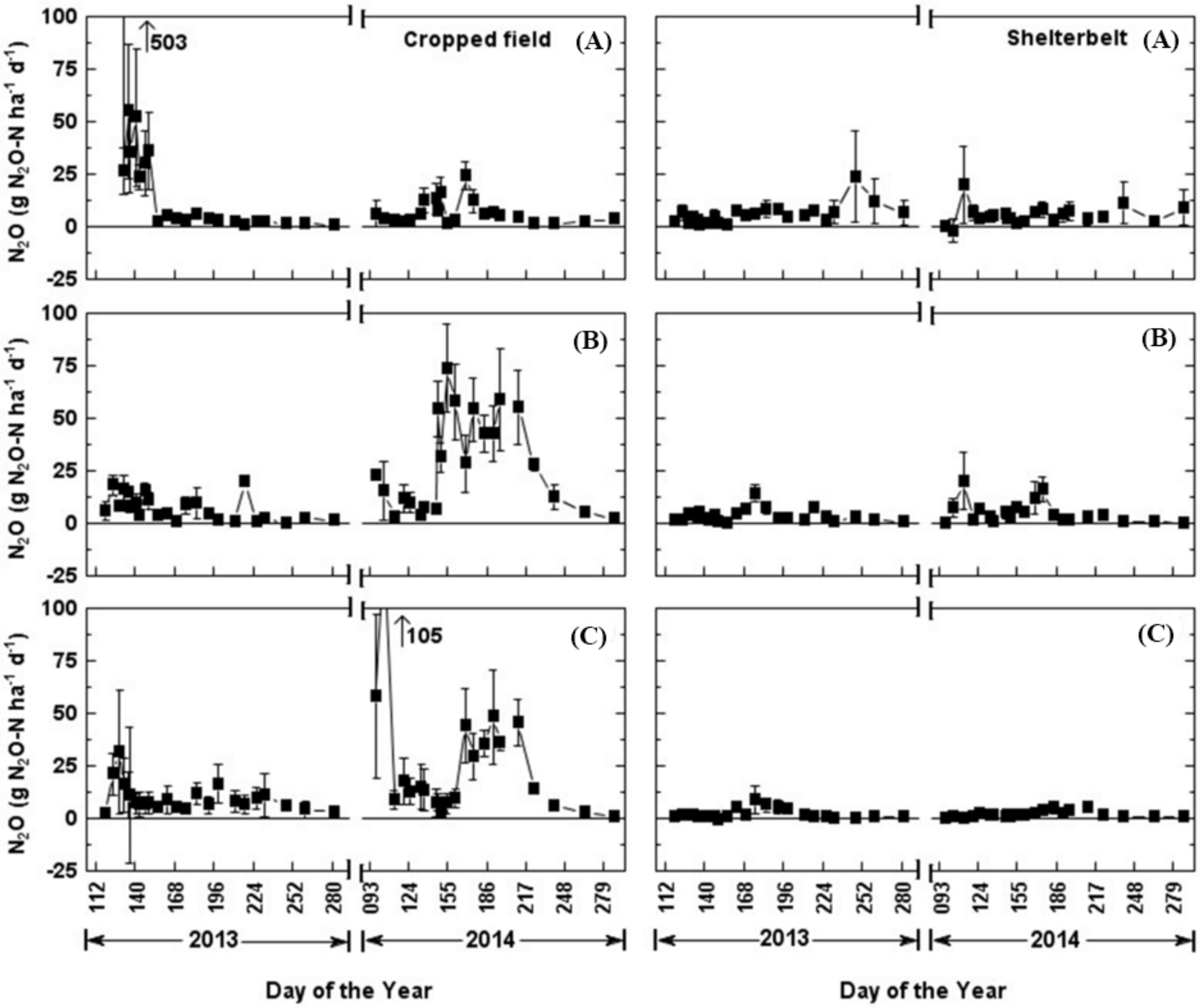

Fig. 3.6 Daily soil N$_2$O fluxes (g N$_2$O-N ha^{-1} d^{-1}) from shelterbelts (right) and cropped fields (left) across (A) Outlook, (B) Saskatoon and (C) Prince Albert for the periods of April to October 2013 and 2014. Error bars represent standard deviation. Negative values indicate uptake.

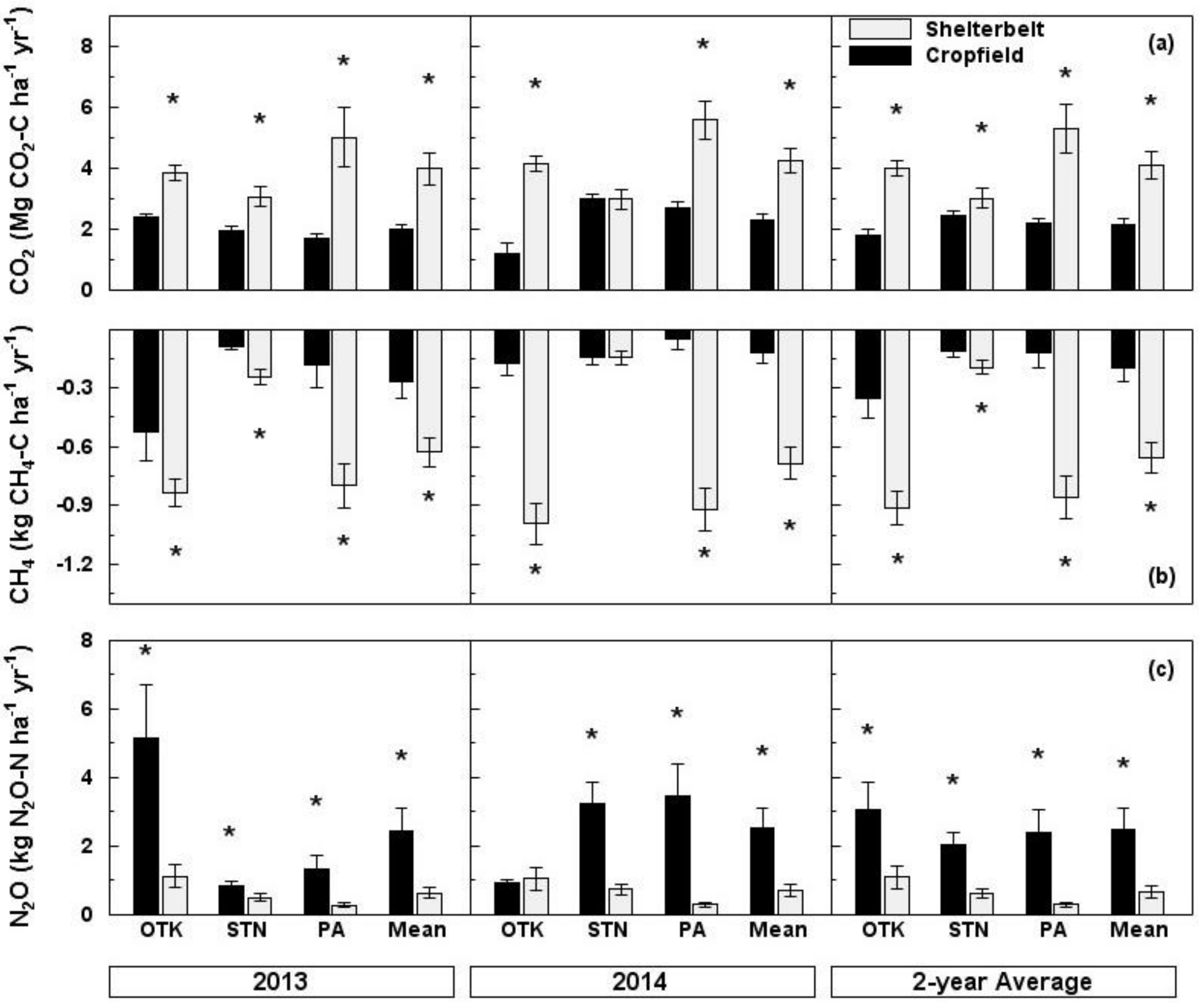

Fig. 3.7 Cumulative fluxes of (A) CO_2 (Mg CO_2-C ha^{-1} yr^{-1}), (B) CH_4 (kg CH_4-C ha^{-1} yr^{-1}) and (C) N_2O (kg N_2O-N ha^{-1} yr^{-1}) from shelterbelts and cropped fields across Outlook (OTK), Saskatoon (STN) and Prince Albert (PA) for the periods of April to October 2013 and 2014. Error bars represent standard error. Negative numbers indicate uptake.

3.5.4 Influence of soil temperature and soil moisture on soil gas fluxes

Soil CO_2 fluxes at all three study locations were significantly correlated with soil temperature in both the shelterbelts ($r = 0.62$; $P < 0.001$) and cropped fields (Pearson coefficient $r = 0.43$ and $P < 0.001$). Soil CO_2 fluxes also were negatively correlated with soil moisture in the shelterbelts (Pearson coefficient $r = -0.31$; $P < 0.001$), though no correlation between CO_2 flux and soil moisture was observed in cropped fields. I was unable to identify any significant correlation between soil CH_4 flux and soil temperature in the cropped field, but there was a strong negative correlation between CH_4 flux and soil temperature ($r = -0.45$; $P < 0.001$) in the shelterbelts. Soil CH_4 flux was significantly correlated with soil moisture in both the shelterbelts ($r = 0.57$; $P < 0.001$) and cropped fields ($r = 0.26$; $P < 0.01$). Whereas N_2O fluxes in the cropped fields were negatively correlated with soil temperature ($r = -0.21$; $P < 0.01$) and positively correlated with soil moisture ($r = 0.33$; $P < 0.001$), there were no significant effects of soil temperature or moisture on N_2O fluxes in the shelterbelts.

3.6 Discussion

3.6.1 Soil Carbon dioxide exchange

Soil CO_2 flux rates are within the range of those reported from both cropped and treed systems (Table 3.7). However, CO_2 fluxes were greater in the shelterbelts, which is likely due to enhanced microbial activity given the modified microclimate and the continuous litter cover that serves as a source of available C (Mosier et al., 2006). Furthermore, the greater cumulative CO_2 emissions in shelterbelts is a result of increased tree root respiration during active growth periods, and to heterotrophic decomposition of tree litter, which increases with increasing soil temperature and moisture.

Despite the greater CO_2 emissions in the shelterbelts compared to the cropped fields, SOC content measured in both the near surface (0-30 cm) and subsurface (30-100 cm) soils was greater in the shelterbelts. An increase in SOC content in the shelterbelts is likely a result of long-term additions of tree litter and roots and reduced soil disturbance following tree establishment. As carbon inputs to soil increase, soil organic matter increases (Paustian et al., 2000).

Table 3.7 Comparison of CO_2, CH_4 and N_2O flux ranges in treed systems and cropped fields across Temperate North America

Reference	Study location	Land use	CO_2 flux ($kg\ CO_2\text{-}C\ ha^{-1}\ d^{-1}$)	CH_4 flux ($g\ CH_4\text{-}C\ ha^{-1}\ d^{-1}$)	N_2O flux ($g\ N_2O\text{-}N\ ha^{-1}\ d^{-1}$)
Present study	Saskatchewan, Canada	Cropped fields	14.5 (12.4)†	-0.8 (2.1)‡	15.0 (15.2)
Present study	Saskatchewan , Canada	Shelterbelts	21.6 (17.2)	-2.9 (3.1)	3.5 (3.6)
Peichl et al. (2010)	Southern Ontario, Canada	temperate pine forest	2 to 50¶	-28 to 44	-8 to 3.5
Mosier et al (2006)	Northeastern Colorado, USA	Irrigated cropping system	0.3 to 77	-4 to 21.6	-1.2 to 132
Izaurralde et al. (2004)	Western Canada	Agricultural toposequences	ND§	ND	48 to 336
Matson et al. (2009)	Saskatchewan, Canada	Boreal forest	ND	-28 to 2,500	-5 to 7
Ellert and Janzen (2008)	Alberta, Canada	Irrigated cropping system	0.2 to 48	-3 to 2	-1 to 72
Evers et al. (2010)	Ontario, Canada	Monocropping system	ND	ND	4.5 to 15.7
Evers et al. (2010)	Ontario, Canada	Tree based systems	ND	ND	-2.8 to 16.7
Kim et al. (2009, 2010)	Northcentral Iowa, USA	Cropped field	ND	-2 (1.0)	39.4 (7.1)
Kim et al. (2009, 2010)	Northcentral Iowa, USA	Riparian buffer	ND	-5 to 9	2.8–11.0

† Values in parenthesis represent standard deviation of the mean

‡ Negative sign indicates uptake.

§ ND, Not determined.

¶ Values represent ranges of measured GHG fluxes, obtained from studies that did not indicate the mean and standard deviation in daily GHG measurements

Shelterbelts, like other agroforestry systems could maintain or increase SOC mainly through root turnover and continuous addition of litter to soil (Jose et al., 2000). However, increase in soil organic matter turnover is associated with increased soil CO_2 fluxes. Strong relationships between soil organic matter and soil respiration have been reported both in field studies (Franzluebbers et al., 1995) and in laboratory incubations (Mallik and Hu, 1997). Thus, although the shelterbelt soils emitted greater amounts of CO_2, the data show that cumulative SOC content was greater in the shelterbelts, suggesting greater quantities of organic matter inputs to the soil by the trees, which serve as substrates for soil microbial decomposition; and perhaps greater rates of tree root respiration.

The activities of soil microorganisms are mainly controlled by soil temperature, moisture content, aeration, organic matter availability and quality. These factors could further be modified by landscape, soil texture, vegetation type and management (Mosier et al., 2006). Modification of the micro-climate in the shelterbelts is likely to have resulted in enhanced soil microbial activity, thereby contributing to increased soil CO_2 emissions. In a study of soil respiration and microbial biomass in a pecan – cotton alley cropping system, Lee and Jose (2003) reported the influence of modified micro-environment on soil respiration in the 47-year-old pecan – cotton alley systems relative to the cotton monoculture. They concluded that the introduction of trees into cropped fields could enhance soil fertility and sustainability of farmlands by improving soil microbial activity and accreting residual soil carbon.

At the Saskatoon site, cumulative CO_2 emissions from the cropped fields were the same as those in the shelterbelts (Fig. 3.7A) and the data demonstrates that this is a product of increased productivity and biological activity following high fertilizer N additions (70 kg N ha^{-1}) to the cropped fields. Short-term positive effects of N fertilization on soil respiration has been reported in many field (Bowden et al., 2000; Bowden et al., 2004; Peng et al., 2011) and laboratory incubation studies (Söderstrõm et al., 1983). The increase in soil CO_2 emissions have been attributed to increased microbial activity and increased root production in response to N additions (Magill et al., 1997; Lovell and Hatch, 1998). Increased CO_2 emissions in agroforestry and forestry systems have been well documented. For example, Peichl et al. (2006) reported greater soil respiration in 13 year Norway spruce and hybrid poplar intercropped with barley (4.5 and 3.7 Mg C ha^{-1} yr^{-1}, respectively) than in a barley monocropping system (2.8 Mg C ha^{-1} yr^{-1}). Soil respiration in forested soils could be up to three times higher than in conventional agricultural lands (Raich and Schlesinger, 1992).

3.6.2 Methane Exchange

Rates of soil CH_4 oxidation in this study were within the range reported in crop lands and forested soils in temperate regions (Table 3.7). Soil moisture accounted for about 56% and 26% of the variability in soil CH_4 fluxes from shelterbelts and cropped fields, respectively, which is in agreement with studies that have reported increasing CH_4 consumption with decreasing soil moisture content in forested soils (Castro et al., 1994; Rosenkranz et al., 2006). Although temperature controls on soil CH_4 fluxes have been reported (MacDonald et al., 1997), this study and most others have found this relationship to be of little significance - with controls on gas diffusivity (e.g. soil bulk density, moisture content and texture) being of greater importance (Ball et al., 1997; Smith et al., 2003; Peichl et al., 2006). My data indicate that shelterbelt soils have significantly higher levels of organic C and total N than the adjacent cropped fields, but that the cropped fields have greater bulk densities and higher nitrate levels. The lower soil bulk density in the shelterbelts suggests greater gas diffusivity, which has been reported to increase CH_4 consumption by methanotrophs in soil crumbs and aggregates (Ball et al., 1997). Methane is formed in soils by the microbial breakdown of organic compounds in strictly anaerobic conditions, at a very low redox potential. Since trees are deep rooting and have higher water demands than arable crops (Thevathasan et al., 2012), it is likely that the greater soil methane uptake (consumption) that was observed in the shelterbelts was mainly a product of an improved aeration status – reflecting a reduction in soil moisture and soil bulk density due to root activity.

Upland soils are an important sink for methane (CH_4) contributing up to 15% of annual global CH_4 destruction (Powlson et al., 1997). However, the size or strength of the sink are affected by land management; e.g. intensification of agricultural practices decreases the soil sink for atmospheric CH_4 (Bronson and Mosier, 1993). Agronomic practices such as tillage, use of pesticides and herbicides and fertilization have been shown to have various degrees of inhibitory effect on CH_4 uptake in arable lands (Hansen et al., 1993; Arif et al., 1996; Mosier et al., 1997; Powlson et al., 1997; Topp and Pattey, 1997; Hütsch, 2001). As well, several studies have demonstrated greater methane consumption in undisturbed forests and grassland soils than in intensively cultivated farmlands (Goulding et al., 1996; MacDonald et al., 1996; Prieme and Christensen, 1999; Robertson et al., 2000).

Studies on methane fluxes in agroforestry systems relative to conventional agricultural fields are limited. Based on this study, however, the incorporation of trees (i.e. shelterbelts) into crop production systems significantly increase the strength of the overall soil CH_4 sink strength, and on a per area basis may contribute substantially to the mitigation agricultural GHG emissions (Fig. 3.7B). Trees take up excess soil moisture, improve soil organic matter status and decrease soil bulk density, all of which create a favourable environment for methane consumption. This is in agreement with Kim et al. (2010) who reported significant increases in soil C, total N and pH and a significant decrease in soil bulk density in 7 to 17 year-old re-established riparian forest buffers relative to cropped fields, although they found no significant methane uptake from the riparian forest system compared to the cropped fields.

3.6.3 Nitrous oxide Exchange

The soil N_2O fluxes observed in this study were within the range of those reported from both cropped and treed systems (Table 3.7). While increased N_2O emissions following thawing of frozen soils in early spring and fertilization during seeding operations contributed significantly to seasonal N_2O emissions in the cropped fields, shelterbelt soils were less responsive to such events (Fig. 3.6). The high fluxes observed in early spring were perhaps a combined effect of residual N from fertilization and residue decomposition in the previous cropping season that serve as substrate for soil microbial community, and high soil moisture, all of which promote denitrification. Increased N_2O emissions during the early spring has been reported at other Canadian cold weather sites and can be attributed to structural shifts in the microbial community that is active in thawing soils (Sharma et al., 2006), and to suppressed soil N_2O consumption (Chapuis-Lardy et al., 2007). Izaurralde et al. (2004) reported that the presence of residual N coupled with elevated soil moisture conditions facilitated N_2O emissions through enhanced denitrification.

In the cropped fields that received fertilization (see Table 3.3), soil N_2O fluxes quickly increased following N fertilization (Fig. 3.6A-3.6C). The impact of N fertilization was more pronounced during the 2014 cropping seasons at Saskatoon (Fig. 3.6B) and Prince Albert (Fig. 3.6C) where fluxes were highest in the months following fertilization and tended to decline to background levels towards the end of the season. Similar observations have been reported in other studies (e.g. Mosier et al., 2006; Ellert and Janzen, 2008).

Positive relationships were observed between soil temperature and moisture and N_2O fluxes in the cropped fields, but not in the shelterbelts. The lack of any temperature and moisture effects on N_2O fluxes from shelterbelt soils presumably reflects the moderating effect of older trees on microclimate and its attendant effects on the soil microbial community, and is in agreement with the results of Peichl et al. (2010) who found that limited available N in temperate pine forests prevented a clear relationship between N_2O fluxes and soil temperature and moisture. In this study, shelterbelt soils (except soils under caragana) had significantly lower nitrate levels than the adjacent cropped fields in June 2013 and July and October 2014 (Table 3.5). Thus, the combined effects of low nitrate levels in shelterbelts coupled with the more moderate (cooler and drier) micro-climate is most likely responsible for the lack of correlation between N_2O flux and soil temperature and moisture. Among the shelterbelt tree species that were studied, soils under caragana shelterbelts contained significantly greater mineral N concentrations and emitted more N_2O relative to other shelterbelt species – attributed to the N-fixing ability of caragana tree species. Caragana has been shown to fix about 75–85% of its N from the atmosphere (Moukoumi et al., 2013) - returning about 20–60 kg N ha^{-1} to the soil in the litter (Issah et al., 2014)

Whereas inter-annual differences between mean cumulative N_2O emissions were not significant for the shelterbelts, site-specific differences ($P < 0.01$) were observed for the cropped fields (Fig. 3.7C), suggesting that in each year the amount of N_2O emissions in the cropped field changed mainly due to changes in management practices and weather conditions. Averaged across all study locations, seasonal cumulative N_2O fluxes were 74% lower in the shelterbelts than in the cropped field soils. This difference may be attributed on one hand to greater soil aeration in shelterbelts, which inhibits denitrification process, and on the other hand by limited mineral N due to exclusion of fertilization on soils occupied by shelterbelts. This is congruent with Beaudette et al. (2010) who found seasonal N_2O emissions that were three times greater in monocropping systems than in tree-based intercropping (TBI) systems and a corresponding reduction in soil moisture in the TBI systems.

3.6.4 Carbon implication of shelterbelt establishment in agricultural fields

A reduction in CO_2 emissions in agricultural soils is synonymous with increasing soil C storage (Paustian et al., 2000). Whether an ecosystem is a source or sink of CO_2 will depend on the net difference between the amount of CO_2 taken up by plants through autotrophic fixation and the amount emitted from the soil surface back to the atmosphere through heterotrophic decomposition of organic materials and root respiration. Thus, measurements of soil CO_2 fluxes alone are not indicative of the source or sink potential of the ecosystem. Net CO_2 flux to the atmosphere can be assessed from changes in soil C (Robertson et al., 2000). However, the high degree of variability inherent to soils can mask small annual changes in soil C; consequently, such assessments are limited to long term soil C monitoring studies (Mosier et al., 2006). The greater soil CO_2 emissions that were observed in the shelterbelts relative to the adjacent cropped field do not necessarily reflect less soil C storage. Results from the soil C sampling, averaged across all study sites indicate that SOC content (0–30 cm soil depth) in the shelterbelts was greater by 28 Mg ha^{-1} than in the adjacent cropped fields, representing a 27% difference per hectare of shelterbelts (Table 3.4). This is a conservative estimate since litter C in the shelterbelts was not included. The age of shelterbelts in this study ranged from 19 to 41 years, thus the 28 Mg ha^{-1} change reported in shelterbelts may represent an annual accrual of 0.7 to 1.5 Mg ha^{-1} yr^{-1} over the life of the shelterbelts, assuming that net C storage in the cropped fields over time is negligible. Greater SOC content in shelterbelts compared to agricultural soils has been reported elsewhere (Sauer et al., 2007), and attributed to increased organic inputs by tree litter and the deposition of wind-blown sediments.

The data presented in Table 3.8 summarizes the seasonal fluxes of non-CO_2 gases (CH_4 and N_2O) from the shelterbelts and adjacent cropped fields in terms of CO_2 equivalents. Averaged across study sites and years, total seasonal exchange of non-CO_2 GHGs in shelterbelts was reduced by 0.55 Mg CO_2e ha^{-1} compared to the cropped fields. Seasonal N_2O was 0.54 Mg CO_2e ha^{-1} lower in the shelterbelts, whereas soil CH_4 sink strength was increased by 0.01 Mg CO_2e ha^{-1}, representing 2% of non-CO_2 GHG mitigation by shelterbelts.

Clearly, the integration of trees into an agricultural landscape can help mitigate agricultural GHGs. The greater N_2O mitigation in shelterbelts may be mainly related to (i) N limitation in soils underneath shelterbelts due to lack of N fertilizer and (ii) the activity of widespread tree root network and their ability to remove residual or excess water and NO_3-N that would otherwise be available for denitrification and leaching on- or off-site. The root absorbed N is eventually returned back to the soil through leaf litter fall, resulting in more efficient N cycling, decreased fertilizer N demand by surrounding soils and, by implication, reduced N_2O emissions from N fertilization (Thevathasan et al., 2012). This safety-net role of tree roots has been demonstrated by Allen et al. (2004), who reported between 48% and 71% reduction in NO_3 leaching from a pecan-cotton alley cropping system with unpruned tree roots compared to root exclusive treatments.

Table 3.8 Comparison of seasonal cumulative exchange of non-CO_2 GHGs (CO_2 equivalents yr^{-1}) for shelterbelts and cropped fields across three sampling locations

	Sites						**Mean**		
	Outlook		**Saskatoon**		**Prince Albert**				
	Shelterbelts	**Fields**	**Shelterbelts**	**Fields**	**Shelterbelts**	**Fields**	**Shelterbelts**	**Fields**	**P-values‡**
CH_4 (kg CO_2e ha^{-1} yr^{-1})									
2013	-19.2 (1.7)†	-12.1 (3.3)	-5.6 (0.9)	-2.0 (0.4)	-18.4 (2.6)	4.2 (2.6)	-14.4 (1.7)	-6.1 (2.1)	0.0001
2014	-22.8 (2.4)	-4.1 (1.3)	-3.3 (0.8)	-3.3 (1.0)	-21.1 (2.5)	1.2 (1.2)	-15.7 (1.9)	-2.9 (1.2)	0.0001
Mean	-21.0 (2.0)	-8.1 (2.3)	-4.5 (0.8)	-2.7 (0.7)	-19.8 (2.6)	2.7 1.9)	-15.1 (1.8)	-4.5 (1.6)	0.0001
N_2O (kg CO_2e ha^{-1} yr^{-1})									
2013	327 (96)	1522 (453)	144 (33)	238 (43)	77 (20)	393 (113)	183 (50)	718 (208)	0.0022
2014	307 (97)	272 (26)	215 (43)	958 (175)	85 (20)	1017 (282)	202 (54)	749 (161)	0.0001
Mean	317 (97)	897 (240)	180 (39)	598 (109)	81 (20)	705 (198)	192 (52)	733 (182)	0.0001
Total (kg CO_2e ha^{-1} yr^{-1})									
2013	308 (97)	1510 (456)	138 (34)	236 (44)	57 (22)	388 (116)	168 (51)	712 (205)	0.0001
2014	284 (100)	268 (28)	212 (45)	955 (176)	64 (22)	1016 (284)	187 (56)	746 (162)	0.0001
Mean	296 (99)	889 (242)	175 (39)	595 (110)	61 (22)	702 (200)	177 (54)	729 (184)	0.0001

† Negative sign indicates uptake; Numbers in parenthesis represent standard error.

‡ Probability of a significant difference between the shelterbelts and cropped fields.

3.7 Conclusion

My study suggests that shelterbelts have substantial potential to increase soil C content and reduce nitrous oxide emissions, while maintaining a strong methane sink. Cumulative CO_2 emissions from soils underneath shelterbelts (3.95 Mg CO_2-C ha^{-1} in 2013 and 4.21 Mg CO_2-C ha^{-1} in 2014) were significantly greater than emissions from cropped fields (2.0 Mg CO_2-C ha^{-1} in 2013 and 2.27 Mg CO_2-C ha^{-1} in 2014). However, SOC content (0–30 cm soil depth) in shelterbelts was 28 Mg ha^{-1} greater than that in the adjacent cropped fields. On a per area bases, this represents a 27% increase in SOC content in the shelterbelts. Cumulative CH_4 uptake was greater in the shelterbelts (-0.63 kg CH_4-C ha^{-1} in 2013 and -0.69 kg CH_4-C ha^{-1} in 2014) than the adjacent cropped fields (-0.26 kg CH_4-C ha^{-1} in 2013 and -0.12 kg CH_4-C ha^{-1} in 2014). Conversely, the cropped field soils emitted significantly greater quantities of N_2O (2.43 kg N_2O-N ha^{-1} in 2013 and 2.53 kg N_2O-N ha^{-1} in 2014) than the shelterbelts (0.62 kg N_2O-N ha^{-1} in 2013 and 0.68 kg N_2O-N ha^{-1} in 2014). Total seasonal exchange of non-CO_2 GHGs was reduced by 0.55 Mg CO_2e ha^{-1} in shelterbelts as compared with the cropped fields, 98% of which was soil N_2O flux.

While increased N_2O emissions following thawing of frozen soils in early spring and fertilization during seeding operations contributed significantly to seasonal N_2O emissions in cropped fields, soils beneath shelterbelts were less responsive to the spring snowmelt events. Patterns of soil temperature, moisture and organic matter distribution beneath shelterbelts suggested modification in soil micro-environment due to sheltering and root activity, which may be responsible for increased soil CO_2 emissions and CH_4 uptake. Further research is needed to measure the spatial extent of influence of shelterbelts on GHGs in cropped fields. Such studies may be addressed by monitoring GHG emissions at various distances from shelterbelts.

4. MODERATING EFFECT OF A HYBRID POPLAR-CARAGANA SHELTERBELT ON GREENHOUSE GAS FLUXES ALONG A GRADIENT FROM THE SHELTERBELT TO AN ADJACENT CROPPED FIELD

4.1 Preface

As discussed in chapter 3, on-farm shelterbelts function as net biological sinks of CO_2 and can play a role in mitigating soil GHG gas emissions in agricultural landscapes. Specifically, it was concluded that shelterbelts play a significant role in mitigating soil GHGs by increasing soil C content and reducing N_2O emissions, while maintaining a strong CH_4 sink. However, the spatial extent of the role of shelterbelts in reducing agricultural GHGs is not known, given the lineal structure of shelterbelts. Earlier studies have determined that shelterbelts have measurable effects on soil properties and crop yield at various distances away from the shelterbelt strip attributed mainly to greater tree root and shelterbelt influence on soil micro-climatic conditions. However, there is a lack of data on the effect of shelterbelts on soil GHG exchange at various distances away from the shelterbelt. The objective of this study was to quantify, changes in soil CO_2, CH_4 and N_2O fluxes along a transect from a shelterbelt to the center of the adjacent field. This was achieved by two-year monitoring of soil GHG exchange at various distances away from a shelterbelt within the parkland region of Saskatchewan Canada, using steady-state vented chambers.

4.2 Abstract

The influence of shelterbelts on soil properties and crop yield at various distances from the shelterbelt have been studied; however, there are no available data detailing the spatial range of shelterbelt effects on soil CO_2, CH_4 and N_2O emissions along the gradient from the shelterbelt to the center of the adjacent cropped field. The objective of this study was to quantify, changes in soil CO_2, CH_4 and N_2O fluxes along a transect from a shelterbelt to the center of the adjacent field. During 2013 and 2014, soil GHG exchange at various distances away from a 33-year-old, two-row hybrid poplar-caragana shelterbelt was monitored within the parkland region of Saskatchewan Canada, using steady-state vented chambers. Gas samples were collected along parallel transects situated at the shelterbelt strip (0H), shelterbelt edge (0.2H), at the edge of the adjacent cropped field (0.5H) and in the cropped field at distances of 40 m (1.5H) and 125 m (5H) from the shelterbelt. Summed over the entire study period, cumulative CO_2 emissions were greatest at 0H (8032 ± 502 kg CO_2-C ha^{-1}) and lowest at 5H (3348 ± 329 kg CO_2-C ha^{-1}); however, the decrease in CO_2 emissions at increasing distances away from the shelterbelt was irregular, with soil temperature and TOC distribution being the dominant controls. Soil CH_4 oxidation was greatest at 0H (-1447 ± 216 g CH_4-C ha^{-1}), but decreased as distance from the shelterbelt increased. Conversely, soil N_2O emissions were lowest at 0H (345 ± 15 g N_2O-N ha^{-1}) but increased with increasing distance from the shelterbelt. Patterns of soil CH_4 uptake and N_2O emissions were strongly correlated with root biomass, bulk density, soil temperature and soil moisture in the upper 30-cm of the soil profile. Shelterbelt influence on soil GHG emissions were observable up to the location at 1.5H, beyond which no further shelterbelt-induced effects on soil GHG exchange were observed. The study highlights the importance of tree species selection and tree root distribution in determining the spatial range of shelterbelt effect on GHG emissions in adjacent fields

4.3 Introduction

During the past century, over 600 million shelterbelt trees have been distributed to Prairies land owners under the provisions of the Prairie Farm Rehabilitation Act, to protect Canadian farms from wind erosion (Howe, 1986; Wiseman et al., 2009). Particularly in Saskatchewan, Canada it is estimated that there are over 60,000 km of planted shelterbelts throughout the province, and considerably more in the Canadian Prairies (Amichev et al., 2014). In the last two decades, shelterbelts have been recognized as a strategy for mitigating atmospheric CO_2 through C sequestration in tree biomass (Kort and Turnock, 1999) and in SOC pools (Sauer et al., 2007). However, there have been no data detailing the dynamics of GHG fluxes at various distances from a shelterbelt in order to uncover the role of shelterbelts in mitigating the emissions of CO_2, CH_4 and N_2O in cultivated soils.

Shelterbelts have a measurable influence on soil properties in adjacent cropped fields (Kowalchuk and de Jong, 1995) and, as such, could affect the exchange of soil-derived GHGs. Changes in soil properties following shelterbelt establishment are a result of modification of local soil microclimate, soil organic matter (SOM) and tree root distribution in adjacent soils (Kort, 1988; Sauer et al., 2007). In general, the leeside of a shelterbelt is characterized by increased springtime soil moisture and relative humidity, along with reduced evaporation and night-time air temperatures (Rosenberg, 1974; Wang et al., 2010; Kort et al., 2011). At the same time, tree roots continuously extract soil moisture from an area 1.5- to 2-times the height of the shelterbelt (i.e., 1.5H–2H) (Kowalchuk and de Jong, 1995). Although the extraction of soil moisture by tree roots is partly offset by increased spring moisture through snow accumulation, it may result in severe competition for moisture between tree roots and crops, particularly in dry years (Kort, 1988).

Shelterbelts have been shown to influence crop yield at various distances away from the shelterbelt strip (Kort, 1988). Kort (1988) reported no crop yield at a distance of 0H to 0.5H; a 50% reduction in yield due to root competition at distances from 0.5H to 1H; and a shelterbelt-induced increase in crop yield at distances from 1.5H to 15H, with the largest increase occurring between 1.5H to 3H. Overall, the average yield in the area under the influence of the shelterbelt (i.e., at a distance of 0H to 15H) was about 3.5% greater than that at the field center, which was not influenced by the trees. In a similar study, Kowalchuk and de Jong (1995) reported peak soil N and P concentrations at 2H, which was described as a zone with less root competition and enhanced shelterbelt-induced reduction in wind speed and evaporation.

Shelterbelts, like other agroforestry systems have the potential to maintain or increase SOC in adjacent soils, mainly through root turnover and continuous addition of litter to the soil (Jose et al., 2000). Studies have shown increased soil organic carbon (SOC) content in soils underneath trees compared to adjacent cropped fields (Sauer et al., 2007; Martens et al., 2003). Bronick and Lal (2005) reported SOC in wooded soils was about twice that in the adjacent cropped field. In an assessment of SOC dynamics and sources in two 35-year old coniferous shelterbelts, Hernandez-Ramirez et al. (2011) reported that SOC in the shelterbelts was more than 57% greater than that in the adjacent cropped fields. The increase in SOM content within the shelterbelts was explained by a combination of long-term additions of tree litter debris and the entrapment of organic matter-rich wind-blown sediments (Sauer et al., 2007).

In addition to their influence on soil physical and chemical properties, shelterbelts and other tree based systems have a positive impact on microbial abundance and diversity (Wojewuda and Russel, 2003; Lacombe et al., 2008). Despite making up only 1–3% of the total soil SOM (Martens, 1995), soil microbial communities play important roles in catalyzing the indispensible transformations in biogeochemical cycles of the biosphere (Wojewuda and Russel, 2003; Lacombe et al., 2008; van der Heijden and Wagg, 2013), as well as the production of soil-derived GHGs (Ellert and Janzen, 2008). Shelterbelts with a well-developed canopy have been shown to protect soil micro fauna from high temperature variations and moisture stress (Martius et al., 2004). As well, Karg et al. (2003) reported greater faunal and microbial biomass in soils beneath a shelterbelt, which decreased with increasing distance from the shelterbelt. Thus, shelterbelt-induced improvements in microbial and faunal biomass could have a measurable effect on the dynamics of soil GHG exchange along the gradient from shelterbelt to the middle of the field. Moreover, Wojewuda and Russel (2003) reported a strong positive correlation among soil microbial biomass, soil respiration and SOM distribution at various distances away from the shelterbelt, suggesting that SOM distribution influences soil microbial activity, soil respiration and perhaps soil GHG emissions along the gradient from shelterbelt to the field.

Although a considerable number of studies have described shelterbelt effects on soil properties and crop yield at various distances from the shelterbelt, there are no available data detailing the range of shelterbelt effects on soil CO_2, CH_4 and N_2O emissions along the gradient from the shelterbelt to the center of the adjacent cropped field. Yet, this information is necessary if we are to develop accurate estimates of the C sequestration and GHG mitigation potential of shelterbelts for regional C budgets and GHG inventories. Data on how shelterbelts influence the dynamics of soil C distribution and GHG fluxes in the surrounding soils will lead to a more accurate estimation of the environmental and economic benefits of shelterbelt establishment, which in turn will support policy and management decisions on shelterbelt systems. Given the thousands of kilometers of planted shelterbelts throughout the Canadian prairies (Wiseman et al., 2009; Amichev et al., 2014), the impact of shelterbelts on C-sequestration and mitigation of soil GHG in adjacent croplands may be of significant environmental importance. The objective of this study was to quantity the influence of a shelterbelt on soil-atmosphere exchange of CO_2, CH_4 and N_2O along a gradient from a shelterbelt to the center of the adjacent cultivated field.

4.4 Materials and Methods

4.4.1 Study Area

Studies were carried out at the Conservation Learning Centre, located approximately 18 km south of Prince Albert, within the parkland region of Saskatchewan, Canada (53^o 01’ N, 105^o 46’ W). Climate data for the 2013 (May through October) and 2014 (April through October) sampling seasons were obtained from the Environment Canada meteorological station located at Prince Albert, SK (Environment Canada, 2015).

The study consisted of a 31-year old shelterbelt strip (planted in 1982), a transition zone and an adjacent cropped field north of the strip. The strip was a two-row shelterbelt comprising of a row of hybrid poplar (*Populus spp.*) and a row of caragana (*Caragana arborescens*). The hybrid poplar shelterbelt had an average diameter at breast height (DBH) of 52.7 cm, with an average height of 25 m and a spacing of 2 m between trees; the caragana shelterbelt had an average height of 6 m and a spacing of 1-m between trees. The entire strip was approximately 200 m long and 10 m wide, while the transition zone was situated along the outer edge of the shelterbelt and the inner edge of the cropped field.

The adjacent cultivated field, which began 12 m from the base of the trees was seeded to barley (*Hordeum vulgare*) on May 21, 2013. The crops received a spring application of urea fertilizer at the rate of 50 kg N ha^{-1} during seeding. In 2014, the field was tilled on June 1st but was not seeded due to excessive amounts of residue and saturated soil. Thus, the soil was under fallow and fertilizers were not applied during 2014 season. The soils in the study area are classified as Orthic Black Chernozems (Udic Boroll), a mix of Hamlin and Blaine Lake association on a gently sloping topography and with a fine sandy loam texture (Soil classification working group, 1998). The site is characterized by remnant native upland areas, wetlands and an undulating topography.

4.4.2 Gas Sampling and Analysis

During the fall of 2012, five transects perpendicular to the shelterbelt were established at various distances along the shelterbelt. The height (H) of a shelterbelt is often used as a standard guide for establishing transects to assess the effect of the shelterbelt on an adjacent field (Kort, 1988; Kowalchuk and de Jong, 1995). For the present study, the transects were set perpendicular to the shelterbelt and parallel to one another at a spacing of 20-m between transects. Each transect consisted of five sampling points located at (i) the center of the shelterbelt (0H; located between the rows of hybrid poplar and caragana); (ii) at the outer edge of the shelterbelt, 5-m from the center location (0.2H); (iii) at the inner edge of the cropped field, 12.5-m from the center of the shelterbelt (0.5H); and (iv) within the cropped field at distances of 40-m (1.5H) and 125-m (5H) from the center of the shelterbelt. Because of the undulating topography, 2 of the 1.5H sampling points were located in depressional areas while the remaining 3 points were located on an upland plain.

Bases for the gas flux chambers were installed during the fall of 2012; gas flux measurements occurred from spring thaw and continued till soil freeze-up in the late fall in both 2013 (May 10th to October 9th) and 2014 (April 22nd to October 15th). The chamber bases were installed to a depth of 5 cm and were used to anchor the flux chambers in place during collection of the gas samples. The gas samples were collected by attaching a flux chamber (22 cm wide × 45.5 cm long × 15 cm tall) to the base and withdrawing 20-mL gas samples immediately afterwards (t_0) and again 20-min later (t_{20}). Each chamber was made of 0.6-cm thick polymethyl methacrylate (PMMA) with a headspace volume of 10 L and a surface area of 1000 cm^2. During chamber installation, litter deposits within the shelterbelt strip were not disturbed or removed; however, all green vegetation within the chamber base was cut before sampling. Except for during seeding and tillage operations, the chamber bases remained in place throughout the study.

Upon deployment, a close-cell polyolefin foam gasket (1-cm thick × 1.2-cm wide) was secured to the underside of the chamber lid to seal against the top edge of base. Gas chambers were vented with a clear flexible vinyl tube (4.8 mm i.d.) attached through an elbow fitting to the cover (Hutchinson and Livingston, 2001). The sampling port consisted of a silicone septum (9.5 mm o.d.) secured by a nylon bolt with a lengthwise opening, which served as a syringe guide. Gas samples were obtained with a 20-mL polypropylene syringe (Monoject™, Luer lock fitting) fitted with a 25-gauge needle, injected into pre-evacuated 12-mL Exetainer™ vials (LabCo Inc., High Wycombe, UK) fitted with butyl rubber stoppers (Rochette and Bertrand, 2008), and transported to the laboratory at the University of Saskatchewan for analysis using gas chromatography.

Gas sampling was carried out twice per week from the start of the spring snowmelt until about four weeks after seeding. Gas sampling intensity was then reduced to once per week during the summer and then to once every two weeks during the fall. At the same time, soil moisture and temperature measurements at 10-cm below ground surface (bgs) were collected from the area immediately adjacent to the gas chambers. Soil temperature measurements were obtained using a stem-style digital thermometer (Reed PS100, Brampton, ON); soil moisture was measured using digital soil moisture meter (HydroSense, Campbell Scientifc, Inc., Logan, UT).

Gas analyses were performed using a Bruker 450 gas chromatograph (Bruker Biosciences Corporation, USA) equipped with a thermal conductivity detector (TCD), flame ionizer detector (FID) and ^{63}Ni electron capture detector (ECD) for the quantification of CO_2, CH_4 and N_2O, respectively (Farrell and Elliot, 2008). Samples were introduced into the chromatograph using a CombiPAL auto-sampler (CTC Analytics AG, Switzerland). Data processing was completed using the Varian Star Chromatography Workstation (ver. 6.2) software. Daily gas fluxes were calculated by fitting a linear regression to the concentration *vs.* time data (Eqn. 1):

$$F_{GHG} = m\frac{V}{A \cdot k_t}$$

where F_{GHG} is the gas flux at t_0 (g m^{-2} d^{-1}); m = slope of the linear regression equation (g L^{-1} min^{-1}); V = volume of the flux chamber (L); A = surface area enclosed by the chamber (m^2); and k_t = time-constant 1440 min d^{-1}). See Chapter 3 for further details of gas analysis techniques.

4.4.3 Soil and root sampling

One soil core (3.2 cm diam.) was collected within the upper 0–30 cm soil depth from each sampling point i.e. beside each gas chamber along all five transects during July 2013, June 2014 and October 2014 to monitor soil NO_3-N and NH_4-N concentrations. During June 2014, soil samples were collected both at the upper (0-30 cm) and lower soil layers (30-100 cm) to determine SOC, TN and pH. Additional soil samples were collected using a hand-held core sampler (5.4-cm i.d. × 3.0-cm long), weighed, dried at 105°C for 24 h, and reweighed to determine the gravimetric soil water content and bulk density. The remaining field-moist samples were air-dried and ground with a rolling pin to break-up any aggregates. A subsample of ground soil (~150 g) was passed through a 2-mm sieve and a 20-g subsample of the sieved soil was placed on a ball grinder for 5 min to create a fine powder (< 250 µm) for total N and SOC analysis.

Soil organic C was determined using a LECO C632 Carbon analyzer (Wang and Anderson, 1998), following a 12*M* HCl pretreatment to remove all inorganic C. Total N was determined by dry combustion using a LECO TruMac CNS analyzer (Figueiredo, 2008). Soil pH in water (1:1 paste; Hendershot et al., 2008) was measured using a Beckman 50 pH Meter (Beckman Coulter, Fullerton, CA, USA). Soil particle size was determined using a modified pipette method (Indorante et al., 1990). Total inorganic N (NO_3-N and NH_4^+-N) were determined using a 2.0*M* KCl extraction (Maynard et al., 2008) with the extracts analyzed colorimetrically (Technicon Autoanalyzer; Technicon Industrial Systems, Tarrytown, NY, USA).

Three root cores (0–30 cm) were collected at each transect using a bucket corer (bucket diameter 5.4 cm) during July 2013 and June 2014. The cores were returned to the laboratory, where they were soaked in water for about 30 min, washed free of soil, and sieved over a 2-mm mesh sieve stacked on top of a 0.5-mm mesh sieve to obtain both small and fine (≤ 5 mm) roots—including both live and dead roots. The samples were then oven-dried (80 ºC for 24 hr) to constant weight and root density (g root biomass m^{-3}) determined.

4.4.4 Statistical Analysis

The effects of shelterbelts on soil properties and gas exchange along the transects from the shelterbelt to the adjacent cropped field were assessed using the SPSS statistical package version 22 (SPSS, 2013). Data were tested for normality and homogeneity of variance using the Shapiro-Wilk test and Levene's test, respectively. When required, a log transformation of the data was used to correct deviations in normality and homogeneity of variance; however, a non-parametric Kruskal-Wallis test was used when data was not corrected by log transformation (Snedecor and Cochran, 1980). One-way analysis of variance was used to test for differences in soil properties and GHG fluxes at various distances from the shelterbelt and Fisher's protected multiple range test (LSD, a = 0.10) was used to compare treatment means. Given the large spatial variation associated with the GHG data, the risk of a type II error in the analysis was considered to be high, although the sample size was relatively large. Consequently, and unless stated otherwise, an alpha value of 0.10 was used to assess statistical significance.

4.5 Results

4.5.1 Environmental conditions

Weather conditions varied during the study, with warmer and drier conditions occurring in 2013 compared to 2014. The mean air temperature was 20% greater in 2013 (15.0ºC) than 2014 (12.4ºC), while total precipitation was about 25% lower in 2013 (331 mm) than in 2014 (439 mm) (Fig. 4.1A; Environment Canada, 2015). In general, soil moisture was greatest during snowmelt in early spring (typically between April and May), and tended to decrease throughout the growing season with an occasional increase following significant precipitation events in both years. Regardless of the inter-annual differences, however, soil water content was consistently greater ($P < 0.05$) in the cropped field (i.e., at locations 1.5H and 5H) than within the shelterbelt (location 0H) or in the transition zone (i.e., locations 0.2H and 0.5H) (Fig. 4.1B).

Seasonal trends in soil temperature were the same in both 2013 and 2014; i.e., the lowest temperatures occurred during early spring and then increased rapidly—reaching a peak in mid-summer (typically between July and August)—followed by a gradual decline during the early fall. There was a distinct temperature gradient along the sampling transects, with the lowest temperatures occurring within the shelterbelt (i.e., at location 0H), but increased with increasing distance from the shelterbelt. In general, mean soil temperatures within the shelterbelt (location 0H) and in the center of the cropped field (location 5H) were significantly ($P < 0.05$) different from the soil temperatures at the other three locations along the transects (Fig. 4.1C).

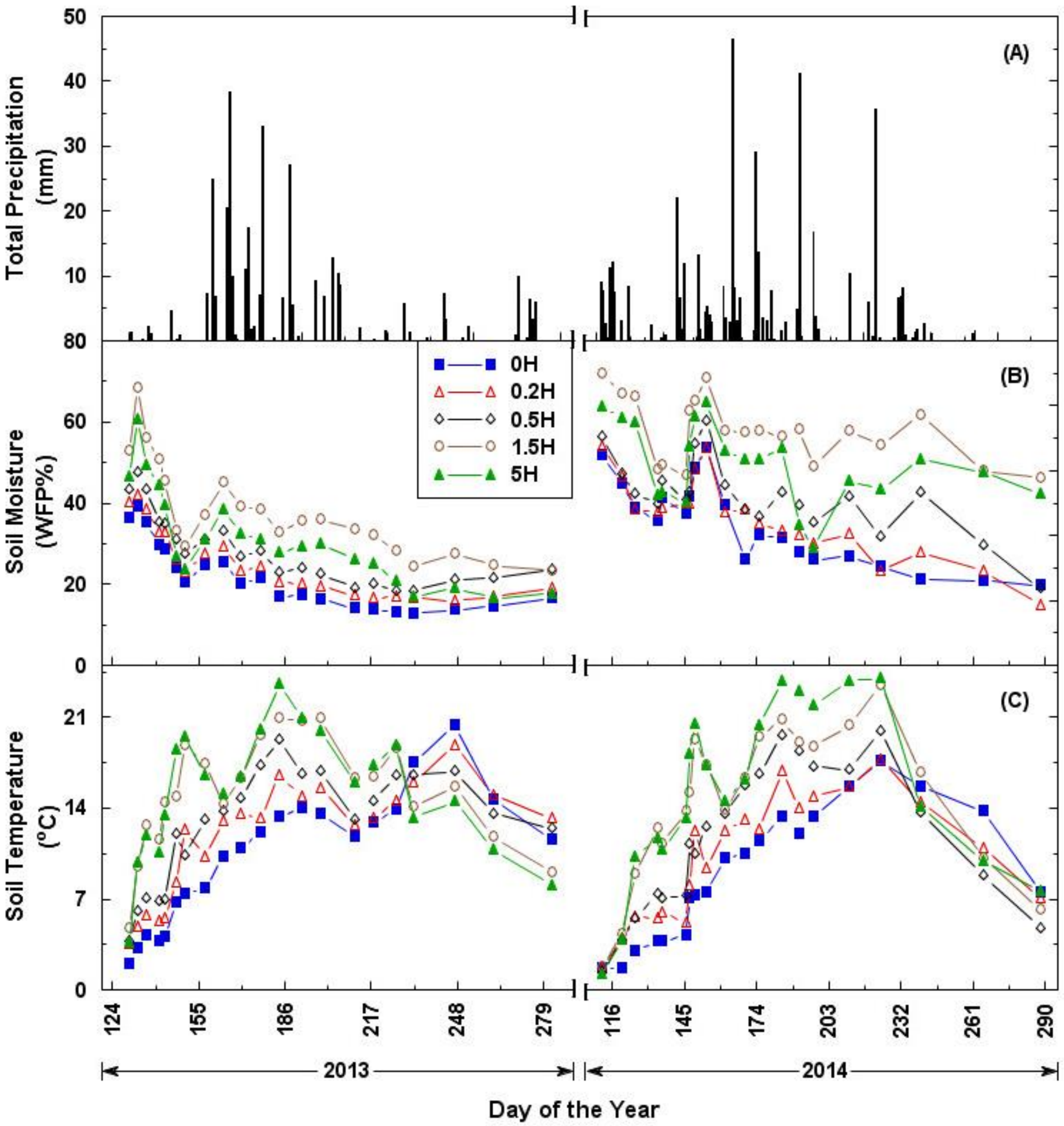

Fig. 4.1 Daily precipitation (A), Water filled pore (WFP) in the surface (0–10 cm) soil (B), and soil temperature (°C) measured at a depth of 10-cm bgs (C). Soil water content and temperature were measured on days when the GHG flux was measured. Note: the plotted data are the mean values for samples collected along replicate (n = 5) transects extending from the center of the shelterbelt into the adjacent cropped field.

Towards the end of the growing season (i.e., from August to October), however, soil temperatures within the shelterbelt and at the inner edge of the transition zone (locations 0H and 0.2H) were warmer than those at sampling locations further away from the shelterbelt.

4.5.2 Soil properties and root biomass along gradient from the shelterbelt

Total organic C concentration in the near-surface (0–30 cm) soils was significantly ($P < 0.05$) greater than that in the subsurface (30–100 cm) soils. As well, TOC within the 0–30 cm depth was greatest at the 0H location and lowest at the 5H location—and there was a general tendency for TOC to decrease as one moved from the shelterbelt into the cropped field (Fig. 4.2A; Table 4.1). Conversely, TOC in the 30–100 cm depth interval did not vary among sampling locations. Similar results were observed for soil pH (Fig. 4.2D), although differences between the near-surface (0–30 cm) and subsurface (30–100 cm) soils were generally negligible. Proximity to the center of the shelterbelt had no significant ($P < 0.05$) effect on either the total soil N content or C:N ratio in either the near-surface or subsurface soils (Fig. 4.2B and 4.2C).

Throughout the study period, concentrations of soil NO_3-N were typically low (1.4–3.5 µg g^{-1}) within the shelterbelt strip, but tended to increase with distance away from the shelterbelt (Table 4.2). Likewise, concentrations of soil NH_4-N tended to be greater in the cropped field (at the 0.5H, 1.5H and 5H locations) than in the shelterbelt (locations 0H and 0.2H) (Table 4.2). In July 2013, soil NO_3-N was significantly ($P < 0.05$) lower at 0H and 0.2H than the other three locations, and was significantly lower at 0.5H than at 1.5H and 5H. During this same period soil NH_4-N was significantly lower at 0H and 0.2H than at the other three sampling locations. In June 2014, soil NO_3-N concentration was significantly ($P < 0.05$) greater in the cropped field (0.5H, 1.5H and 5H) than in the shelterbelt (0H and 0.2H). Soil NH_4-N during this period was significantly greater at 1.5H and lower at 0.2H than the other three locations. Differences in soil NH_4-N and NO_3-N in Oct 2014 were generally small and were not significant (Table 4.2).

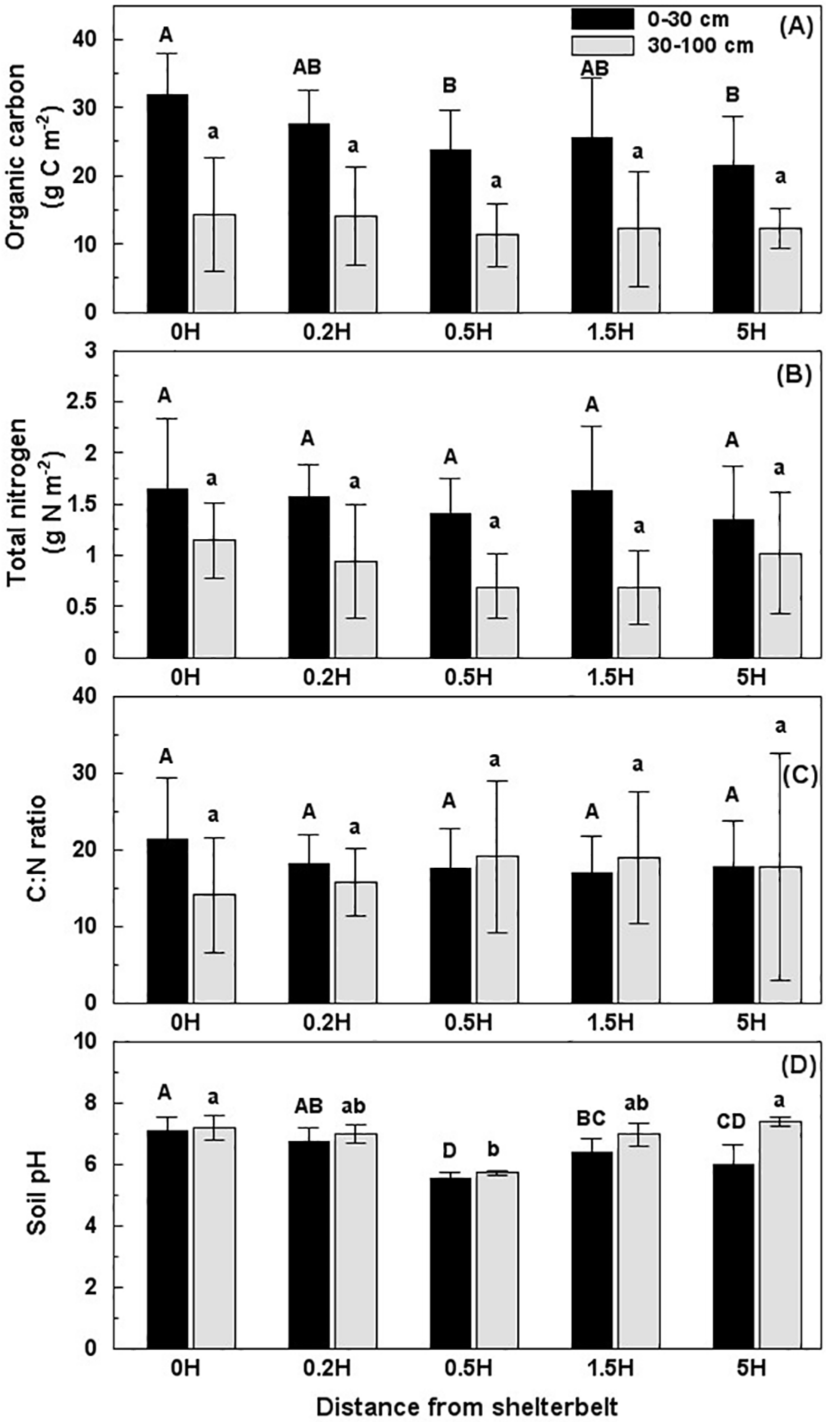

Fig. 4.2 Organic C (A), total N (B), C:N ratio (C), and soil pH (D) measured along replicate (n = 5) transects extending from the center of the shelterbelt into the adjacent cropped field. Soil cores were collected from each flux chamber location along the transects and composited by depth [i.e., 0–30 cm (black bars) and 30–100 cm (gray bars)]. Within depth classes, bars labelled with the same upper or lower case letter are not significantly different.

Table 4.1 Soil organic matter content and bulk density measured at 0-30 cm and 30-100 cm soil depth along the gradient from the shelterbelt to an adjacent field.

Sites†	0-30 cm	30-100 cm
	Organic C mass storage (Mg ha^{-1})	
0H	117.7 A	140.8 A
0.2H	103.9 AB	164.9 A
0.5H	91.3 B	114.0 A
1.5H	96.5 AB	122.3 A
5H	84.1 B	135.1 A
HSD	***6.8***	***39.0***
P	***<0.0130***	***<0.9028***
	Bulk density (Mg m^{-3})	
0H	1.24 B	1.40 A
0.2H	1.26 AB	1.41 A
0.5H	1.29 AB	1.40 A
1.5H	1.27 AB	1.42 A
5H	1.30 A	1.42 A
HSD	***0.02***	***0.07***
P	***<0.01167***	***<0.1923***

† 'H' indicates height of the shelterbelt

Table 4.2 Available soil nitrate (NO_3-N) and ammonium (NH_4-N) within 0-30 cm soil depth measured along replicate (n = 5) transects extending from the center of the shelterbelt into the adjacent cropped field in July 2013, June 2014 and October 2014.

Sites†	**July, 2013‡**	**June, 2014**	**Oct, 2014**
	Soil NH_4-N (µg N g soil^{-1})		
0H	10.2 B	10.7 AB	5.9 A
0.2H	8.3 B	8.8 B	3.3 A
0.5H	12.2 A	12.3 AB	8.2 A
1.5H	13.6 A	14.4 A	7.1 A
5H	12.8 A	12.4 AB	6.2 A
HSD	0.5	1.5	1.6
P	<0.0001	<0.100	<0.3002
	Soil NO_3-N (µg N g soil^{-1})		
0H	1.8 C	2.7 B	1.4 A
0.2H	2.5 C	3.5 B	3.5 A
0.5H	4.9 B	7.5 A	4.8 A
1.5H	7.3 A	8.9 A	3.8 A
5H	8.3 A	7.9 A	4.3 A
HSD	0.6	1.1	1.4
P	<0.0001	<0.0002	<0.4618

† 'H' indicates height of the shelterbelt

‡ On each sampling day, columns labelled with the same letters not significantly different

Small and fine root biomass (≤ 5 mm dia.) in the near-surface (0–30 cm) soils was greatest within the shelterbelt and decreased with increasing distance away from the shelterbelt in both 2013 and 2014 (Fig. 4.3). In 2013, root biomass was significantly ($P < 0.05$) lower at 1.5H and 5H than at 0H and 0.2H; and was intermediate at 0.5H. Inter-annual differences were not significant ($P < 0.05$), and similar trends were observed in 2014. In the cropped field, mean root biomass values were slighltly higher in 2014 than in 2013, even though there was no crop in 2014. The presence of weeds in the field and the inclusion of dead roots as part of root biomass is suspected to have contributed to the higher mean biomass values observed in 2014.

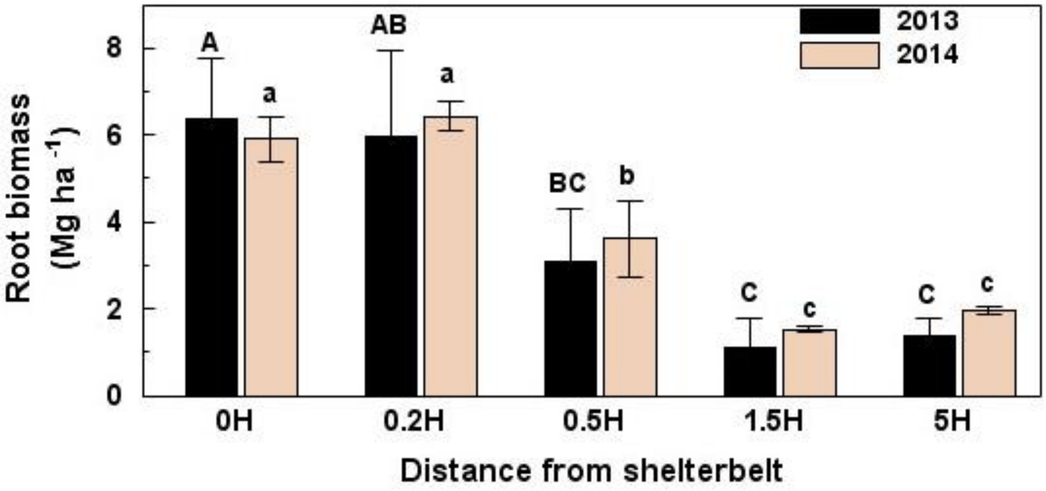

Fig. 4.3 Root biomass (small + fine roots only) at 0-30 cm soil depth measured along transects extending from the center of the shelterbelt (0H) to the center of the adjacent cropped field (5H). Within years, bars labeled with the same upper or lower case letters are not significantly different

4.5.3 Soil CO_2, CH_4 and N_2O exchange in distant gradient from the shelterbelt

Throughout the study period, daily CO_2 fluxes ranged from 0.2 to 110 kg CO_2-C ha^{-1} d^{-1}, but appeared to follow a seasonal trend that paralleled that of soil temperature; i.e., with maximum fluxes occurring during periods when soil temperatures were warmest (i.e., June – August). As well, CO_2 fluxes during the sampling period were generally lowest when soil temperatures were coldest; i.e., during the early spring and fall. Carbon dioxide emissions in the cropped field (at locations 1.5H and 5H) exhibited additional peaks that were associated with soil disturbance during seeding operations (Fig. 4.4A). Indeed, a major CO_2 emission event at seeding in June of 2014 contributed 24% and 17% of the cumulative total CO_2 emissions measured at 1.5H and 5H, respectively. In both 2013 and 2014, cumulative CO_2 emissions were greatest within the shelterbelt (at the 0H location) and lowest near the center of the cropped field (at the 5H location); however, the change in cumulative CO_2 emissions with increasing distance from the shelterbelt did not follow a consistent pattern (Fig. 4.4B). In general, cumulative CO_2 emissions were significantly ($P < 0.10$) greater within the shelterbelt (i.e. 0H; 8032 ± 1123 kg CO_2-C ha^{-1}) than in the transition zone (i.e. 0.2H at 0.5H; 5034 ± 399 kg CO_2-C ha^{-1}) and the cropped field (i.e. 1.5H and 5H; 4610 ± 1722 kg CO_2-C ha^{-1}), whereas there were no significant differences ($P < 0.10$) in cumulative CO_2 emissions between the transition zone and the cropped field. Nevertheless, cumulative CO_2 emissions at distances of 0.2H to 1.5H from the shelterbelt were not significantly ($P < 0.10$) different, but were significantly greater than those at 5H (Fig. 4.7A).

Methane fluxes were generally negative ($\tilde{x}$ = -1.20 g CH_4-C ha^{-1} d^{-1}) indicating a net consumption of atmospheric CH_4 (Fig. 4.5A). Whereas daily CH_4 fluxes did not show any clear seasonal pattern, they were sensitive to changes in soil moisture content along the gradient from the shelterbelt to the adjacent field (see Figs. 4.5A and 4.1B). Cumulative CH_4 fluxes were negative at all but the 1.5H location (Figs. 4.5B and 4.7B)—indicating that the soils were net sinks for CH_4—with CH_4 consumption being greatest within the shelterbelt (i.e., at the 0H and 0.2H locations). In general, the size of the CH_4 sink decreased with increasing distance from the center of the shelterbelt, and net cumulative CH_4 uptake was significantly ($P < 0.10$) greater within the shelterbelt (i.e. 0H; -1447 ± 484 g CH_4-C ha^{-1}) than in the transition zone (i.e. 0.2H at 0.5H; -752 ± 381 g CH_4-C ha^{-1}) and the cropped field (i.e. 1.5H and 5H; -19 ± 342 g CH_4-C ha^{-1}). More so, cumulative CH_4 uptake was significantly ($P < 0.10$) greater within the transition zone than in the cropped field but lower than those within the shelterbelt. A significant CH_4 emission event was observed in the cropped field at the 1.5H location following snowmelt in May 2013 (Fig. 4.5A), and this single emission event accounted for more than 90% of the cumulative annual CH_4 emission at this location. In contrast, significant CH_4 uptake was observed within the shelterbelt (at the 0H and 0.2H locations) during spring snowmelt in May of 2014.

Daily N_2O fluxes exhibited a considerable amount of temporal variability—especially within the cropped field (Fig. 4.6A). Significant N_2O emission events were associated with spring snowmelt, N fertilizer application, and (to a lesser extent) significant precipitation events during the early summer. However, N_2O fluxes measured within the shelterbelt (i.e., at the 0H and 0.2H locations) were less affected by increased spring soil moisture conditions (Fig 4.6A). Soil N_2O emissions following spring snowmelt in May 2013 contributed 50% and 38% of cumulative N_2O emissions at the 1.5H and 5H locations, respectively. Likewise, soil N_2O emissions following spring snowmelt in 2014 contributed 53% and 37% of cumulative N_2O emissions for the 1.5H and 5H locations, respectively. In general, cumulative N_2O emissions were greatest in the cropped field (at 1.5H and 5H) and decreased with increasing proximity to the shelterbelt (Fig 4.6B). Indeed, cumulative N_2O emissions within the shelterbelt (i.e. 0H; 345 ± 34 g N_2O-N ha^{-1}) were significantly ($P < 0.10$) lower than those at the transition zone (i.e. 0.2H at 0.5H; 573 ± 227 g N_2O-N ha^{-1}) and those in the cropped field (i.e. 1.5H and 5H; 1326 ± 732 g N_2O-N ha^{-1}). Further, cumulative N_2O flux at the transition zone (at 0.2H and 0.5H) was significantly different from those at cropped field (at 1.5H and 5H), although differences in N_2O emissions across the cropped area (i.e., at 0.5H, 1.5H and 5H) were generally not significant (Fig. 4.7C).

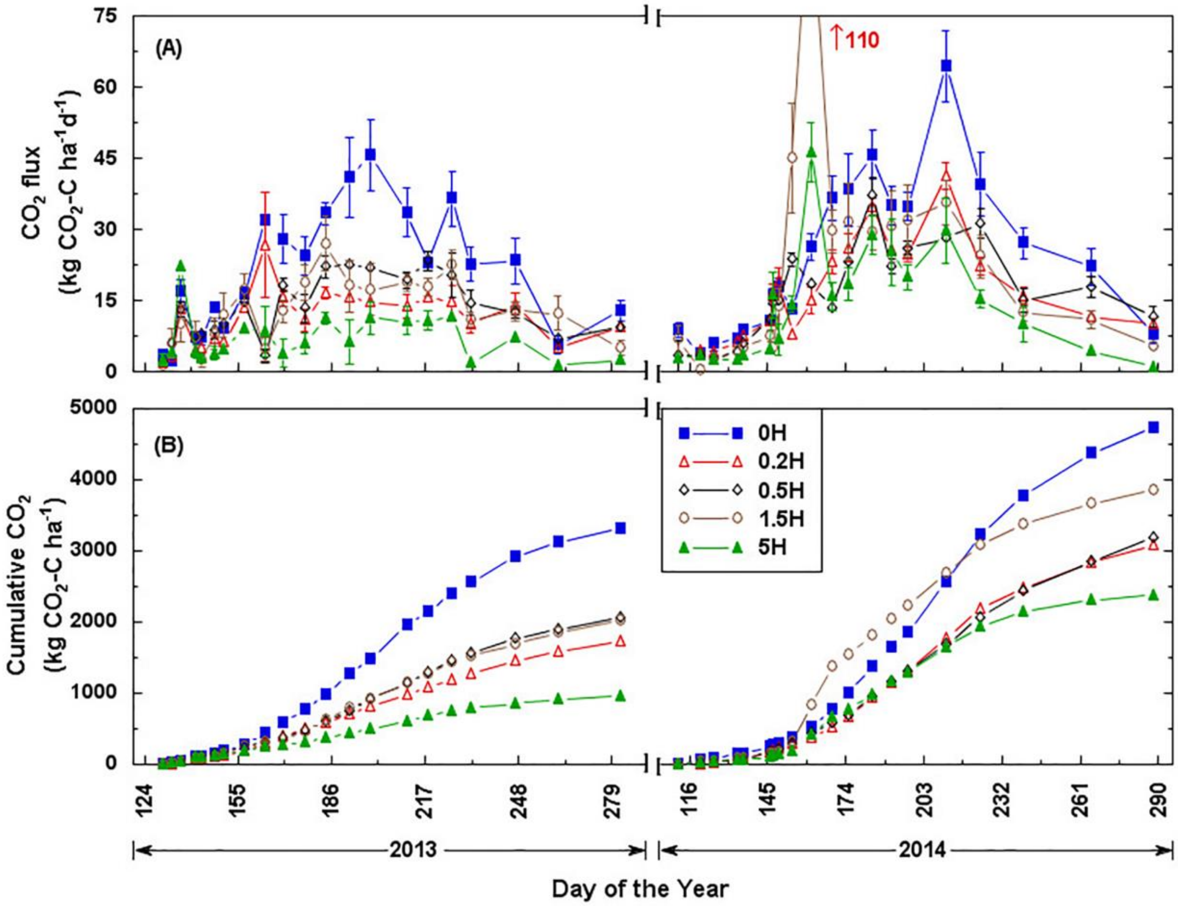

Fig. 4.4 Daily soil CO_2 fluxes (kg CO_2-C ha^{-1} d^{-1}) (A) and cumulative CO_2 emissions (kg CO_2-C ha^{-1}) (B) measured along replicate (n = 5) transects extending from the center of the shelterbelt (0H) to the center of the adjacent cropped field (5H) in 2013 and 2014.

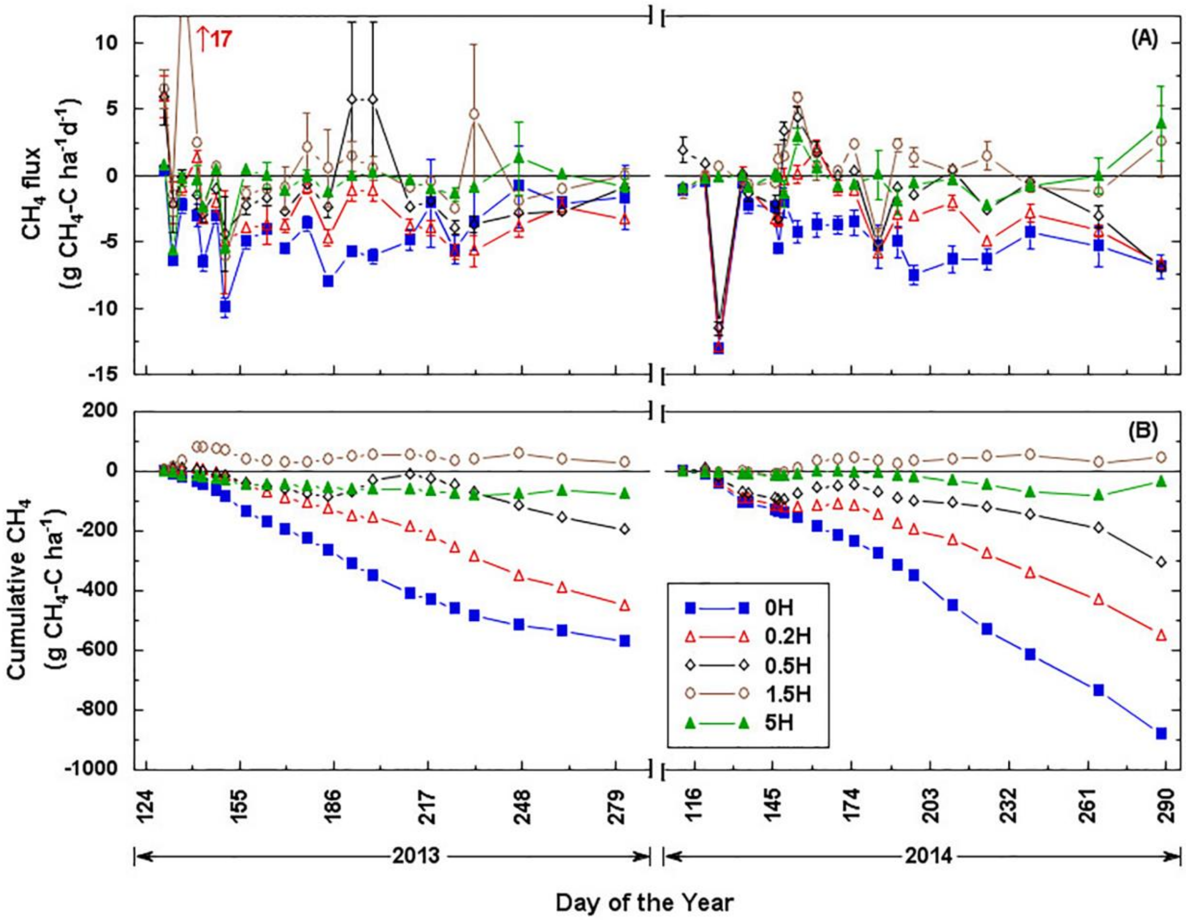

Fig. 4.5 Daily soil CH_4 fluxes (g CH_4-C ha^{-1} d^{-1}) (A) and cumulative CH_4 emissions (g CH_4-C ha^{-1}) (B) measured along replicate (n = 5) transects extending from the center of the shelterbelt (0H) to the center of the adjacent cropped field (5H) in 2013 and 2014.

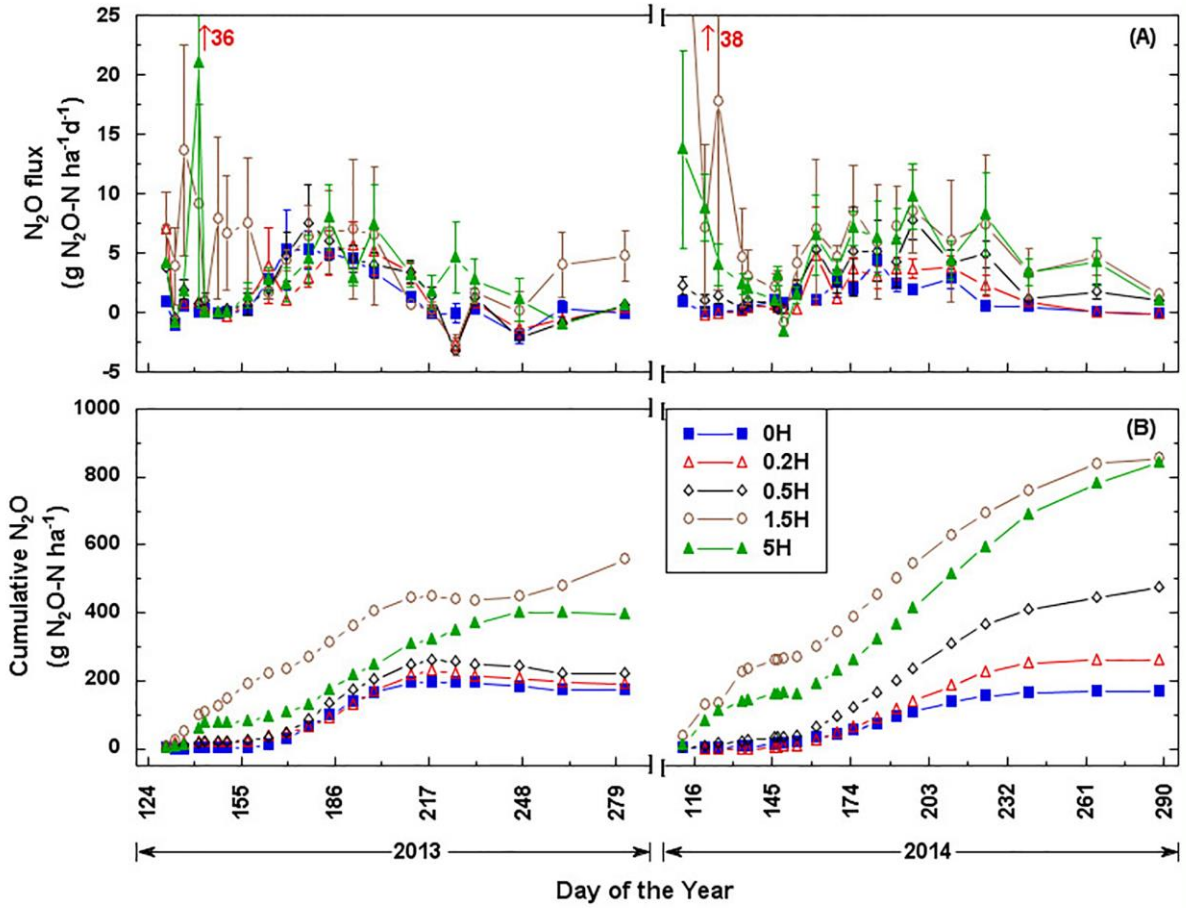

Fig. 4.6 Daily soil N_2O fluxes (g N_2O-N ha^{-1} d^{-1}) (A) and cumulative N_2O emissions (g N_2O-N ha^{-1}) (B) measured along replicate (n = 5) transects extending from the center of the shelterbelt (0H) to the center of the adjacent cropped field (5H) in 2013 and 2014.

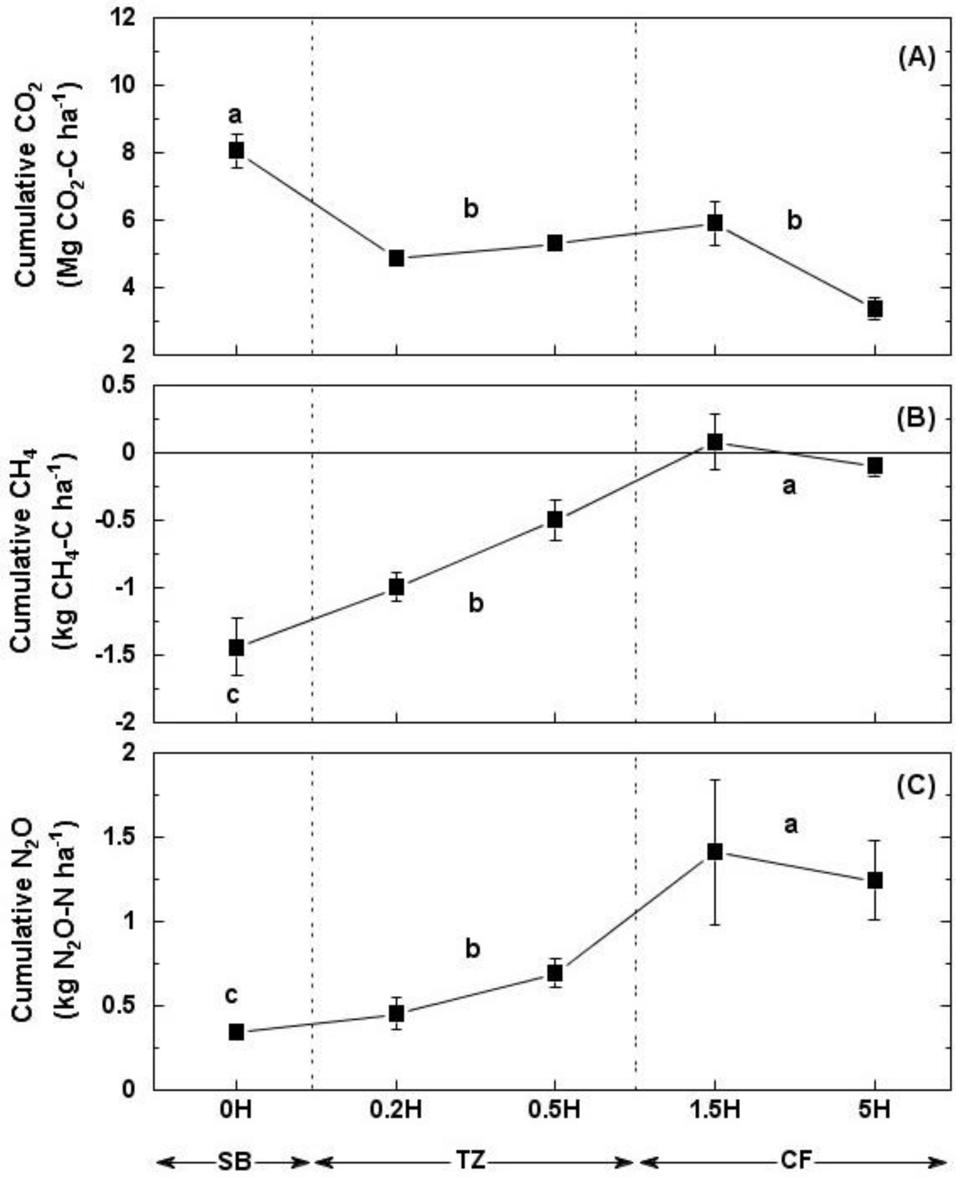

Fig. 4.7 Cumulative fluxes of (A) CO_2 (Mg CO_2-C ha^{-1}), (B) CH_4 (kg CH_4-C ha^{-1}) and (C) N_2O (kg N_2O-N ha^{-1}) measured along replicate (n = 5) transects extending from the center of the shelterbelt [(SB) (i.e. location 0H)] to the transition zone [(TZ) (i.e. locations 0.2H and 0.5H)] and to the center of the adjacent cropped field [(CF) (i.e. locations 1.5H and 5H)] during 2013 and 2014. Error bars represent standard error (n = 5). Negative numbers indicate uptake.

4.5.4 Soil temperature and soil moisture effects on GHG fluxes

In general, there was a weak but significant ($P < 0.01$) correlation between soil-derived CO_2 emissions and soil temperature in 2013 ($r = 0.38$) and 2014 ($r = 0.41$). Along the transect from the center of the shelterbelt to the center of the cropped field, the relationship between soil-derived CO_2 fluxes and soil temperature was strongest within the shelterbelt (i.e., at 0H; $r = 0.82$; $P < 0.01$) followed by the outer edge of the shelterbelt (i.e., at 0.2H; $r = 0.62$; $P < 0.01$). However, no correlations of soil-derived CO_2 emissions and soil temperature were observed in the other three locations. Soil-derived CH_4 fluxes were weakly correlated with soil temperature in 2013 ($r = 0.36$; $P < 0.01$) and 2014 ($r = 0.52$; $P < 0.01$). Like with CO_2 emissions, the relationship between soil-derived CH_4 fluxes and soil temperature was strongest within the shelterbelt (i.e., at 0H; $r = 0.76$; $P < 0.01$) followed by the outer edge of the shelterbelt (i.e., at 0.2H; $r = 0.65$; $P < 0.01$); but no correlations of soil-derived CH_4 emissions and soil temperature were observed in the other three locations. In general, there was no correlation between soil-derived N_2O emissions and soil temperature in both years; however, significant correlations between soil-derived N_2O fluxes and soil temperature were observed at two locations in the cropped field only; i.e., at 0.5H ($r = 0.43$; $P < 0.01$) and 1.5H ($r = 0.37$; $P < 0.05$).

Soil water content was weakly correlated with soil-derived CO_2 flux, but only within the shelterbelt at 0H ($r = 0.31$; $P = 0.05$) and at 1.5H location ($r = 0.60$; $P = 0.01$). Soil-derived N_2O fluxes were weakly correlated with soil moisture in 2013 ($r = 0.31$) and 2014 ($r = 0.40$). Along the transect from the center of the shelterbelt to the center of the cropped field, there was a weak correlation ($r = 0.38$; $P < 0.05$) between soil water content and N_2O flux at the 1.5H position in the cropped field, but not at other locations. Soil moisture was correlated with CH_4 flux at 0H ($r = 0.65$), 0.2H ($r = 0.\ 31$), 1.5H ($r = 0.\ 35$) and 5H ($r = 0.\ 42$), but not at 0.5H.

4.5.5 Soil properties and root biomass controls on cumulative GHG emissions

Along the transect from the center of the shelterbelt to the center of the cropped field, cumulative CO_2 emissions were positively correlated with TOC and total N, and negatively correlated with bulk density (Table 4.3). In general, cumulative CO_2 emissions tended to increase with increasing root biomass—though a significant correlation ($r^2=0.62$; $P < 0.05$) was observed in 2013 only. Cumulative CO_2 emissions also tended to increase with increasing soil pH—though a significant correlation (0.64; $P < 0.05$) was observed only in 2014.

A simple linear regression was performed to estimate CH_4 fluxes based on root biomass. A significant regression equation was found ($r^2=0.94$; $P < 0.05$). Soil-derived CH_4 flux could be predicted from root biomass by the following equation.

$$CH4\ flux = -197.83(root\ biomass) + 18.108 \tag{4.1}$$

Likewise, cumulative CH_4 emissions were negatively correlated with TOC ($r^2 = -0.81$; $P < 0.05$), C:N ratio ($r^2=-0.95$; $P < 0.05$) and soil pH ($r^2 = -0.64$; $P < 0.05$); while at the same time, cumulative CH_4 emissions increased with increasing soil bulk density ($r^2 = 0.74$; $P < 0.05$). However, there was no significant correlation between cumulative CH_4 emissions and total soil N content. Soil-derived N_2O emissions were negatively correlated with root biomass ($r^2 = -0.81$; $P < 0.05$), TOC ($r^2 = -0.59$; $P < 0.10$) and C:N ratio ($r^2 = -0.77$; $P < 0.05$). However, there were no significant correlations between N_2O emissions and either soil pH or total N.

4.5.6 Field-scale estimation of GHG exchange

The total area of the study site was 2.6 ha (i.e. length of shelterbelt = 200-m x length of the transect from the beginning of the shelterbelt to the center of the field =130-m). Within the study site, the total area occupied by the shelterbelt was 0.2 ha, while the area of the transition zone and cropped field were 0.4 ha and 2 ha, respectively. Given these measurements, overall cumulative emissions across the study site was 12991 ± 3840 kg CO_2-C ha^{-1}; where CO_2 emissions from the shelterbelt (1606 ± 225 kg CO_2-C ha^{-1}), transition zone (2165 ± 172 kg CO_2-C ha^{-1}) and cropped field (9220 ± 3444 kg CO_2-C ha^{-1}) represented 12.4%, 16.7% and 71% of net site CO_2 emissions, respectively. Likewise, net site CH_4 uptake was -650 ± 602 g CH_4-C ha^{-1}; and CH_4 uptake from the shelterbelt (-290 ± 96 g CH_4-C ha^{-1}), transition zone (-323 ± 164 g CH_4-C ha^{-1}) and cropped field (-38 ± 342 g CH_4-C ha^{-1}) represented 45%, 50% and 6% of net site CH_4 uptake, respectively. Overall cumulative N_2O emissions across the study site was 2967 ± 1569 g N_2O-N ha^{-1}; where N_2O emissions from the shelterbelt (69 ± 6.9 g N_2O-N ha^{-1}), transition zone (246 ± 98 g N_2O-N ha^{-1}) and cropped field (2652 ± 1465 g N_2O-N ha^{-1}) represented 2.3%, 8.3% and 89.4% of net site N_2O emissions, respectively.

Table 4.3 Correlation matrix of cumulative CO_2, CH_4 and N_2O with site characteristics, i.e. root biomass, organic C, total N, bulk density and pH along the gradient from the shelterbelt to an adjacent field.

	Cumulative CO_2	Cumulative CH_4	Cumulative N_2O	Root biomass	TOC	TN	C:N	BD	pH
Cumulative CO_2	—								
Cumulative CH_4	-0.539	—							
Cumulative N_2O	-0.008	0.771**	—						
Root biomass	0.399	-0.943***	-0.811**	—					
TOC	0.639*	-0.808**	-0.587†	0.820**	—				
TN	0.570†	-0.428	-0.240	0.481	0.847**	—			
C:N	0.513	-0.946***	-0.766**	0.921***	0.853**	0.446	—		
BD	-0.614†	0.740*	0.521	-0.774**	-0.987***	-0.912***	-0.769**	—	
pH	0.444	-0.642*	-0.383	0.681*	0.872**	0.814**	0.660*	-0.906***	—

†, *, **, *** Significant at the 0.10, 0.05, 0.01, and 0.001 level of probability.

4.6 Discussion

Soil CO_2 fluxes observed in the present study were within the range of those reported in forested (Ball et al., 2007; Peichl et al., 2010) and cropped (Mosier et al., 2006; Ellert and Janzen, 2008) systems in temperate soils. The strong positive relationship between soil temperature and CO_2 emissions across locations at varying distances from the shelterbelt highlights the role of temperature as a major driver of biological processes that result in the production and release of CO_2 from the soil. Temperature controls on soil CO_2 emissions have been reported in other studies (Anderson, 1973; Akinremi et al., 1999; Parkin and Kaspar, 2003; Smith et al., 2003; Peichl, 2010).The greater cumulative CO_2 emissions observed within the shelterbelt (at 0H) compared to the other four sampling locations was attributed to tree root respiration during active growth periods and microbial decomposition of tree litter and residues that were enhanced by a modified climate. Locations nearest to the shelterbelt had the greatest amount of small and fine root biomass (Fig. 4.3), which may have contributed to the greater CO_2 emissions observed within the shelterbelt.

Modified microclimate in soils under shelterbelts have been reported to enhance soil biological activity and protect soil microfauna from temperature and moisture stress (Wojewoda and Russel, 2003; Martius et al., 2004). Thus, the modified soil microclimate generated by the combined effect of tree canopy and root activity in the shelterbelt is likely to have resulted in enhanced activity of the soil microbial communities within and nearest the shelterbelt. Consequently, the enhanced microbial activity in the shelterbelt zone may have contributed to the increased CO_2 emissions observed at the 0H location. Jenkinson (1977) referred to the soil microbial community as “the eye of the needle through which all organic materials must pass” thus intimating the key role that soil microorganisms play in SOM turnover and C sequestration. The activities of soil microorganisms are regulated by soil temperature, moisture aeration and organic matter availability and quality (Mosier et al., 2006). Thus the strong significant relationship between CO_2 emissions within the shelterbelt and soil temperature and moisture may be indicative of rapid microbial decomposition of organic materials within the shelterbelt, especially during the summer period characterized by increased soil temperature and moisture. This is in agreement with Lee and Jose (2003) who studied soil respiration and microbial biomass in soils under agroforestry. They reported that planted trees produced a modified micro-environment that had a positive effect on soil respiration, resulting in increased microbial biomass and soil C storage.

There was a strong correlation between cumulative CO_2 emissions and TOC (Table 4.3) suggesting that increased SOM turnover is associated with increased soil CO_2 fluxes along the gradient from the shelterbelt to the field. This is similar to other reports of strong correlations between SOM and soil CO_2 emissions in both laboratory incubations (Mallik and Hu, 1997) and field studies (Franzluebbers et al., 1995; Lee and Jose, 2003). Consequently, the greater soil CO_2 emissions from the shelterbelt strip are not necessarily indicative of net ecosystem C loss since SOC content was highest in this zone (compare Figs. 4.2A and 4.7A); instead, it may be indicative of the greater rate of SOC turnover, root respiration and decomposition of organic materials by soil microbial community relative to sampling locations in the cropped field.

Soil CH_4 fluxes observed in the present study conform to reported values in forested (Matson et al., 2009; Peichl et al., 2010) and cropped (Mosier et al., 2006; Kim et al., 2010) systems in temperate soils. Although there was no correlation between daily soil moisture and CH_4 fluxes at individual locations, cumulative CH_4 exchange at the gradient from shelterbelt to the field center was inversely related to soil moisture content (Figs. 4.1B and 4.7B). This finding is in agreement with numerous studies that have reported increasing CH_4 uptake with decreasing soil moisture (Castro et al., 1994; Rosenkranz et al., 2006; Matson et al., 2009). Soil temperature exhibited a similar relationship with soil CH_4 exchange as soil moisture (Figs. 4.1C and 4.7B). However, several studies have found temperature controls on soil CH_4 to be of limited importance, with controls on gas diffusivity such as soil moisture, texture, bulk density and root biomass being of relatively greater significance (Bubier et al., 2005; Ball et al., 2007; Smith et al., 2003; Peichl et al., 2010).

Methane production in soil involves the microbial breakdown of organic compounds in strictly anaerobic conditions and at very low redox potential. Thus, the greater cumulative CH_4 uptake observed within the shelterbelt relative to other locations can be explained by the lower soil moisture in within the shelterbelt. The observed strong correlations between cumulative soil CH_4 exchange and root biomass, bulk density and TOC conforms with earlier studies suggesting that soil properties that affect soil moisture levels and gas diffusivity will also affect soil CH_4 exchange (Smith et al., 2003; Peichl et al., 2010). Increasing root biomass with increasing proximity to the shelterbelt reflects greater soil water uptake, improved soil aggregates and SOM additions in deeper soil layers due to root senescence. Moreover, the continuous addition and decomposition of tree litter increases SOM content, causing the soil bulk density at locations nearest to the shelterbelt to be lower. Thus, the greater soil methane uptake within the shelterbelt was perhaps a product of increased soil moisture uptake by tree roots and reduced soil bulk density within the shelterbelt.

Daily soil N_2O fluxes that were observed fall within the range reported in forested (Kim et al., 2009; Peichl et al., 2010) and cropped (Mosier et al., 2006; Ellert and Janzen, 2008) systems. The increase of N_2O emissions along the transect from the shelterbelt to the adjacent field was correlated positively with soil temperature, moisture and available N, but was negatively related to root biomass. Thus, the lower N_2O emissions within the shelterbelt compared to other locations may be attributed to reduced microbial N transformations following root uptake of excess water and available N. Beaudette et al. (2010) reported a significant decrease in soil moisture and a corresponding increase in N_2O emissions that were three times greater in monocropping systems than in tree-based intercropping (TBI) systems. The observed relationships between N_2O emissions and root biomass, available-N, soil temperature and moisture along the transect agree with other studies that have shown strong controls of these soil properties on soil N_2O emissions (Smith et al., 2003; Izaurralde et al., 2004; Mosier et al., 2006; Ball et al., 2007).

The significant difference in CO_2 emissions within the shelterbelt and the transition zone indicate distinct changes in soil properties from the location within the shelterbelt to the outer edge of the adjacent cropped field. For example, although SOC and root biomass did not differ markedly between the shelterbelt (i.e. 0H) and the outer edge of the shelterbelt (i.e. 0.2H), these soil properties were substantially lower at the outer edge of the cropped field (i.e. 0.5H) relative to the location at 0H. However, the lack of any significant difference in CO_2 emissions between the transition zone and the cropped field may have been due to the increase in CO_2 emissions resulting from topographic effects and shelterbelt-induced microclimate at the 1.5H location.

Cumulative CH_4 uptake at the transition zone was significantly lower than within the shelterbelt, but was significantly greater than within the cropped field, indicating that the transition zone was indeed an intermediate position between the greater CH_4 uptake within the shelterbelts and lower uptake in the cropped field. Concurrently, soil bulk density and root biomass tended to align with CH_4 exchange within the transition zone.

Although there was no significant difference in cumulative N_2O emissions from 0H to 0.5H, cumulative N_2O emissions were significantly greater at the transition zone (i.e. average at 0.2H and 0.5H) than within the shelterbelt at 0H. Moreover, cumulative N_2O emissions at the transition zone were significantly lower than those in the cropped field (i.e. average at 1.5H and 5H). These results are in agreement with available N and soil moisture data indicating that the soil emitted more N_2O as one moved from the shelterbelt into the cropped field and the transition zone was an intermediate position.

The small cumulative CO_2 emissions observed at 5H may be indicative of lower turnover of SOM and lower root respiration, which is reflected by the lower TOC and root biomass at this location (Figs. 4.2A and 4.7A). Unlike locations nearest to shelterbelts where CO_2 emissions were mainly due to decay of freshly added litter and root respiration, CO_2 emissions from the field were mainly due to soil disturbance by farm machines during seeding operations. Consequently, CO_2 emissions from the locations in the cropped field occurred mainly due to soil exposure and oxidation of already sequestered soil C following soil disturbance. Agronomic practices such as tillage bring about the breakup of soil-aggregate complexes, which are important in stabilizing SOM; leading to soil C depletion, especially in fields planted to annual crops characterized with low root production (Rutberg et al., 1996).

The larger CO_2 emissions observed at 1.5H relative 5H could be attributed, at least in part, to topographic effects at the 1.5H location. Two out of five replicate sampling points at the 1.5H location were located on depressional landscape positions characterized by increased soil moisture content and SOM accumulation resulting in greater CO_2 emissions compared to sampling points located on the plain. Another possible explanation for the larger CO_2 emissions at the 1.5H may be increased crop productivity and an increase in SOM (Fig 4.2A) resulting from shelterbelt-induced micro-climatic conditions at this location. This is consistent with earlier studies that reported elevated SOM and available N and enhanced crop yield at the zone from 1.5H to 3H; attributed to shelterbelt induced reduction in evaporation and wind speed at this zone (Kort, 1988; Kowalchuk and de Jong, 1995; Wojewuda and Russel, 2003).

The lower CH_4 uptake observed at locations within the cropped field (i.e. 1.5H and 5H) was correlated with lower SOM and root biomass and increased soil bulk density (Table 4.1; Figs. 3 and 4.7B). The continuous use of heavy machines during fertilizer application and other agronomic practices within the cropped fields may have increased soil bulk density due to soil compaction, resulting in lower soil CH_4 uptake. Earlier studies have shown that agronomic practices such as tillage, fertilization and use of pesticides and herbicides have various degrees of inhibitory effect on CH_4 uptake in arable lands (Hansen et al., 1993; Arif et al., 1996; Mosier et al., 1997; Powlson et al., 1997; Topp and Pattey, 1997; Hütsch, 2001).

Dry lands are an important sink for atmospheric CH_4 as they contribute up to 15% of annual global CH_4 uptake (Powlson et al., 1997). However, management practices applied on the land could increase or decrease soil CH_4 uptake. While the intensification of crop production decreases the soil sink size for atmospheric CH_4 (Bronson and Mosier, 1993); incorporation of trees into cropped fields could significantly increase soil CH_4 sink size through the removal of excess soil moisture, increase in SOM and decrease in soil bulk density. Kim et al. (2010) studied soil CH_4 fluxes in riparian forest buffers and reported significant increments in soil TOC, TN, pH and decrease in soil bulk density within the riparian forest system compared to the crop field, suggesting that the incorporation of trees into croplands progressively modifies the soil into larger soil CH_4 sinks.

The largest N_2O emissions occurred within the cropped field (i.e. at 1.5H and 5H), and were mainly due to N fertilization during seeding operations (i.e. in 2013) and increased soil moisture especially during spring melt. This is in conformity with the results on available N and soil moisture, which had greatest amounts within the cropped field. Soil N_2O emissions from the cropped field was exacerbated by the lower root biomass of the field crops, which may not have been as effective in removing excess soil moisture and available N as the roots of the shelterbelt trees. Indeed, although increased N_2O emissions during early spring snowmelt and fertilization during seeding operations contributed largely to seasonal N_2O emissions at locations within the cropped field (i.e. 1.5H and 5H), sampling locations nearest to the shelterbelt (i.e. 0H, 0.2H) were less affected by snowmelt events (Fig. 4.6). High N_2O fluxes in the cropped field following early spring thaw and after fertilization have been observed in other studies (Mosier et al., 2006; Sharma et al., 2006; Chapuis-Lardy et al., 2007; Ellert and Janzen, 2008) and the presence of available N in saturated soils have been implicated as major drivers.

4.7 Extent of shelterbelt effect on greenhouse gas emissions in adjacent cropped field

The extent of shelterbelt influence on soil GHG exchange in adjacent cropped fields appear to hinge on tree root distribution and activity, and shelterbelt-induced micro-climatic conditions created mainly through shading from solar radiation and wind speed. Tree roots function in enhancing SOM in deeper soil layers, providing energy for soil microbial communities, recycling nutrients from the subsoil below the crop rooting zone, reducing nutrient leaching and improving soil physical properties (Scroth, 1995). The ability of tree roots to remove excess soil moisture and available N resulted in significant decrease in soil N_2O emissions and increase in soil CH_4 sink size. However, tree roots may severely compete with crops for water and soil N, which may affect total yields (Kort, 1988).

Sheltering of adjacent soils from solar radiation by shelterbelts may have induced varying levels of the observed changes in GHG emissions at various distances away from the shelterbelt. Sheltered soils are characterized by lower, but less fluctuating soil temperatures and reduced evaporation (Long, 1984). This helps to protect soil microorganisms, which take part in a myriad of processes that influence soil GHG exchange, from temperature and moisture stress (Martius et al., 2004). However, the area under shelter from solar radiation may vary depending on the time of the day and orientation of the shelterbelt (Brandle et al., 2004).

My data indicates that the effect of the shelterbelt on soil N_2O and CH_4 exchange in adjacent soils appeared to extend to the location at 1.5H, beyond which no further significant shelterbelt-induced changes in GHG exchange were observed. Tree roots were reported to be limited beyond 1.5H, although shelterbelt-induced microclimate was at its peak at this zone (Kowalchuk and de Jong, 1995). During the entire study period, N_2O emissions were reduced by 42% from 0H to 1.5H relative to the field center (5H). Similarly, soil CH_4 oxidation was increased by 96% from 0H to 1.5H relative to the location at the center of the field.

The changes in CO_2 emissions at various distances from the shelterbelt as observed in this study align closely with changes in crop yield at various distances from shelterbelts as reported by Kort (1988). Further, the dynamics of CO_2 emissions from 0H to 1.5H show patterns of long-term additions of tree litter and root remains and reduced soil disturbance following tree establishment. The large network of tree roots that extend into the adjacent cropped field imply that there is almost a continuous flux of organic materials to the soil C pool, a portion of which decays and contributes to CO_2 emissions and another portion that is resistant and contributes to soil C sequestration (Kabba et al., 2007).

4.8 Conclusion

This study provides the first insights to the dynamics of GHG exchange along the gradient from a shelterbelt to the center of the field. The soil properties proximal to shelterbelts are progressively modified to enhance C content and reduce nitrous oxide emissions, while maintaining a strong methane sink. In the present study, shelterbelt influences on soil GHG emissions at various distances from the shelterbelt appeared to extend up to 1.5H, beyond which no significant shelterbelt-induced effects were observed. However, the spatial extent of shelterbelt-induced effect on soil GHG exchange may vary in shelterbelts composed of other tree species. To better understand the role of shelterbelts in mitigating agricultural GHG emissions, there is a need for long-term studies with shelterbelt systems of various species, age and designs.

5. PREDICTING GREENHOUSE GAS MITIGATION FROM SHELTERBELTS ON A PRAIRIE FARM USING THE HOLOS MODEL[2]

5.1 Preface

Shelterbelts have been recognized for their role in C sequestration in soils and in the above- and belowground biomass of trees. More so, as discussed in previous chapters, net GHG emissions in soils under shelterbelts are significantly lower than in soils of adjacent fields. Yet, two pertinent questions arise: (a) by how much do shelterbelts reduce GHG emissions from a whole farm during a specified period of time? And (b) what shelterbelt tree species are more effective in mitigating GHG emissions. Data detailing the role of shelterbelts in mitigating GHG emissions and the contribution of tree species may create new opportunities for land owners to explore the option of reducing their GHG emissions foot print using shelterbelt systems. Indeed, a quantitative assessment on the effect of planted shelterbelts on farm GHG emissions is needed for a more accurate estimation of the environmental and economic benefits of shelterbelt establishment, which in turn will support policy and management decisions on shelterbelt systems. The objective of the study was to assess the impact of three tree species and five levels of shelterbelt establishment on the global warming potential of a model farm during 60 years of cultivation. To achieve this objective, the HOLOS model and ancillary calculations were used to estimate C sequestration in tree biomass and soil, C losses to the atmosphere through microbial decomposition, CH_4 and N_2O fluxes, and CO_2 emissions from farm energy use.

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[2] This chapter, co-authored with Dr. Richard Farrell and Prof. Ken Van Rees, has been submitted to the Canadian Journal of Soil Science. While the data analyses and initial write-up was completed by the lead author (Chukwudi Amadi), editing and reviews of the manuscript was carried out by the co-authors.

5.2 Abstract

Planted shelterbelts have been promoted as a strategy for carbon (C) sequestration and mitigation of agricultural greenhouse gas (GHG) emissions. However, the on-farm role that shelterbelts have on GHG emissions has not been fully explored. Thus, the objective of this study was to examine the impact of three tree species [hybrid poplar (*Populus spp*), white spruce (*Picea glauca*) and caragana (*Caragana arborescens*)] at five planting densities on GHG mitigation for a model farm (cereal – pulse rotation) using the Holos model. The planting densities of the shelterbelts represented 0%, 0.6%, 1.2%, 3.0% and 5.0% of the total farm area. The Holos model was used to estimate C sequestration in the tree biomass, crops and soil, C losses to the atmosphere through microbial decomposition, CH_4 and N_2O fluxes, and CO_2 emissions from farm energy use over a 60 year time frame. The greatest reduction in farm GHG emissions was predicted for hybrid poplar (33,226 Mg CO_2e; 23.0%) followed by white spruce (25,194 Mg CO_2e; 17.5%), and caragana (11,897 Mg CO_2e; 8.2%) at the highest planting density. The greater GHG mitigation potential estimated for hybrid poplar shelterbelts was attributed to greater biomass production and consequently, more rapid C input to soil through litter fall and root turnover compared to white spruce and caragana. Between 90 to 95% of GHG mitigation by all shelterbelts was through C sequestration in tree biomass and in stable SOC pools, while reduction in N_2O emissions contributed 5.1 – 9.6% of total GHG mitigation by shelterbelts. Increased CH_4 oxidation contributed only 0.002 – 0.12% while reduction in CO_2 emissions from farm energy use contributed 1.5 – 4.2% of total GHG mitigation by shelterbelts, depending on the species and planting density of the shelterbelts. The GHG predictions from Holos indicate that species selection will be important for maximizing C sequestration and GHG mitigation potential from shelterbelt systems; conversely shelterbelt removal from the agricultural landscape will increase net on-farm GHG emissions.

5.3 Introduction

The removal of atmospheric C and its storage in the terrestrial biosphere is one option that has been proposed to mitigate greenhouse gas (GHG) emissions (IPCC, 2001). Arable lands, therefore, present a viable opportunity for removing large amounts of atmospheric GHGs if trees are incorporated into farming systems (Albrecht and Kandji, 2003; Evers et al., 2010; Nair, 2011). Consequently, the benefits of incorporating trees into agricultural landscapes is receiving wider recognition not only in terms of agricultural sustainability but also in issues of climate change (Albrecht and Kandji, 2003; Lorenz and Lal, 2014). Shelterbelts (also known as windbreaks) are linear arrays of trees and shrubs planted to alter environmental conditions in agricultural systems while providing a variety of economic, social and ecological benefits valued by land owners (Mize et al., 2008) including C sequestration (Kort and Turnock, 1999).

Despite the relatively small land area they occupy on the agricultural landscape, shelterbelts can sequester large amounts of C per unit area. For example, potential C sequestration rate in above- and belowground components in shelterbelt systems have been estimated at 6.4 Mg C ha^{-1} yr^{-1}, compared to 2.6, 3.4 and 6.1 Mg C ha^{-1} yr^{-1} for riparian forest buffers, alley-cropping and silvopasture systems, respectively (Udawatta and Jose 2011). In Saskatchewan Canada, Kort and Turnock (1999) estimated C sequestration in aboveground biomass of 17–90 year-old, single row shelterbelts at 105, 24–41, and 11 Mg C km^{-1} for hybrid poplar, conifer, and shrub shelterbelts, respectively.

Planted shelterbelts also increase C sequestration in stabilized SOC pools, although at a limited spatial scale on the landscape (Udawatta and Jose, 2011). In Nebraska USA, Sauer et al. (2007) reported that SOC concentrations in the 0-7.5 cm soil layer under a red cedar (*Juniperus virginiana*) – Scots pine (*Pinus sylvestris*) shelterbelt (3.04%) was 55% greater than in the adjacent cultivated field (1.96%); with 12% greater SOC in the 7.5-15 cm soil depth. Thus, during a period of 35 years, SOC sequestration in the shelterbelts within the 0-15 cm soil depth was 3.71 Mg greater than that in the cropped field, representing an average annual difference of 0.11 Mg ha^{-1}. The greater SOC content in the shelterbelts was attributed to increased inputs from tree litter and wind-blown sediments, reduced soil disturbance from agronomic practices and reduced soil erosion.

The potential of atmospheric CO_2 reduction by agroforestry systems occurs not only through carbon accumulations in tree biomass and soil, but also through various indirect benefits associated with agroforestry. For example, planting shelterbelts reduces farm energy since the areas occupied by trees are exempt from fertilizer application and other agronomic practices such as tillage and pesticide applications. This not only implies a reduction in N_2O emissions, but also a reduction in CO_2 emissions from diesel use and during the manufacture of fertilizers and pesticides (Brandle et al., 1992; Little et al., 2008). Other indirect benefits include: enhanced storage of C in wood products and the substitution of wood for fossil fuel, which reduces the need for unsustainable deforestation (Roy, 1999; Montagnini and Nair, 2004).

Integrating trees into the agricultural landscape has been reported to reduce soil N_2O emissions and increase CH_4 oxidation (Beaudette et al., 2010; Evers et al., 2010; Chapter 3). Trees are deep rooting and can inhibit the denitrification process in the soil profile by taking up residual N and excess soil water that would otherwise be available for N_2O emission or NO_3 leaching; and eventually return them back to the soil through litterfall. This process is recognized as the safety-net role of tree roots (Allen et al., 2004), and the result is more efficient N cycling and reduced N_2O emissions from N fertilization (Thevathasan et al., 2012). As well, the production of CH_4 has been positively correlated with elevated soil moisture (Smith et al., 2003). However, the ability of tree roots to take up excess moisture and N in surrounding soils (Beaudette et al., 2010) can create favourable conditions for methane oxidation, which in turn, increases the size of methane sink in soils under treed systems (Suwanwaree and Robertson, 2005; Matson et al., 2009).

While shelterbelts have started to be recognized for their potential in mitigating GHG emissions through C sequestration, increased CH_4 oxidation and reduced N_2O emissions (Amadi et al., 2016), there are no quantitative assessments of the magnitude of GHG mitigation by shelterbelts at the farm scale. Such estimates, however, are needed for a more accurate estimation of the environmental and economic benefits of shelterbelts and, in turn, can be used to develop policy and management decisions on the use of trees in agricultural systems. The objective of this study, therefore, was to use the Holos model to examine the impact of three tree species—hybrid poplar (*Populus spp*), white spruce (*Picea glauca*) and caragana (*Caragana arborescens*)—at five planting densities on GHG mitigation for a model farm.

5.4 Materials and Methods

5.4.1 Model scenarios

Based on the 2011 agriculture census (Saskatchewan Ministry of Agriculture, 2012), there are approximately 26,900 cultivated farms in Saskatchewan Canada, with an average farm size of 688 ha. Thus, GHG emissions were simulated for a cereal-pulse rotation with varying levels of planted shelterbelts for a model farm of 688 ha. Three commonly cultivated crops: wheat (*Triticum aestivum*), dry peas (*Pisum sativum*) and oats (*Avena sativa*) were selected using a continuous wheat-pea-oats rotation, with reduced tillage (i.e. few tillage passes with most residue retained on the surface) and moderate input crop management practices.

At the provincial scale, it is estimated that there are more than 60,000 km of planted shelterbelts in Saskatchewan, which are comprised of different tree and shrub species and are planted in various designs and orientations (Amichev et al., 2014). However, at the farm-scale the amount of planted shelterbelts within an individual farm unit can vary considerably ranging from no shelterbelt (i.e. zero shelterbelt planting) to many rows of tree planting, reaching up to 5% of the total farm area (Stoeckeler, 1965; Kort, 1988; Schoeneberger, 2009). Three common shelterbelt species: hybrid poplar, white spruce and caragana were used to simulate the overall GHG emissions from the model farm for a period of 60 years (i.e. from the first year of tree planting to 60 years after planting). In order to account for the range of variations in the amount of planted shelterbelts on a typical farm, I considered five different scenarios of single row shelterbelts [0% (baseline), 0.6%, 1.2%, 3.0% and 5.0% of the total farm area] (Table 5.1).

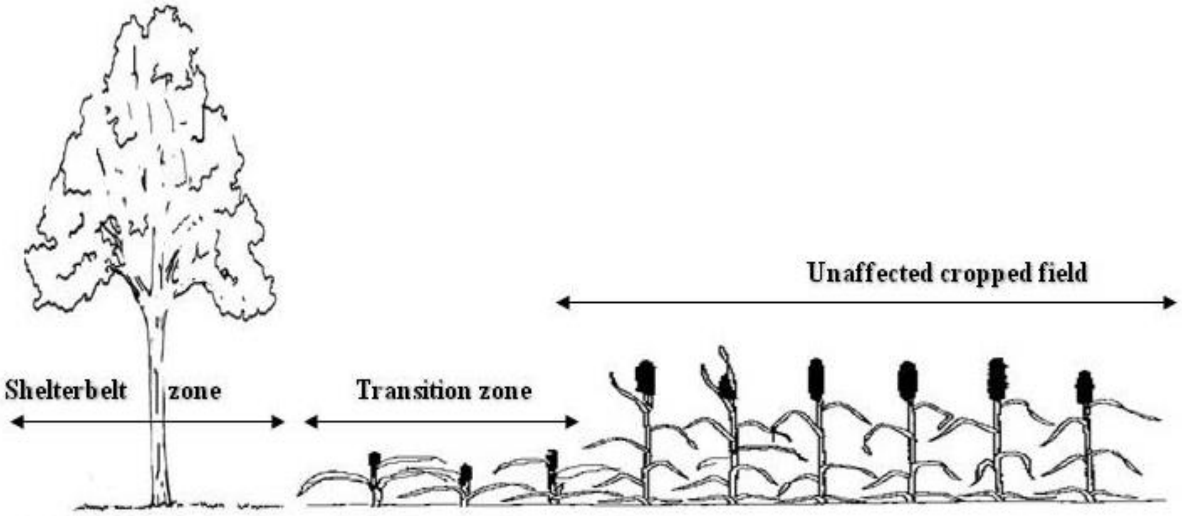
Table 5.1 Summary of scenarios representing percent area of a farmland planted to shelterbelts and the area of cultivated land in a 688 ha hypothetical farm

Scenario number	Shelterbelt area (%)	Shelterbelt area (ha)	Cropped area (%)	Cropped area (ha)
1	0	0	100	688.0
2	0.6	4.1	99.4	683.9
3	1.2	8.3	98.8	679.7
4	3.0	20.6	97	667.4
5	5.0	34.4	95	653.6

5.4.2 Farm zones

Three major zones were identified for simulating GHGs on the farm: the shelterbelt, transition between the shelterbelt and cropped field, and the cropped field itself. The shelterbelt zone is the area under the crown width of the linear planted shelterbelts. Crown width values of 14.04, 7.86 and 9.49 m were used for hybrid poplar, white spruce and caragana, respectively (Amichev et al., 2015). Shelterbelts were assumed to be in good condition and the soil in the shelterbelt area was undisturbed and excluded from agronomic activities such as tillage, fertilizer application and seeding.

The transition zone is the area that is indirectly influenced by shelterbelts, e.g., by shading, root activity, litter depositions and micro climatic influences. The transition zone width is 1.5 times the shelterbelt height (1.5H) and then multiplied by the total length of the shelterbelt. The cropped area was determined by subtracting the shelterbelt area and the transition area from the total farm area (Fig. 5.1).



Shelterbelt zone = crown width x length of shelterbelt
Transition zone = (1.5 x tree height – crown width) x length of shelterbelt
Unaffected cropped field = 688 ha – shelterbelt zone – transition zone

Fig. 5.1 Schematic showing the zones (i.e. shelterbelt, transition and unaffected cropped zones) within the model farm

5.4.3 Holos model

Holos is a whole-farm model developed by Agriculture and Agri-Food Canada (Little et al., 2010; http://www.agr.gc.ca/eng/science-and-innovation/science-publications-and-resources/holos/?id=1349181297838), for simulating how various management scenarios reduce GHGs emissions. Holos is a farm-scale empirical model, with a yearly time-step, based primarily on IPCC (2006) methodology, modified for Canadian conditions. The Holos model considers all significant emissions and removals on the farm, as well as emissions from the manufacture of inputs (fertilizer, herbicides) and emissions of N_2O derived from N applied on the farm. The model estimates a whole-farm GHG emission, calculating C storage in linear tree plantings, emissions for soil-derived N_2O, manure-derived CH_4 and N_2O, CO_2 from on-farm energy use and the manufacturing of fertilizer and herbicide, and CO_2 emission/removal from management induced changes in soil carbon stocks. This systems approach allows net whole-farm emissions to be calculated from management changes on any part of the farm (Beauchemin et al., 2010).

5.4.4 Geographical location and climatic conditions of the farm

The Holos model uses emission factors adjusted for variations in climatic and soil conditions across Canada, which are drawn from a database of ecodistricts, with soil information obtained from the Canadian Soil Information System National Ecological Framework (Marshall et al., 1999). The model farm was located in Ecodistrict 772 (i.e. within the Semiarid Prairies ecozone) and the soil type was a Dark Brown Chernozem, of 'medium' soil texture, managed using reduced/minimum tillage practices. Average growing season (May–October) precipitation for the ecodistrict was 259 mm and potential evapotranspiration was 659 mm.

5.4.5 Carbon storage in tree biomass

Holos calculates C storage in aboveground tree biomass based on the methodology described by Little et al. (2008). Annual C accumulation per tree was estimated as a function of tree age and coefficients of annual C accumulation, as shown in the following equation:

$$C_{tree} = [a * (age - 2)]^b \qquad (1)$$

where C_{tree} represents annual C accumulation per tree (kg C year^{-1}), a and b are coefficients of annual C accumulation which vary by tree species and soil type; *age* is the age of the shelterbelt (years). The model assumes that C accumulation in trees starts at least 2 years after planting. Values for coefficient *a* were 0.3232, 0.1345 and 0.4511 and values for coefficient *b* were 0.9651, 0.8970 and 0.6446 for hybrid poplar, white spruce and caragana, respectively (Little et al., 2010).

To estimate the annual C accumulation of the lineal shelterbelt planting on the farm, the following equation was used (Kort and Turnock, 1999):

$$C_{planting} = C_{tree} * \frac{length}{planting\_space} * \#rows \qquad (2)$$

where $C_{planting}$ represents annual C accumulation per linear planting (Mg C year^{-1}), *length* is the total length of shelterbelt in each scenario (km), *planting space* is the spacing of individual trees (m) and *#rows* is the number of tree rows. A planting space of 2-m was used to estimate $C_{planting}$ for hybrid poplar and white spruce. Planting space for caragana shelterbelts in the field ranged between 0.5 to 0.7-m; however $C_{planting}$ for caragana was calculated using a spacing of 10-m harvested sections within the shelterbelt (Kort and Turnock, 1999). Carbon accumulation in belowground biomass for hybrid poplar, white spruce and caragana was estimated as 40%, 30% and 50% of the aboveground C content as suggested by Kort and Turnock (1999). Furthermore, equations (1) and (2) were derived from leafless trees and do not account for C storage in tree leaves, root turnover and exudates.

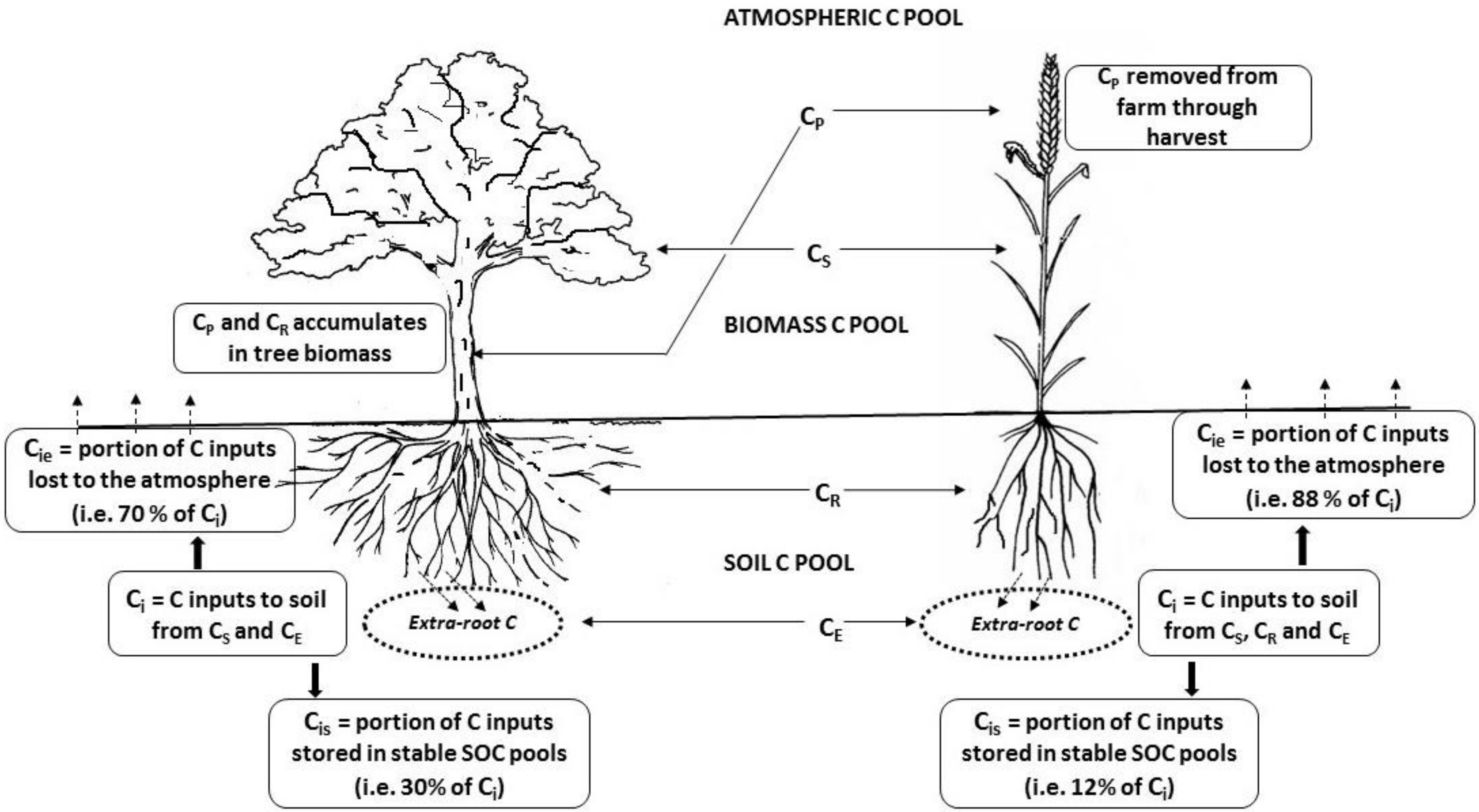


Fig. 5.2 Carbon inputs to soil within the shelterbelt and the cropped area comprising C allocation in plant products (C_P), plant residue such as straw and litter fall (C_S), coarse and fine roots (C_R) and root exudates and root turnover (C_E), based on the concept of Bolinder et al. (2007)

5.4.6 Soil carbon sequestration

Soil C sequestration in the shelterbelt area and cropped area were estimated using the NPP approach described by Bolinder et al. (2007). The NPP approach quantifies annual C storage in above- and belowground biomass by allocating C within different crop plant parts; and estimates annual plant residue input to soil from litter, root turnover and exudates. Soil C sequestration was defined as the fraction of plant residue incorporated into the soil and then integrated into stable SOC pools (Fig. 6.2). The NPP represents C increase in a whole plant and is made up of C associated with different plant compartments as expressed in the following relationship:

$$\mathrm{NPP} = \mathrm{C_P} + \mathrm{C_R} + \mathrm{C_S} + \mathrm{C_E} \tag{3}$$

where C_P is the C stored in harvestable plant products, i.e. grain or tree bole; C_R is the C in plant roots; C_S is the C in the aboveground residues (i.e. crop residues, straw or litterfall); and C_E represents the C derived in root products including root turnover and exudates (Bolinder et al., 2007). Values that were applied to tree species and crops in this study are provided in Table 5.2. Carbon allocation to different plant compartments was estimated as follows:

$$\mathrm{C_P} = \text{Yield} * \text{C content} \tag{4}$$

$$\mathrm{C_R} = \text{Yield} / (\text{shoot: root} * \text{harvest index}) * \text{C content} \tag{5}$$

$$\mathrm{C_S} = \text{Yield} * (1 - \text{harvest index}) / \text{harvest index} * \text{C content} \tag{6}$$

$$C_E = C_R * Y_E \tag{7}$$

where *yield* is the dry matter (DM) yield of aboveground products (kg ha^{-1} yr^{-1}); *harvest index* = DM yield of grain/total aboveground DM yield; and Y_E is the extra-root C from root turnover and exudates relative to recoverable roots. Total annual C input to the soil from various plant components was estimated as follows:

$$\mathrm{C_i} = [\mathrm{C_P} * \mathrm{S_P}] + [\mathrm{C_R} * \mathrm{S_R}] + [\mathrm{C_S} * \mathrm{S_S}] + [\mathrm{C_E} * \mathrm{S_E}] \tag{8}$$

where C_i is the annual C input to soil from plants and S is the proportion of C in the respective plant component that enters the soil. The value of S ranges from 0 to 1 indicating 0 to 100% of a plant fraction incorporated into the soil annually (Bolinder et al., 2007).

Carbon sequestration into soil stable C pools C_{is} is the proportion of C inputs that is potentially integrated into the stable soil C pool. Since the cropped area (including the transition zone) was tilled annually exposing the soil to rapid SOC oxidation while the shelterbelt zone was relatively undisturbed, it was assumed that 12% of C_i was incorporated into stable C pools within the cropped area of the farm (Winans et al., 2015), while 30% of C_i was sequestered into stable C pools within the shelterbelt area (Thevathasan and Gordon, 2004). Thus C_{is} in the cropped and the shelterbelt areas were expressed as follows:

$$C_{is} \text{ (cropped area)} = 0.12\ C_i \tag{9}$$

$$C_{is} \text{ (shelterbelt area)} = 0.30\ C_i \tag{10}$$

Within the cropped area, the C in grains and other harvestable products would be removed from the field and, therefore, not returned to the soil; thus the total C input to soil was defined as follows:

$$C_i = [C_P * 0] + [C_R * 1] + [C_S * 1] + [C_E * 1] \tag{11}$$

Crop yield and C input within the transition zone is not uniform across the entire zone. Kort (1988) reviewed numerous studies on the effect of shelterbelts on crop yield at various distances away from the shelterbelt. He reported no crop yield at a distance of 0H to 0.5H; a 50% reduction in yield due to severe competition for nutrients and water between crop and tree roots at distances from 0.5H to 1H; and a shelterbelt-induced increase in crop yield at distances from 1H to 15H, with the largest increase occurring between 1.5H to 3H relative to the unaffected cropped field. To accommodate the changes in crop yield and C input within the transition zone, it was assumed that the average C input in the transition zone was the same as in the cropped field.

Within the shelterbelt area, C in tree bole (CP) and roots (CR) had been accounted for as C in leafless tree biomass in the previous section. It was assumed that all tree leaves produced per year were deposited to the soil as leaf litter (CS) and CE represented C in root turnover and exudates. Thus C input in the shelterbelt area was defined as:

$$C_i = [C_P * 0] + [C_R * 0] + [C_S * 1] + [C_E * 1] \tag{12}$$

For hybrid poplar and white spruce, C in leaf biomass was calculated to be 9.8% and 16%, respectively of aboveground biomass based on biomass C content of various tree components reported by Peichl et al. (2006). For caragana, leaf biomass C was calculated as 29% of aboveground biomass (Moukoumi et al., 2012). For all trees the fine root biomass C was assumed to be equal to leaf biomass C (Thevathasan and Gordon, 2004). Thus coarse root biomass C was estimated as the belowground biomass minus the fine root biomass. As such, CE was estimated based on root turnover rates of coarse and fine roots reported by Yuan and Chen (2010). Coarse roots (i.e. > 2 mm dia.) had a turnover rate 0.4 for all three tree species; whereas fine root (i.e. ≤ 2 mm dia.) turnover rates were 1.28, 0.84 and 1.15 for hybrid poplar, white spruce and caragana, respectively (Yuan and Chen, 2010). Carbon content in root exudates of all tree species was assumed to be same as in the crops (Table 5.2).

5.4.7 Carbon loss to the atmosphere

Carbon loss from the soil C_{ie} was estimated as the proportion of C inputs that were not integrated into the stable soil C pool, but were released back to the atmosphere through microbial decomposition processes (Winans et al., 2015). Annual CO_2 emissions from cropped and shelterbelt areas were estimated as 88% and 70% of total C inputs to the soil, and expressed as:

$$C_{ie} \text{ (cropped area)} = 0.88\ C_i \tag{13}$$

$$C_{ie} \text{ (shelterbelt area)} = 0.70\ C_i \tag{14}$$

Table 5.2 Values of crop yield (Mg DM ha^{-1} yr^{-1}) C content (%), harvest index, root: shoot ratio, extra root C (YE) and root turnover used for the calculation C inputs to soil within the model farm

	Spring wheat		Dry Pea		Oats	
	Value	Reference	Value	Reference	Value	Reference
Yield (Mg DM ha^{-1} yr^{-1})†	1333	Little et al. (2008)	943	Little et al. (2008)	1008	Little et al. (2008)
C content (%)	0.45	Bolinder et al. (2007)	0.45	Bolinder et al. (2007)	0.45	Bolinder et al. (2007)
Harvest index	40	Bolinder et al. (2007)	42	Bolinder et al. (2007)	53	Bolinder et al. (2007)
Shoot : root	5.6	Campbell and de Jong (2001)	9.2	House et al. (1984)	2.5	Izaurralde et al. (1993)
YE	1	Bolinder et al. (1997)	1	Bolinder et al. (1997)	1	Bolinder et al. (1997)

	Hybrid poplar		White spruce		Caragana	
	Value	Reference	Value	Reference	Value	Reference
C content (%)	0.48	Freedman and Keith (1995)	0.50	Freedman and Keith (1995)	0.51	Freedman and Keith (1995)
Fine root turnover	1.28	Yuan and Chen (2010)	0.84	Yuan and Chen (2010)	1.15	Yuan and Chen (2010)
Coarse root turnover	0.4	Yuan and Chen (2010)	0.4	Yuan and Chen (2010)	0.4	Yuan and Chen (2010)
YE	1	Bolinder et al. (1997)	1	Bolinder et al. (1997)	1	Bolinder et al. (1997)

† Yields are default values in Holos estimated from (McConkey et al., 2007) for the ecodistrict of the farm location

5.4.8 Soil nitrous oxide emissions

Holos calculates direct N_2O from soils based on N inputs, modified by climate, tillage, soil texture and topography. For the cropped area, total N additions to soil comprised synthetic N fertilizer additions and N derived from above- and belowground crop residue decomposition. Fertilizer N inputs were estimated from total N requirement by crops (McConkey et al., 2007), while N inputs from crop residues was calculated from crop yields, using coefficients derived from Janzen et al. (2003). Thus, during the 60 year-long crop rotation fertilizer N application to the cropped area were default values of 45 kg N ha^{-1} yr^{-1} for spring wheat and oats and 0 kg N ha^{-1} yr^{-1} for dry peas (McConkey et al., 2007). For the shelterbelt area total N additions to soil included N in leaf litter and N in root turnover. The N content in leaf litter was estimated as 2.0 % for hybrid poplar (Thevathasan and Gordon, 1997); 1.17 % for white spruce (Wang and Klinka, 1997) and 3 % for caragana (Moukoumi et al., 2012). Foliar N content of each tree species was assumed to be the same as N content in root turnover.

Holos calculates soil-derived N_2O emission from total N inputs, using Canada-specific algorithms modified from those developed for calculating the national GHG inventory (Rochette et al., 2008). The total N input was multiplied by an emission factor, adjusted for growing season precipitation and the potential evapotranspiration for the ecodistrict, using data from CanSIS averaged from 1971 to 2000 (Marshall et al., 1999). Modifiers for soil type, texture, tillage system, and topography were based on Rochette et al. (2008). The emission factor was calculated as follows:

$$EF_{eco} = 0.022 * P/PE - 0.0048 \tag{15}$$

where EF_{eco} represents ecodistrict emission factor [kg N_2O-N (kg N)$^{-1}$]; P is growing season precipitation by ecodistrict (May – October) (mm); and PE is the growing season evapotranspiration (May-October) (mm). Based on equation (15), an emission factor of 0.0047 kg N_2O-N (kg N)$^{-1}$ was used to estimate N_2O emission in all zones of the model farm. Soil N_2O emissions from the cropped field was defined as:

$$N_2O\text{-}N_{cropinputs} = (N_{fert} + N_{res}) * \text{unaffected cropped field} * EF_{eco} \tag{16}$$

where $N_2O\text{-}N_{cropinputs}$ represents the N emissions from cropland due to crop inputs to soil (kg N_2O-N), N_{fert} is N input from synthetic N fertilizers (kg N); N_{res} is N input from crop residue returned to soil (kg N). Soil N_2O emissions from the shelterbelt area was defined as:

$$N_2O\text{-}N_{treeinputs} = (N_{leaf\_litter} + N_{root\_turnover}) * \text{shelterbelt area} * EF_{eco} \qquad (17)$$

where $N_2O\text{-}N_{treeinputs}$ represents the N emissions from the shelterbelt area due to tree inputs to the soil (kg N_2O-N), N_{leaf_litter} is N input from tree leaf litter (kg N); $N_{root\_turnover}$ is N input from tree root turnover (kg N). Soil N_2O emissions in the transition zone was estimated as one half of N_2O emissions in the cropped field. This was based on more efficient N cycling reported in this zone. Tree roots extend to the transition area and take up excess soil N and moisture, which reduces the processes that result in N_2O emissions (Evers et al., 2010).

5.4.9 Soil methane fluxes

In general, cropped fields are slight sources or sinks of soil CH_4 (Bronson and Mosier, 1993); however, the incorporation of trees into cropped fields could significantly increase soil CH_4 sink size through the removal of excess soil moisture, an increase in SOM and a decrease in soil bulk density (Hutsch et al., 1994). In a study of GHG exchanges in an age-sequence of temperate pine forests, Peichl et al. (2010) reported increasing CH_4 oxidation with increasing root biomass ($r^2 = 0.76$). Similarly, in the study of the GHG emissions along a transect from a hybrid poplar-caragana shelterbelt into an adjacent cropped field (Chapter 4), a strong negative correlation (0.94; $P < 0.05$) between cumulative CH_4 emissions and root biomass was observed, indicating that levels of CH_4 uptake increased with increasing root biomass. The increasing soil CH_4 oxidation with increasing root biomass reflects greater soil water uptake, improved soil aggregates and SOM additions in deeper soil layers; which in turn increases soil aeration and gas diffusivity, resulting in reduced CH_4 production in the soil. For this study, therefore, soil CH_4 flux CH_{4soil} in the shelterbelt and cropped field was estimated using a regression equation representing the relationship between root biomass and CH_4 emission (Chapter 4):

$$CH_{4soil} = -197.83(root\ biomass) + 18.108 \qquad (18)$$

Soil CH_4 flux in the transition zone was estimated as one half of the CH_4 flux in the shelterbelt area. This assumption is based on reported reduction of root biomass in this zone relative to the shelterbelt zone and root competition for resources with crops (Kort, 1988; Chapter 4).

5.4.10 Carbon dioxide emissions from farm energy use

Holos estimates CO_2 emissions from the use of fossil fuel on the farm and categorizes them as primary and secondary emission sources (Gifford, 1984). Primary sources include use of fossil fuel for cropping, i.e. tillage, seeding/fertilizer application and harvesting. Secondary sources of CO_2 emissions from fossil fuels include emissions related to the manufacture of fertilizers and herbicides. Carbon dioxide emissions associated with the transportation of goods to the farm or manufacture of farm machines were not considered.

Carbon dioxide emissions associated with cropping was estimated at 133 kg CO_2 ha^{-1} (Little et al., 2008). Emissions related to the manufacture of N and P fertilizers was estimated at 3.59 kg CO_2 (kg N)$^{-1}$ and 0.5699 kg CO_2 (kg P_2O_5)$^{-1}$, respectively while emissions associated with manufacturing N fertilizers was not applicable to peas as it received no N fertilization. Energy emissions related to the manufacture of herbicide production was 1.334 kg CO_2 ha^{-1}. Based on the above values, annual farm CO_2 emissions from farm energy ($CO_{2energy}$) in the cropped area was 0.30 Mg CO_2e ha^{-1} yr^{-1} for spring wheat and oats and 0.14 Mg CO_2e ha^{-1} yr^{-1} for dry peas. The shelterbelt zone was excluded from CO_2 emissions from fossil fuel use; however, it was assumed that the emissions associated with planting trees was equivalent to emissions associated with growing spring wheat for one year (i.e. 0.30 Mg CO_2e ha^{-1}).

5.4.11 Whole-farm greenhouse gas emissions

Whole-farm GHG emissions GHG_{whole_farm} was defined as the sum of all sources and sinks of GHG emissions across the entire farm (i.e. the shelterbelt, transition zone and cropped field) and was expressed in Mg CO_2e to account for the global warming potential of the respective gases. Whole-farm GHG emissions per year was expressed as:

$$GHG_{whole\_farm} = [C_{planting} * (44/12) * (-1)] + [C_{is} * (44/12) * (-1)] + [C_{ie} * (44/12) * (+1)] + [N_2O\text{-}N * 296 * (+1)] + [CH_{4soil} * 23 * (+1)] + [CO_{2energy} * (+1)] \quad (19)$$

where GHG_{whole_farm} represents whole farm GHG emissions (Mg CO_2e yr^{-1}), (44/12) is the conversion factor from C to CO_2e, *296* is conversion factor from N_2O to CO_2e, and *23* is the conversion factor from CH_4 to CO_2e.

5.5 Results

5.5.1 Carbon storage in tree biomass

Carbon fixation in above- and belowground biomass was 4.22, 2.70 and 0.83 Mg C ha^{-1} yr^{-1} for hybrid poplar, white spruce and caragana shelterbelts, respectively (Table 5.3). For all three tree species, simulated C storage in tree biomass to age 60 increased with increasing farm area planted to shelterbelts; however, C storage in tree biomass varied between the three tree species (Fig. 5.3). At the end of 60 years of growth, the maximum predicted C accumulation was 8,712, 5,581 and 1,705 Mg C for the 5.0% scenario for hybrid poplar, white spruce and caragana, respectively (Fig. 5.3; Table 5.4).

5.5.2 Soil C inputs, sequestration and loss

Within the cropped zone, average C inputs into the soil from crop residues (i.e. straw, roots, and root exudates from the wheat-peas-oats rotation) was 1.11 Mg C ha^{-1} yr^{-1}; resulting in C sequestration of 0.13 Mg C ha^{-1} yr^{-1} into soil stable C pools and C loss of 0.98 Mg C ha^{-1} yr^{-1} into the atmosphere through microbial decomposition processes (Table 5.3). However, within the shelterbelt zone, C inputs to soil (i.e. leaf litter, root turnover and exudates) were 2.26, 1.35 and 0.66 Mg C ha^{-1} yr^{-1} for hybrid poplar, white spruce and caragana, respectively. As a result, the C sequestered into stable soil C pools was 0.68, 0.41 and 0.23 Mg C ha^{-1} yr^{-1}; while C loss to the atmosphere was 1.58, 0.94 and 0.43 Mg C ha^{-1} yr^{-1} for the hybrid poplar, white spruce and caragana shelterbelts, respectively (Table 5.3).

Whole-farm C sequestration into stable soil C pools increased with increasing shelterbelt area; but the increase in soil C sequestration varied with tree species (Fig. 5.4A – 5.4C). For example, at the baseline scenario (i.e. scenario 0%), total soil C within the farm after 60 years was 5,495 Mg C. Incorporating shelterbelts into the farm increased the amount of C storage in the soil relative to baseline levels, reaching a maximum soil C storage of 6,617, 6,058 and 5,701 Mg C for hybrid poplar, white spruce and caragana, respectively (Table 5.4).

Total C loss to the atmosphere through microbial decomposition processes over 60 years of farming was 40,297 Mg C at the baseline scenario (i.e., no shelterbelts). However, with increasing shelterbelt area, C loss from the soil increased with hybrid poplar and decreased with caragana shelterbelts, but appeared to be comparatively constant with white spruce species reflecting differences in amounts of tree litter inputs and root respiration among these tree species (Table 5.4).

Table 5.3 Estimated C inputs into the soil (C_i) derived from plant product biomass (C_P), plant residues (C_S), root biomass (C_R), root turnover and root exudates (C_E), including C storage in a stable soil C pool (C_{is}), C loss to the atmosphere (C_{ie}), emissions from farm energy (CO_2energy), methane (CH_{4soil}), nitrous oxide (N_2O_{soil}) and total GHG emissions (GHG_{total}) per hectare of shelterbelt area, transition zone and unaffected cropped field within a 688 ha model farm

	C_P	C_R	C_S	C_E	C_i	C_{is}	C_{ie}	$CO_{2energy}$	CH_{4soil}	N_2O_{soil}	GHG_{total}
	---------------------- (Mg C ha^{-1} yr^{-1}) ----------------------							(Mg CO_2 ha^{-1} yr^{-1})	(kg CH_4-C ha^{-1} yr^{-1})	(kg N_2O-N ha^{-1} yr^{-1})	(Mg CO_2e ha^{-1} yr^{-1})
Unsheltered cropped field											
Spring wheat	0.6	0.27	0.9	0.27	1.44	0.17	1.27	0.30	-0.030	0.64	4.53
Peas	0.42	0.11	0.59	0.11	0.81	0.1	0.71	0.14	-0.002	0.29	2.46
Oats	0.45	0.34	0.4	0.34	1.09	0.13	0.96	0.30	-0.043	0.61	3.53
Transition zone											
Spring wheat	0.6	0.27	0.9	0.27	1.44	0.17	1.27	0.30	-0.100	0.32	4.43
Peas	0.42	0.11	0.59	0.11	0.81	0.1	0.71	0.14	-0.047	0.145	2.42
Oats	0.45	0.34	0.4	0.34	1.09	0.13	0.96	0.30	-0.016	0.305	3.44
Shelterbelt zone											
Hybrid poplar	3.01	1.21	0.30	1.95	2.26	0.68	1.58	0.005	-0.199	0.18	-12.13
White spruce	2.08	0.62	0.33	1.02	1.35	0.41	0.94	0.005	-0.094	0.07	-7.94
Caragana	0.55	0.28	0.16	0.51	0.66	0.23	0.43	0.005	-0.031	0.10	-2.28

Table 5.4 Estimates of C in tree biomass, soil C sequestration, Soil CO_2 emissions, CH_4 fluxes, N_2O emissions, farm energy emissions and whole-farm GHG emissions across the five scenarios of shelterbelt establishment after 60 years of crop production in a 688 ha model farm

Parameter/ Shelterbelt species considered	Scenario (Proportion of farm planted to shelterbelt)				
	0%	0.6%	1.2%	3%	5%
C in tree biomass (Mg C*)*					
Hybrid poplar	0	1045	2091	5227	8712
White spruce	0	669	1338	3348	5581
Caragana	0	205	409	1023	1705
Soil C sequestration (Mg C*)*					
Hybrid poplar	5495	5630	5764	6168	6617
White spruce	5495	5563	5630	5833	6058
Caragana	5495	5520	5544	5618	5701
Soil CO_2 *emissions (Mg* C*)*					
Hybrid poplar	40297	40446	40596	41043	41541
White spruce	40297	40290	40283	40261	40237
Caragana	40297	40163	40028	39624	39174
Soil CH_4 *exchange (Mg* CH_4-C*)*					
Hybrid poplar	-1.0	-1.2	-1.5	-2.1	-2.8
White spruce	-1.0	-1.2	-1.3	-1.7	-2.1
Caragana	-1.0	-1.0	-1.0	-1.0	-1.0
Soil N_2O *emissions (Mg* N_2O-N*)*					
Hybrid poplar	21.2	20.7	20.3	18.8	17.3
White spruce	21.2	20.4	19.7	17.3	14.8
Caragana	21.2	21.1	21.0	20.7	20.3
Farm energy CO_2 *emissions (Mg* C*)*					
Hybrid poplar	2788	2772	2755	2706	2652
White spruce	2788	2772	2755	2706	2652
Caragana	2788	2772	2755	2706	2652
Whole-farm GHG emissions (Mg CO_2e*)*					
Hybrid poplar	144205	140218	136231	124269	110979
White spruce	144205	141185	138165	129089	119011
Caragana	144205	142778	141350	137067	132309

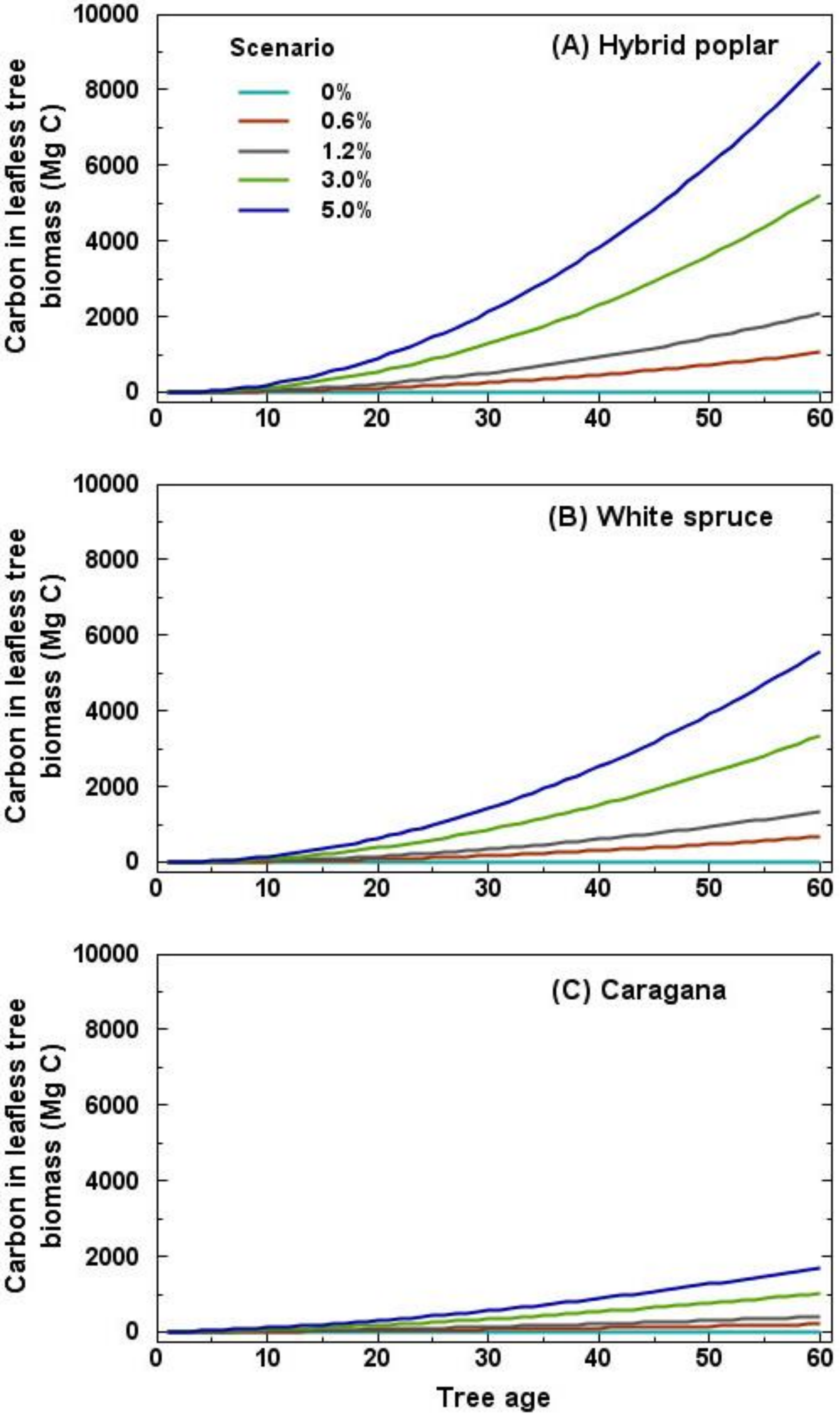


Fig. 5.3 Carbon storage in tree biomass for hybrid poplar, white spruce and caragana across the five scenarios of shelterbelts establishment in a 688 ha model farm for a period of 60 years

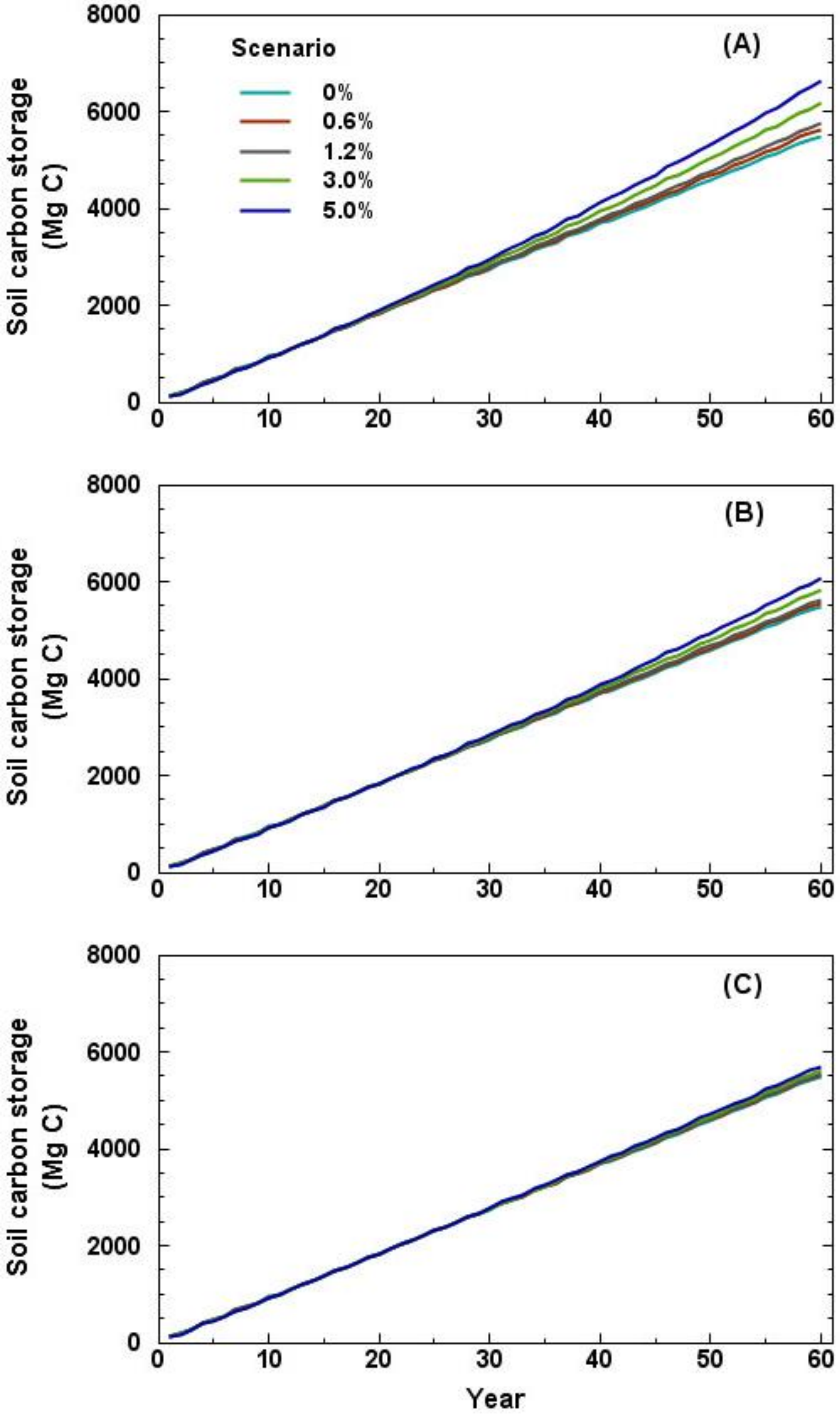


Fig. 5.4 Carbon storage in the soil across five scenarios of shelterbelt establishment in a 688 ha model farm considering (A) hybrid poplar, (B) white spruce and (C) caragana tree species for a period of 60 years

5.5.3 Soil methane and nitrous oxide exchange

Within the cropped field zone, average soil CH_4 exchange (i.e. after 60 years of wheat-peas-oats rotation) was estimated at -0.025 kg CH_4-C ha^{-1} yr^{-1}(a net sink); while average soil N_2O emissions was 0.51 kg N_2O-N ha^{-1} yr^{-1} (Table 5.3). In the shelterbelt zone, the predicted soil CH_4 consumption rate was greatest under hybrid poplar (-0.20 kg CH_4-C ha^{-1} yr^{-1}), followed by white spruce (-0.09 kg CH_4-C ha^{-1} yr^{-1}) and caragana (-0.03 kg CH_4-C ha^{-1} yr^{-1}). In contrast, the lowest rate of soil N_2O emission was predicted for white spruce (0.07 kg N_2O-N ha^{-1} yr^{-1}), followed by caragana (0.10 kg N_2O-N ha^{-1} yr^{-1}) and hybrid poplar (0.18 kg N_2O-N ha^{-1} yr^{-1}).

For the baseline scenario (i.e. scenario 0), total soil CH_4 oxidation and N_2O emissions for the farm after 60 years was -1.0 Mg CH_4-C and 21.2 Mg N_2O-N, respectively. The incorporation of various amounts of shelterbelts into the cropped field resulted in increased soil CH_4 uptake and reduced N_2O emissions, although the changes in both gases varied with tree species (Fig. 5.5A – 5.5C; 5.6A – 5.6C). Maximum whole farm CH_4 uptake (-2.8 Mg CH_4-C, at scenario 5) was achieved when the shelterbelt species in the farm was hybrid poplar, followed by white spruce (-2.1 Mg CH_4-C) and caragana (-1.1 Mg CH_4-C). However, lowest farm soil N_2O emissions (14.8 Mg N_2O-N) was reached with white spruce, followed by hybrid poplar (17.3 Mg N_2O-N) and caragana (20.3 Mg N_2O-N) (Table 5.4).

5.5.4 Farm energy carbon dioxide emissions

The emissions of CO_2 due to fuel use (i.e. from running farm machines and the manufacture of fertilizers and herbicides) averaged 0.25 Mg CO_2 ha^{-1} yr^{-1} in the cropped zone and 0.005 Mg CO_2 ha^{-1} yr^{-1} in the shelterbelt zone (Table 5.3). Total farm energy use after 60 years was 2,788 Mg C without planted shelterbelts; however, total farm energy was decreased by as much as 136 Mg C for the largest ratio of shelterbelt area (scenario 5) (Table 5.4).

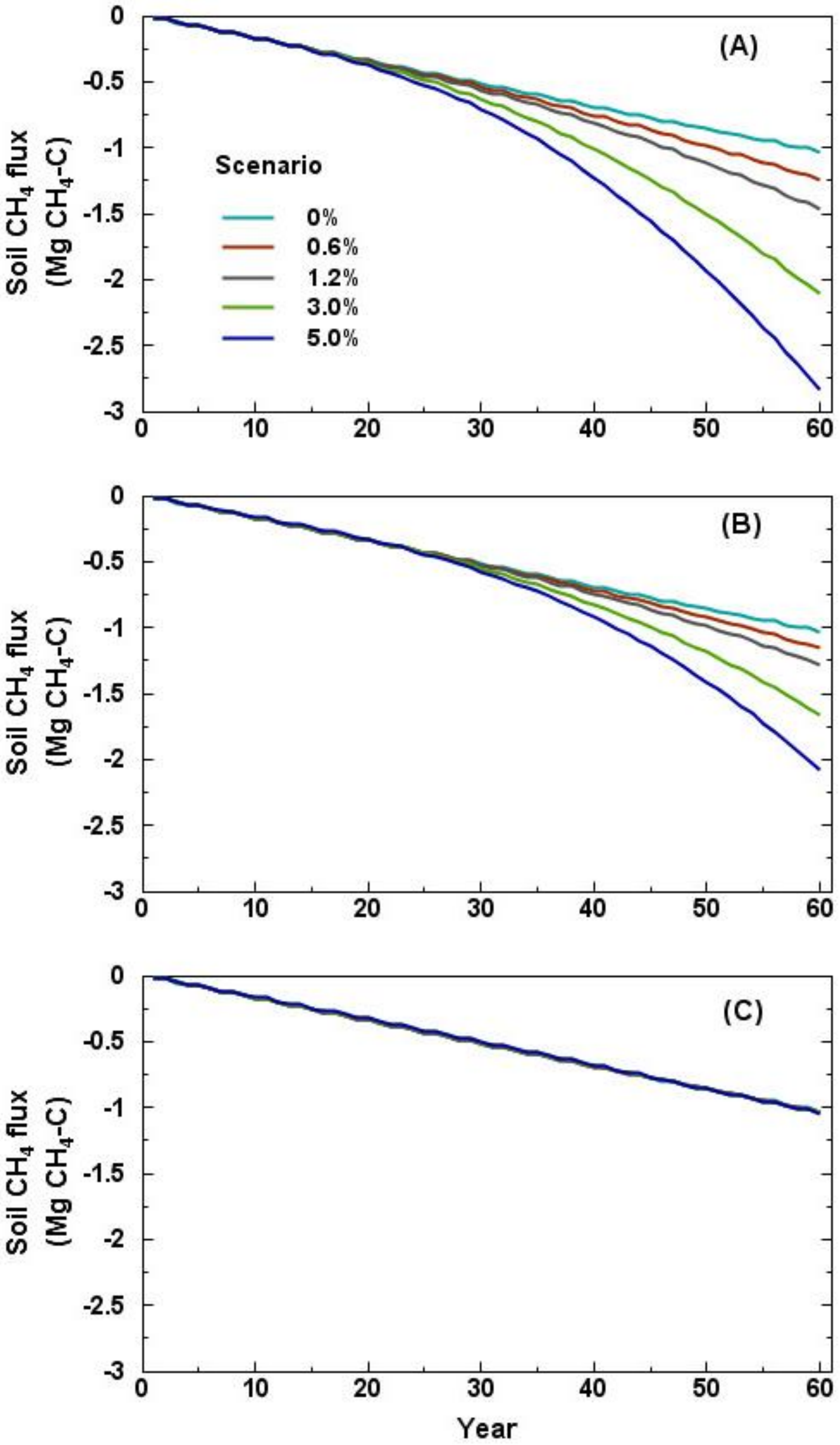


Fig. 5.5 Cumulative CH_4 oxidation across five scenarios of shelterbelt establishment in a 688 ha model farm considering (A) hybrid poplar, (B) white spruce and (C) caragana tree species for a period of 60 years

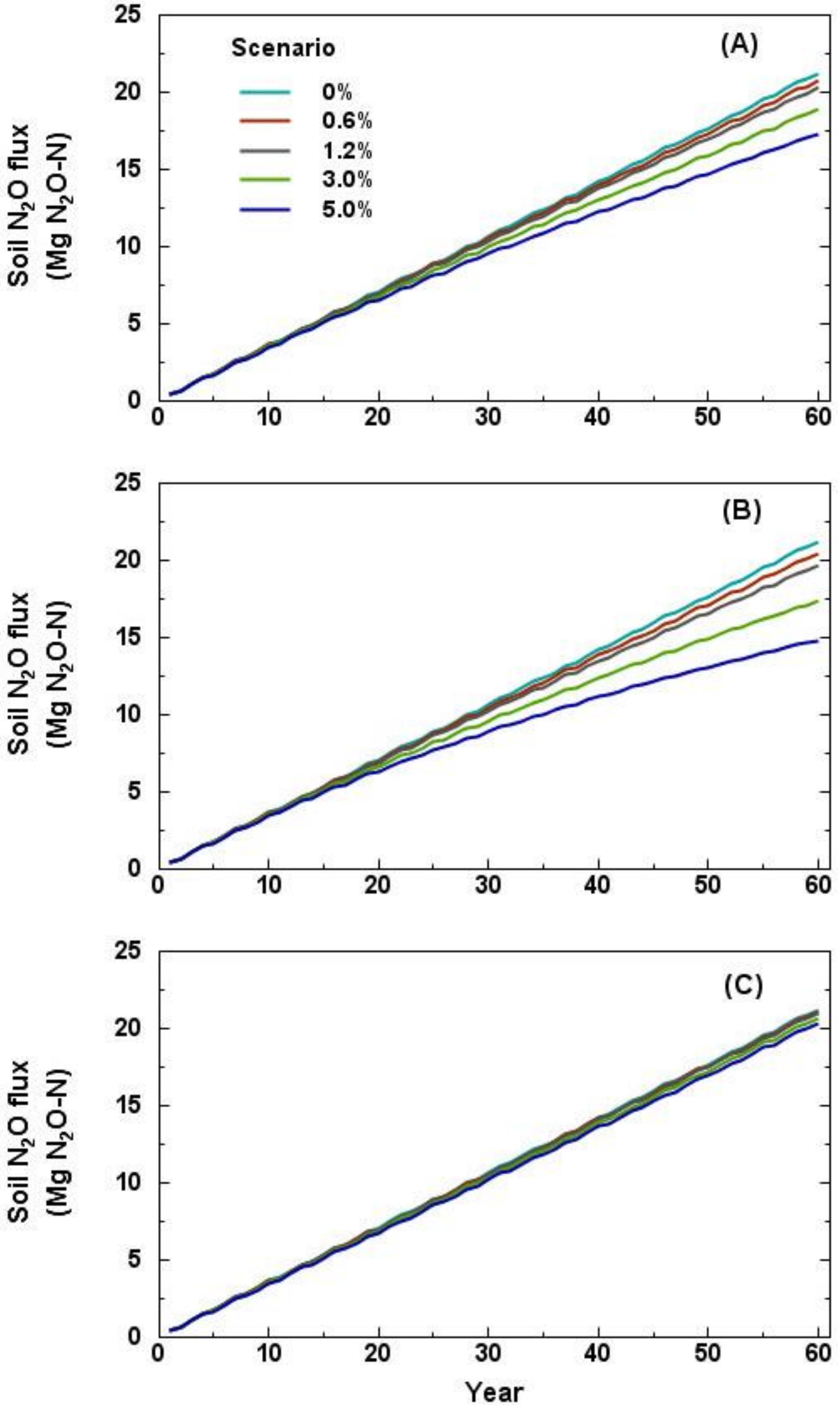


Fig. 5.6 Cumulative N_2O emissions across five scenarios of shelterbelt establishment in a 688 ha model farm considering (A) hybrid poplar, (B) white spruce and (C) caragana tree species for a period of 60 years

5.5.5 Whole-farm greenhouse gas emissions

Crop production (i.e. wheat-peas-oats rotation) in the cropped field zone resulted in a net annual GHG emission of 3.51 Mg CO_2e ha^{-1} yr^{-1}; however, the shelterbelt area was an annual net sink of atmospheric GHGs over the 60-year period irrespective of the shelterbelt species (Table 5.3). The largest sink (-12.1 Mg CO_2e ha^{-1} yr^{-1}) was achieved with hybrid poplar, followed by white spruce (-7.9 Mg CO_2e ha^{-1} yr^{-1}) and caragana (-2.3 Mg CO_2e ha^{-1} yr^{-1}). Total farm GHG emissions over 60 years was 144,205 Mg CO_2e for the baseline scenario and decreased with increasing shelterbelt area planted (Fig. 6.7A – 6.7C). The greatest reduction in total farm GHG emissions (110,979 Mg CO_2e, at scenario 5) was simulated for hybrid poplar shelterbelts representing a 23.0% decrease in cumulative farm emissions. Planting white spruce shelterbelts decreased overall farm emissions by 17.5% (119,011 Mg CO_2e), while caragana shelterbelts reduced farm emissions by 8.2% (132,309 Mg CO_2e) at the largest planting (Table 5.4).

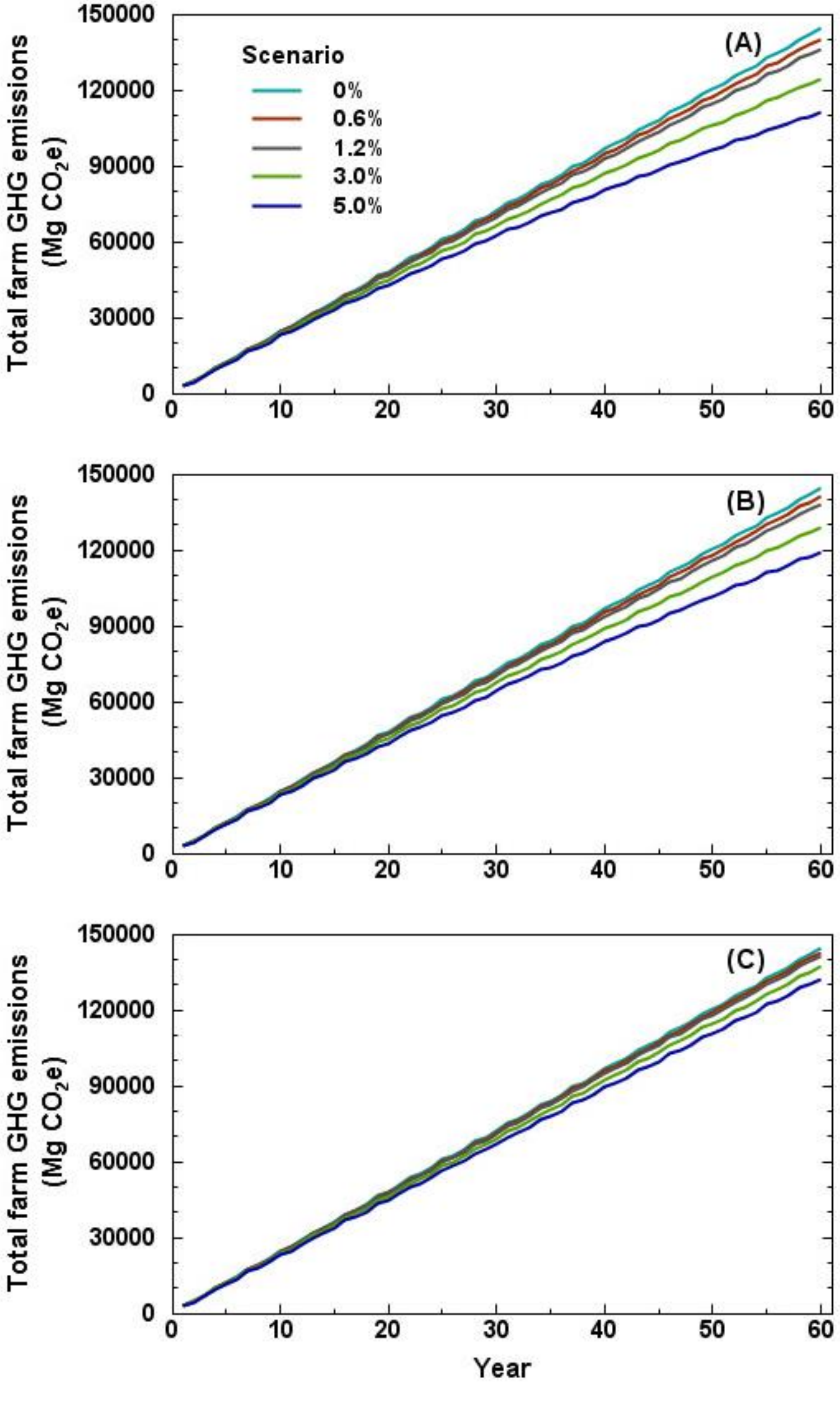


Fig. 5.7 Total farm GHG emissions across five scenarios of shelterbelt establishment in a 688 ha model farm considering (A) hybrid poplar, (B) white spruce and (C) caragana tree species for a period of 60 years

5.6 Discussion

5.6.1 Carbon sequestration in tree biomass and stable SOC pools

The Holos model simulations showed that tree species selection is important for maximizing C sequestration; e.g., greater C accumulation was predicted for hybrid poplar, followed by white spruce and caragana. The estimated rate of C sequestration in above- and belowground biomass of hybrid poplar in the present study (i.e. 4.2 Mg C ha^{-1} yr^{-1}) was comparable to the C sequestration values of 3.29 to 5.18 Mg C ha^{-1} yr^{-1} obtained using the 3PG model (Amichev et al., 2015), but was much lower than the value of 8.57 Mg C ha^{-1} yr^{-1} measured by Kort and Turnock (1999) using destructive sampling. Estimated C storage in white spruce in this study was slightly lower than those predicted for white spruce in the 3PG model, but was more than two times greater than the measured value in Kort and Turnock (1999). Predicted C sequestration in caragana trees using the Holos model was slightly lower than that predicted in the 3PG model and those measured by destructive sampling (Table 5.5).

The differences in C sequestration rates among these studies can be attributed to site-specific factors such as tree spacing, crown widths, climate, soil zone and overall productivity of the site. The growth equation within the Holos model appeared to predict C sequestration satisfactorily for hybrid poplar, white spruce and caragana tree species when compared to estimates obtained using the 3PG model. However, further work is needed on partitioning the above- and belowground C equations for various tree species in the Holos model.

The estimated annual gain in SOC in the cropped zone (i.e. the wheat-peas-oats rotation) of the present study (0.10 – 0.17 Mg C ha^{-1} yr^{-1}) was comparable to the C sequestration value of 0.16 Mg C ha^{-1} yr^{-1} estimated for a continuous wheat rotation in Southern Saskatchewan using the CENTURY model (Campbell et al., 2005), and was within the range obtained through field measurement (i.e., 0.09 – 0.29 Mg C ha^{-1} yr^{-1}) (Campbell et al., 2001). Likewise, annual SOC sequestration within the shelterbelt (0.23 – 0.68 Mg C ha^{-1} yr^{-1}) in this study, was within the range of SOC sequestration values reported for agroforestry practices in Canada (0.2 – 1.0 Mg C ha^{-1} yr^{-1}) (Janzen et al., 2001) and in the United States (0.23 – 1.15 Mg C ha^{-1} yr^{-1}) (Eagle et al., 2011). The greater annual SOC sequestration in the shelterbelt zone relative to the cropped zone was attributed to the role of trees in enhancing the quantity and quality of shoot and root litter C inputs and in modifying microclimatic conditions such as soil moisture and temperature regimes

(Laganière et al. 2010). Correspondingly, the greater SOC sequestration in hybrid poplar shelterbelts was attributed to greater biomass production and consequently, more rapid C input to soil through litter fall and root turnover compared to the white spruce and caragana shelterbelts.

The estimation of annual C sequestration in the soil depends on several crop- and tree- specific values, as well as site specific factors such as soil zone, tillage practices, fertilizer application rates and weather regime. While the Holos model can factor indirect emissions related to agronomic practices and site-specific conditions, the NPP method used in estimating SOC sequestration in this study did not consider the finite capacity of soils to store C. Soil organic C levels are assumed to stabilize at a new steady state after 20 years of management (IPCC 2006); however, a longer period of 45 years has been reported for agroforestry systems (Hernandez-Ramirez et al. 2011).

Table 5.5. Annual carbon sequestration in the above- and belowground biomass of commonly used shelterbelt tree species in Saskatchewan, Canada

Source/ method	Location	Age (yr)	Tree species	Planting space (m)	C in tree biomass (Mg C ha^{-1} yr^{-1})
This study HOLOS model	Saskatchewan, Canada	60	Hybrid poplar	2	4.2
		60	white spruce	2	2.7
		60	caragana	2	0.83
Kort and Turnock (1999) † Destructive measurement	Saskatchewan, Canada	33	Hybrid poplar	2.5	8.57
		54	white spruce	3.5	1.24
		49	caragana	0.3	1.56
		53	green ash	2.5	2.23
Amichev et al (2015) 3PG model	Saskatchewan, Canada	60	Hybrid poplar	2	3.29 - 5.18
		60	white spruce	2	2.24 - 4.13
		60	caragana	2	1.31 - 2.67
		60	green ash	2	2.02 - 3.92
		60	Manitoba maple	2	2.8 - 5.26
		60	Scots pine	2	1.44 - 3.26

† Belowground carbon for deciduous, coniferous and shrub shelterbelts was calculated to be 40%, 30% and 50% of the aboveground carbon content, respectively (Freedman and Keith, 1995).

5.6.2 Methane and Nitrous oxide fluxes

The greater annual CH_4 oxidation observed in the shelterbelt zone compared to the cropped zone reflects the greater root biomass of planted trees within the shelterbelt zone. Among the tree species compared, the greatest CH_4 oxidation was predicted for hybrid poplar shelterbelts and this was related to the greater root biomass in hybrid poplar compared to white spruce and caragana shelterbelts (Table 5.3). Estimated annual CH_4 oxidation within the shelterbelt zone in the present study (-0.031 to -0.199 kg CH_4-C ha^{-1} yr^{-1}) was at the lower range of CH_4 oxidation (-0.14 to -0.99 kg CH_4-C ha^{-1} yr^{-1}) measured in shelterbelts within the Dark-brown soil zone in Saskatchewan, Canada (Chapter 3), and those (-0.43 to -3.0 kg CH_4-C ha^{-1} yr^{-1}) measured in a 67-year old pine forest in Eastern Canada (Peichl et al., 2010). Annual CH_4 oxidation within the cropped field in the present study (-0.002 to -0.043 kg CH_4-C ha^{-1} yr^{-1}) was comparable to a slight CH_4 sink (-0.019 kg CH_4-C ha^{-1} yr^{-1}) reported in Chapter 3. In a three year study of GHG intensity in irrigated cropping systems in Northeastern Colorado, Mosier et al. (2006) reported a much wider range of CH_4 fluxes (0.392 to -0.151 kg CH_4-C ha^{-1} yr^{-1}) across various tillage, N fertilization and crop rotation regimes. This study suggests that a management practice or crop rotation that enhances root proliferation can increase CH_4 oxidation in arable soils.

The greater annual N_2O emissions estimated in the cropped zone relative to the shelterbelt zone (Table 5.3) was reflective of the greater N inputs in the cropped field (i.e. 45 kg N ha^{-1} yr^{-1} plus N in the crop residue) relative to the shelterbelt zone where N input was mainly a function of N concentration in leaf litter and root turnover. This result is in agreement with the data in Chapter 3, where significantly greater N_2O emissions from cropped fields were observed compared to shelterbelts within the Boreal and Prairie Ecozones of Saskatchewan. The greatest reduction in N_2O emissions was estimated for white spruce which is attributed to lower N concentrations in the needles (1.17%) compared to hybrid poplar (2%) and caragana leaves (3%). However, the greater annual N_2O predicted for caragana compared to white spruce was not unexpected as caragana trees are N-fixing—acquiring more than 80% of their N requirement through N-fixation (Moukoumi et al., 2012). This result is consistent with data in Appendix A., where significantly greater N_2O emissions in caragana shelterbelts compared to Scots pine shelterbelts were observed; suggesting that trees with relatively low foliar N concentrations (such as conifers) may be more efficient in reducing soil N_2O emissions compared to tree species with comparatively greater foliar N concentrations.

Planting shelterbelts composed of pure stands of N-fixing trees (e.g. caragana) may be beneficial in terms of C sequestration; however, they may be significant sources of atmospheric N_2O emissions, which may constitute an even greater environmental hazard (Albrecht and Kandji, 2003). During shelterbelt establishment, it may be more effective to inter-plant N-fixing trees with non-N-fixing trees, as this would not only improve the N nutrition of the non-N-fixing trees, but also decrease N_2O loses by reducing the amount of fixed N in the soil. Mixing tree plantings with N-fixing trees has been reported to increase biomass production, thus C sequestration, and result in greater retention of relatively stable SOC (Resh et al., 2002; Nair et al., 2009). However, more research is needed to elucidate the role of N-fixing tree species on GHG dynamics in tree-based systems. Clearly, the success of agroforestry systems in tackling issues of climate change will depend on adequate understanding of trade-offs between C sequestration and the emission of GHGs such as CH_4 and N_2O.

5.6.3 Overall farm emissions

The Holos model was useful in predicting the impact of three shelterbelt species under five planting scenarios on GHG mitigation for a model farm for a 60 year period. My data indicate that despite the relatively small proportion of the farm occupied by shelterbelts, the mitigation potential of the shelterbelts (over a 60 yr timeframe) ranges from 11,896 to 33,226 Mg CO_2e depending on the species and planting density of the shelterbelts. The model simulations from Holos demonstrate the importance of tree species selection in maximizing the C sequestration and GHG mitigation potential from shelterbelt systems. Previous studies have attributed the mitigation of atmospheric GHG in agroforestry systems to the fixation of C in above- and belowground biomass, increased C sequestration in the soil, enhanced CH_4 oxidation and reduced N_2O and energy emissions due to the exclusion of N fertilization on areas occupied by trees (Evers et al., 2010; Schoeneberger et al., 2012). However, these studies did not report the relative contributions of these components to the overall GHG mitigation in agroforestry systems. Modelling simulations from this study indicate that 90 – 95% of GHG mitigation by shelterbelts was through C sequestration in tree biomass and in stable SOC pools, while the reduction in N_2O emissions contributed 5.1 – 9.6% of the total GHG mitigation by shelterbelts. Increased methane oxidation contributed only 0.002 –

0.12%, while a reduction in CO_2 emissions associated with reduced farm energy consumption contributed 1.5 – 4.2% of the total GHG mitigation by shelterbelts.

The major appeal of shelterbelt systems as a GHG mitigation strategy is based on its ability to sequester large amounts of C on a relatively small land unit (i.e. $\leq$ 5%), while leaving the bulk of the land for agricultural production (Ruark et al., 2003). Based on my modelling data, a viable strategy for reducing GHG emissions in the agricultural landscape is the incorporation of trees on the farm—be it in the form of shelterbelts, riparian buffers or other agroforestry systems. In addition, marginal agricultural lands or parcels that are not farmed due to land degradation could be targeted for tree plantings without jeopardizing food production. Older shelterbelts in the province should also be rehabilitated in order to maintain or enhance the mitigating potential of shelterbelts on agricultural landscapes.

5.7 Conclusion

The Holos model predictions suggest that, despite the relatively small area they occupy on the agricultural landscape, shelterbelt systems can capture substantial amounts of atmospheric C and store them in tree biomass and soil, reduce N_2O emissions and improve soil CH_4 oxidation. Of the three species tested in the model, hybrid poplar was the most effective species for maximizing C sequestration and mitigating GHGs, followed by white spruce and caragana. Caragana shelterbelts that incorporated conifers would appear to be more effective at reducing N_2O emissions. Future research is needed to determine which tree species would be the most effective at mitigating GHGs with future changing climates. The potential role that shelterbelts can play in mitigate GHGs suggests that future policy should ensure that trees are planted in agricultural landscapes and that current shelterbelts should be maintained or rehabilitated to fully exploit the GHG mitigating capabilities of shelterbelts.

6. SYNTHESIS AND CONCLUSION

Agricultural activities and land use changes are significant contributors to atmospheric levels of CO_2, CH_4 and N_2O; accounting for about one third of the global anthropogenic impact (Cole et al., 1997). The widespread use of synthetic N fertilizers and tillage operations are the leading causes of accelerated GHG emissions in conventional agricultural systems. Indeed, this study comes at a time of increased awareness of climate change and one in which agricultural producers face the challenge of maintaining (or enhancing) food production needed for a growing human population, while reducing associated GHG emissions (Evers et al., 2010).

In Saskatchewan Canada, shelterbelts have been planted for more than a century (i.e. since 1901) under the provisions of the Prairie Farm Rehabilitation Act (Howe, 1986). Throughout the decades, over 600 million trees have been distributed to Prairies land owners (Wiseman et al., 2009) and it is estimated that there are over 60,000 km of planted shelterbelts throughout the province of Saskatchewan, and considerably more in the entire Canadian Prairie (Amichev et al., 2014). Over the past two decades, shelterbelts have been recognized for their potential to mitigate atmospheric CO_2 through C sequestration in tree biomass (Kort and Turnock, 1999) and in SOC pools (Sauer et al., 2007). However, there have been no studies focusing on the role of shelterbelts in mitigating the emissions of CO_2, CH_4 and N_2O in cultivated soils. Likewise, there are no data on the effects of common shelterbelt tree species on GHG emissions. The research presented in this dissertation addresses some of the knowledge gaps on the dynamics of GHG emissions in shelterbelts planted on agricultural landscapes. This is the first study to directly examine the emissions of CO_2, CH_4 and N_2O in shelterbelts and adjacent cropped fields.

The general goal of this research was to improve our understanding of the capacity of planted shelterbelts to sequester C and mitigate atmospheric GHGs in adjacent cropped fields. To the best of my knowledge, this is the very first study to provide relevant data on the ability of shelterbelts to mitigate GHG emissions in agricultural landscapes. A combination of field experiments and modelling studies were used to accomplish this goal.

6.1 Summary of Findings

Field measurements of soil respiration in nine shelterbelts and adjacent cropped fields (Chapter 3) showed significantly greater CO_2 emissions in the shelterbelts (4.1 Mg CO_2-C ha^{-1} yr^{-1}) than in the adjacent fields (2.1 Mg CO_2-C ha^{-1} yr^{-1}). However, SOC content (0–30 cm soil depth) was 27% greater, representing a net C difference of 28 Mg C ha^{-1} in the shelterbelts than in the cropped fields. The greater SOC content in the shelterbelts was attributed to long-term inputs from tree litter and wind-blown sediments, reduced soil disturbance from agronomic practices, and reduced soil erosion. Consequently, the greater CO_2 emissions in shelterbelts were a combined result of microbial activity on tree litter and respiration from the vast root network of the trees. My results indicate that, although the shelterbelt soils emitted greater amounts of CO_2, it is not necessarily indicative of net ecosystem C losses; rather it shows that the shelterbelt soils are a larger C pool in which the processes of C sequestration and decay occur at a higher rate compared to the cropped fields. This is in agreement with a second study that examined GHG fluxes at various positions along a transect extending from the shelterbelt to the adjacent field (Chapter 4). Summed over the entire study period, cumulative CO_2 emissions were greatest at 0H (8032 ± 502 kg CO_2-C ha^{-1}) and lowest at 5H (3348 ± 329 kg CO_2-C ha^{-1}). However, the decrease in CO_2 emissions at increasing distances away from the shelterbelt was not a linear relationship, but appeared to be controlled by the distribution of SOC along the transect from the shelterbelt to the cropped field. Most notable is the strong correlation between cumulative CO_2 emissions and SOC suggesting that increased soil CO_2 fluxes were associated with SOM turnover along the gradient from the shelterbelt to the field. Moreover, sampling locations nearest to the shelterbelt had the greatest amount of small and fine root biomass, which may have contributed to the greater CO_2 emissions observed within the shelterbelt.

In contrast, the lesser CO_2 emissions observed in the cropped field may be indicative of lower turnover of SOM and/or lower root respiration, which is reflected by the lower SOC and root biomass observed in the cropped field relative to the shelterbelts. Unlike in the shelterbelts where CO_2 emissions were mainly due to the decay of freshly added litter and root respiration, CO_2 emissions from the cultivated fields occurred mainly due to soil disturbance by farm machines during agronomic operations. Consequently, CO_2 emissions in the cropped field occurred mainly due to soil exposure and oxidation of already sequestered soil C following soil disturbance. Agronomic practices such as tillage bring about the breakup of soil-aggregate complexes, which are important in stabilizing SOM; leading to soil C depletion, especially in fields planted to annual crops characterized with low root production (Rutberg et al., 1996).

Studies have reported increased CH_4 oxidation in undisturbed forest and grassland soils compared to intensively cultivated fields (Goulding et al., 1996; MacDonald et al., 1996; Prieme and Christensen, 1999). In agreement with these studies, my results both from field (Chapters 3, 4 and 5) and modeling (Chapter 6) studies indicate that the incorporation of shelterbelts into cultivated fields significantly increased soil CH_4 oxidation, although this may be limited to the area occupied by trees. Results of the comparison of CH_4 fluxes (Chapter 3) showed that CH_4 oxidation was significantly greater in the shelterbelts (-0.66 kg CH_4-C ha^{-1} yr^{-1}) than in the adjacent cropped fields (-0.19 kg CH_4-C ha^{-1} yr^{-1}). The data on CH_4 fluxes at various distances from the shelterbelt to the cropped field showed similar trends. That is, the size of the CH_4 sink decreased with increasing distance from the center of the shelterbelt, where net cumulative CH_4 uptake was significantly ($P < 0.10$) greater within the shelterbelt (i.e. 0H; -1447 ± 484 g CH_4-C ha^{-1}) than in the transition zone (i.e. 0.2H at 0.5H; -752 ± 381 g CH_4-C ha^{-1}) and the cropped field (i.e. 1.5H and 5H; -19 ± 342 g CH_4-C ha^{-1}). The data also showed a strong linear relationship between soil CH_4 exchange and root biomass, bulk density and TOC, suggesting that management practices that affect soil properties such as soil moisture levels and gas diffusivity will also affect soil CH_4 exchange. Planted shelterbelts take up excess soil moisture, improve soil organic matter status and decrease soil bulk density, all of which create a favourable environment for methane consumption.

In contrast, the lower CH_4 uptake observed within the cropped field relative to the shelterbelts may be related to lower SOC and root biomass and greater soil bulk density, which may have affected gas diffusivity at the cultivated soils. The continuous use of heavy farm machinery, fertilizer application and other agronomic practices within the cropped fields may have increased soil bulk density due to soil compaction, which in turn contributed to the low soil CH_4 oxidation in the cropped fields. Earlier studies have shown that agronomic practices such as tillage, fertilization and use of pesticides and herbicides have various degrees of inhibitory effects on CH_4 uptake in arable lands (Hansen et al., 1993; Arif et al., 1996; Mosier et al., 1997; Powlson et al., 1997; Topp and Pattey, 1997; Hütsch, 2001).

Upland soils are an important sink for atmospheric CH_4 as they contribute up to 15% of annual global CH_4 uptake (Powlson et al., 1997). However, management practices applied on the land could increase or decrease soil CH_4 uptake. The intensification of crop production decreases the soil sink size for atmospheric CH_4 (Bronson and Mosier, 1993); however, the incorporation of trees into cultivated fields could significantly increase soil CH_4 sink size through the removal of excess soil moisture, increase in SOM and decrease in soil bulk density.

The results of field studies (Chapters 3, 4 and 5) and the modeling study (Chapter 6) indicated that planted shelterbelts reduced N_2O emissions in cultivated fields. The data showed significantly greater annual N_2O emission (2.5 kg N_2O-N ha^{-1} yr^{-1}) in the cultivated fields and these emissions decreased with increasing proximity to the shelterbelts (0.65 kg N_2O-N ha^{-1} yr^{-1} at 0H). The reduced N_2O emissions in the shelterbelts is likely related to the following: (i) areas planted to shelterbelts were excluded from N fertilization (ii) trees are deep rooting and can remove residual or excess N that would otherwise be available for losses - eventually returning this N back to the soil through litter fall, and (iii) reduced heterotrophic denitrification due to reduced soil moisture in the shelterbelts and surrounding transition zones. Similar findings have been reported by Beaudette et al. (2010), who observed a significant decrease in soil moisture and N_2O emissions that were three times greater in mono-cropping systems than in tree-based intercropping (TBI) systems.

Conversely, the increased N_2O emissions in the cropped fields were mainly due to various rates of N-fertilization and increased soil moisture during spring thaw. Whereas increased N_2O emissions following the thawing of frozen soils in early spring and after fertilization during seeding operations were significant contributors to the cumulative seasonal N_2O emissions in cropped fields, soils beneath shelterbelts were less responsive to the spring snowmelt events. Moreover, in cropped fields that received fertilization, N_2O emissions quickly increased in months following N fertilization and declined to background levels towards the end of the season.

Although shelterbelts have been shown to reduce N_2O emissions from cultivated fields, the magnitude of this effect varied with tree species. The modeling work (Chapter 6) showed that N_2O exchange in shelterbelt soils was related to foliar N concentration, where trees with low foliar N (e.g. conifers) emitted less N_2O compared to N-fixing trees with high foliar N (e.g. caragana). This was in agreement with the field study that examined N_2O flux in caragana compared to coniferous shelterbelts under elevated soil moisture conditions (Appendix A.). Summed across the entire two-year study period, annual N_2O emissions were significantly greater within the caragana trees than in the non-N-fixing trees. Moreover, whereas N_2O emissions were significantly greater at the wet (irrigated) sections of the caragana shelterbelt (2.25 kg N_2O-N ha^{-1}) compared to the dry (non-irrigated) sections (0.99 kg N_2O-N ha^{-1}), no significant differences in cumulative soil N_2O emissions were observed in the wet and dry sections of the conifer shelterbelts. The greater N_2O response to elevated soil moisture in the caragana shelterbelts may have been enabled by greater concentrations of NO_3-N and NH_4-N observed in soils under caragana - a result of its N-fixing capability.

Clearly, shelterbelts composed of pure stands of N-fixing trees can produce significantly greater N_2O emissions than coniferous species, especially under elevated soil moisture conditions. Planting shelterbelts composed of pure stands of N-fixing trees (e.g. caragana) may be beneficial in terms of C sequestration; however, they may be significant sources of atmospheric N_2O emissions, which may constitute an even greater environmental hazard (Albrecht and Kandji, 2003). However, there may be opportunities for N-fixing trees to become great environmental assets by inter-planting N-fixing trees with non-N-fixing trees. Mixing tree plantings with N-fixing trees not only improves the N nutrition of the beneficiary trees, but can also decrease N_2O loses due to excess available N from the N-fixing trees.

Throughout the entire study, soil-derived GHG emissions within the shelterbelts varied across tree species, age of shelterbelts and the orientation of shelterbelts (Table 6.1). For example, the coniferous tree species in Outlook and Prince Albert (i.e. Scots pine and white spruce, respectively) emitted considerably lower N_2O than the caragana species and other tree species in these study locations. As well, soils within older shelterbelts tended to exhibit greater CH_4 uptake, more C content and increased soil respiration. Similarly, considerable variations in soil-derived GHG emissions were observed within the cropped field plots, however these variations were mainly due to differences in land use (i.e. crop type) and N fertilizer application rates (Table 6.2).

Although not part of the objectives of this study, an attempt was made to determine the impact of shelterbelt species and orientation on GHG emissions. The nine shelterbelts in this study were categorized into shrubs (caragana), conifers (Scots pine) and mixed (all shelterbelts with more than one tree species) (Table 6.3). The shelterbelts were further grouped based on planting orientation namely east-west and north-south. A multivariate analysis of variance showed that fluxes of CO_2, CH_4 and N_2O emissions were significantly ($P < 0.10$) affected by shelterbelt tree species. Moreover, soil CH_4 and N_2O were significantly affected by shelterbelt orientation ($P < 0.10$), whereas CO_2 emissions did not vary due to orientation.

Table 6.1. Summary of mean annual soil-derived greenhouse gas emissions from shelterbelts across the three study locations as influenced by shelterbelt species, age and orientation.

Location	Site ID	Main species	Age (yr)	Shelterbelt orientation†	Planting space (m)	CO_2 (Mg ha^{-1} yr^{-1})	CH_4 (kg ha^{-1} yr^{-1})	N_2O (kg ha^{-1} yr^{-1})
Outlook	O-SB1	Scots pine	19	E - W	2.5	3.42 (0.24)‡	-0.79 (0.02)	0.16 (0.07)
	O-SB2	Green ash and caragana	41	N - S	2.7	3.75 (0.14)	-0.81 (0.18)	2.35 (0.43)
	O-SB3	Caragana	34	N - S	1	4.74 (0.30)	-1.14 (0.13)	0.70 (0.17)
Saskatoon	S-SB1	Jack pine and caragana	40	N - S	3.5	5.01 (0.28)	-1.00 (0.09)	0.22 (0.03)
	S-SB2	Mixed spp	38	N - S	2	2.50 (0.04)	-0.52 (0.14)	0.12 (0.08)
	S-SB3	Caragana	38	N - S	1	8.32 (0.31)	-1.06 (0.12)	0.49 (0.09)
Prince Albert	P-SB1	Poplar and caragana	27	E - W	1.6	3.16 (0.50)	-0.21 (0.06)	0.55 (0.18)
	P-SB2	White spruce	41	N - S	2	3.56 (0.49)	-0.14 (0.03)	0.47 (0.12)
	P-SB3	Caragana	19	N – S	2	2.28 (0.37)	-0.24 (0.06)	0.79 (0.27)

† E-W (East-West); N-S (North-South)

‡ Numbers in parenthesis represent standard deviation.

Table 6.2. Summary of mean annual soil-derived greenhouse gas emissions from cropped fields across the three study locations as influenced by crop type and fertilizer N application rates.

Location	Site ID	Crop		Fertilizer N (kg ha^{-1})		CO_2	CH_4	N_2O
		2013	2014	2013	2014	(Mg ha^{-1} yr^{-1})†	(kg ha^{-1} yr^{-1})	(kg ha^{-1} yr^{-1})
Outlook	O-CF1	Wheat	Soybean	50	0	2.45 (0.13)	-0.79 (0.09)	1.53 (0.47)
	O-CF2	Soybean	Wheat	0	70	1.70 (0.15)	-0.14 (0.02)	2.96 (1.36)
	O-CF3	Soybean	Wheat	0	70	1.20 (0.32)	-0.13 (0.03)	4.60 (1.63)
Saskatoon	S-CF1	Haskap	Asparagus	0	70	2.00 (0.12)	-0.09 (0.18)	1.10 (0.26)
	S-CF2	Haskap	Asparagus	0	70	2.27 (0.04)	-0.12 (0.08)	1.75 (0.70)
	S-CF3	Haskap	Asparagus	0	70	2.30 (0.03)	-0.14 (0.09)	4.43 (1.43)
Prince Albert	P-CF1	Corn	Wheat	50	100	2.45 (0.21)	-0.09 (0.03)	1.48 (0.16)
	P-CF2	Barley	Common bean	40	100	2.48 (0.19)	-0.12 (0.06)	1.66 (0.19)
	P-CF3	Barley	Common bean	40	100	2.38 (0.11)	-0.13 (0.04)	2.92 (0.83)

† Numbers in parenthesis represent standard deviation.

Interaction effects of shelterbelt species by orientation were significant ($P < 0.10$) for CH_4 and N_2O fluxes, but no interaction effects were observed for CO_2 flux. Differences in gas exchange due to orientation may be related to microclimatic differences created from the interception of wind speed. The micro-climate in the sheltered zone is altered as a result of wind speed reduction and changes in turbulent transfer rates, which in turn, are influenced by shelterbelt orientation (Cleugh and Hughes, 2002). Since the interaction effects of shelterbelt species by orientation were significant for soil CH_4 and N_2O fluxes, it is likely that the observed differences are a combined effect of species, orientation and perhaps, other factors not considered, such as differences in tree age, soil N composition, time of the day and season of the year when the data was collected. Additional studies are needed to investigate the effect of tree species and shelterbelt orientation on soil-derived CO_2, CH_4 and N_2O fluxes in shelterbelts.

Table 6.3 Effect of shelterbelt species and orientation on exchange of soil-derived CO_2, CH_4 and N_2O

Factor	CO_2 (Mg CO_2-C ha^{-1} yr^{-1})	CH_4 (kg CH_4-C ha^{-1} yr^{-1})	N_2O (kg N_2O-N ha^{-1} yr^{-1})
Species			
Conifers	2.96 (0.27)c†	-0.66 (0. 97)b‡	0.14 (0.17)b
Shrubs	5.11 (0.56)a	-0.81 (0.10)a	0.66 (0.14)a
Mixed	4.23 (0.37)b	-0.69 (0.08)ab	0.67 (0.14)a
Orientation			
East-West	4.22 (0.46)a	-0.90 (0.1)a	0.19 (0.17)b
North-South	3.87 (0.27a	-0.57 (0.06)b	0.63 (0.11)a

† Within columns—and for each Factor— numbers followed by the same lower case letter are not significantly different.

‡ Negative sign indicates uptake; Numbers in parenthesis represent standard deviation.

The role of shelterbelts on GHG mitigation in a 688 ha model farm during a 60-year period was estimated (Chapter 6). The estimate considered the combined contributions through C sequestration in tree biomass and soil as well as reduction of GHG. The data showed that when shelterbelts occupied 5% of the cropped field, overall farm emissions were reduced by 33,226 – 11,897 Mg CO_2e representing 8.2 – 23% reduction in farm GHG emissions; depending on which tree species were used. It was determined that the faster growing trees (e.g. hybrid poplar) were more effective in accumulating C in tree biomass and soil and in mitigating soil GHG emissions.

6.2 Conclusion

Clearly, this research has demonstrated that planted shelterbelts have significant potential to enhance C storage and reduce nitrous oxide emissions, while maintaining a strong methane sink. The age of shelterbelts in this study ranged from 19 to 41 years, thus the greater SOC (28 Mg ha^{-1}) reported in shelterbelts may represent an annual accrual of 0.7 to 1.5 Mg ha^{-1} yr^{-1} over the life of the shelterbelts, assuming that net C storage in the crop fields over time is negligible. Moreover, averaged across study sites and years of study, total exchange of non-CO_2 GHGs in shelterbelts was reduced by 0.55 Mg CO_2e $ha^{-1}yr^{-1}$ as compared with cropped fields. Given that there are about 60,000 km of planted shelterbelts throughout the province (Amichev et al., 2014) and an estimated shelterbelt width of 5-m (re-calculated from Kort and Turnock, 1999), then annual GHG mitigation potential of shelterbelts planted in Saskatchewan, Canada can be estimated at 495,000 Mg CO_2e yr^{-1}, (i.e. 330,000 Mg C yr^{-1} through SOC storage and 165,000 Mg CO_2e yr^{-1} through reduced emissions of non-CO_2 GHGs).

6.3 Future Research

The research presented in this dissertation provides a first insight into the role of planted shelterbelts in mitigating GHG emissions in cultivated soils. Long-term monitoring studies are needed to understand how the role of shelterbelts in mitigating GHG emissions is affected under various climate conditions. The Canadian Prairies are prominent for high inter-annual variability in climate and extreme weather conditions which can substantially affect the rate of soil-atmosphere GHG exchange in shelterbelts and cropped fields. Although the field studies presented here elucidate typical GHG trends over two seasons, long-term, multiple year monitoring is needed for a more accurate understanding of the role that shelterbelts play in mitigating GHG emissions in the agricultural landscape. Such data will be useful for developing GHG models with more accurate predictions of the role of shelterbelts, especially under a changing climate.

Furthermore, shelterbelts are only one of many agroforestry systems in temperate North America and the world at large. Yet there is still a dearth of information on the contribution of other agroforestry systems such as silvopasture, alley cropping, forest farming and riparian forest buffers to the mitigation of GHG emissions in agricultural soils. Investigating the impact of these systems on GHG emissions will not only expand the scope of this work, but will bring us closer to understanding the full potential of Agroforestry for mitigating agricultural GHG emissions. Such studies must consider the effect of tree species, climate and soil types.

Nitrogen-fixing trees have the potential to increase soil C storage through enhanced biomass production and litter inputs to soil, but the observation that the inclusion of N-fixing trees may induce greater N_2O emission requires further study. Over a 100-year period, 1 kg N_2O and 1 kg CH_4 will have the same global warming effect as 296 kg and 23 kg CO_2, respectively; and N_2O is long lived in the atmosphere and more efficient in absorbing infrared radiation. Hence, taking on a land management practice that is effective in increasing carbon storage but results in even small increases in N_2O or CH_4 emissions may result in a net negative effect on the atmospheric radiative force (Malhi et al., 2010). Clearly, the challenges and success of agroforestry in tackling climate change will hinge on adequate understanding of trade-offs between C sequestration and the emission of trace gases such as CH_4 and N_2O.

There is a need to complete a study addressing the economic and ecological costs and benefits of integrating shelterbelts into cultivated fields. Such a study should address issues around the adequate proportion of land that should be planted to shelterbelts without jeopardizing crop yields. For example, based on an assumption that shelterbelts occupied 5% of cropland, grew to a height of 12-m and lived to an age of 50 years, Stoeckeler (1965) concluded that shelterbelts required an average of 18 years to increase net yields of cereal grains and make up for losses due to land occupation. More of such data can inform policy decisions on shelterbelt establishment and create more incentive for landowners to adopt the idea of including shelterbelts in the cultivated fields.

The manual, chamber-based method of measuring GHG emissions can be labour intensive and time consuming. Estimations of daily and seasonal emissions are upscaled from measurements taken at short-time intervals — typically less than one hour — two to three times per week, which limits the accuracy and precision of these estimates. Future research should adopt the use of more recently developed equipment such as automated chambers, which allow for semi-continuous measurement of GHG fluxes. Although the high monetary value of these newer instruments can be limiting, the improved accuracy and precision of daily and seasonal measurements will facilitate future field-scale accounting of soil GHG fluxes (David, 2014).

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APPENDIX A. EFFECT OF ELEVATED SOIL MOISTURE ON SOIL-DERIVED GREENHOUSE GAS EMISSIONS FROM SHELTERBELTS IN THE SEMI-ARID CANADIAN PRAIRIE

A.1 Preface

Based on discussions in previous chapters, it is clear that planted shelterbelts increase soil C concentrations and contribute significantly to reducing GHG emissions from soils in adjacent cropped fields. However, the response of shelterbelt soils to elevated soil moisture conditions, which is known to be a major control of GHG emissions in agricultural soils, is not well understood. A predicted consequence of climate change is increased alterations in precipitation, yet there is limited data on the GHG response of the organic matter-rich shelterbelt soils under elevated moisture conditions. Given that roughly half of total GHG emissions from primary agriculture are linked to agricultural soils, any efforts to mitigate increasing atmospheric concentrations of GHGs from agriculture requires a thorough understanding of the hydrological influences on soil GHG exchange. The objective of the present study was to quantify and compare CO_2, CH_4 and N_2O fluxes from shelterbelts under elevated soil moisture (irrigated) and semi-arid conditions (non-irrigated). Studies were carried out at the Canada-Saskatchewan Irrigation Diversification Centre (CSIDC) near Outlook, Saskatchewan using non-steady state vented chambers.

A.2 Abstract

Soil moisture is known to be a major control of GHG emissions from agricultural soils. However, there is little data regarding GHG exchange from the organic matter-rich soils characteristic of shelterbelts—especially under elevated soil moisture conditions. In the present study, I quantified CO_2, CH_4 and N_2O fluxes from shelterbelts under elevated soil moisture (irrigated) and semi-arid (rainfed) conditions. Studies were carried out at the Canada-Saskatchewan Irrigation Diversification Centre (CSIDC) near Outlook, Saskatchewan. Non-steady state vented chambers were used to monitor soil GHG fluxes from three shelterbelts in 2013 and 2014. The shelterbelts consisted of a single row of caragana with a north–south orientation and a single row of Scots pine with either a north–south or east-west orientation. Each shelterbelt was divided into two areas based on whether or not it received irrigation. During the 2-year study period, N_2O emissions from the irrigated shelterbelts (IR-SB) (0.93 kg N_2O-N ha^{-1}) were significantly greater than those from the rainfed shelterbelts (RF-SB) (0.99 kg N_2O-N ha^{-1}). Whereas irrigation did not stimulate N_2O emissions in either of the Scots pine shelterbelts, a greater N_2O response to elevated soil moisture was observed in the caragana shelterbelt , likely enabled by greater concentrations of available N (i.e., NO_3-N + NH_4-N) in the shelterbelt soil—due to biological N_2 fixation by the caragana. Soil CH_4 oxidation was significantly lower in the IR-SB compared to the RF-SB (-0.85 and -1.20 kg CH_4-C ha^{-1}, respectively), during the entire study period. Irrigation activities stimulated CO_2 production/emission in 2014, but had no effect on CO_2 emissions during the much drier 2013 season. Nevertheless, soil CO_2 across the entire study period was tendentially greater in the IR-SB than in the RF-SB sites. Correlation analyses indicate a strong dependence of CO_2 and CH_4 fluxes on soil moisture in both IR-SB and RF-SB sites; however, no significant correlations between N_2O flux and soil moisture were observed in either the IR-SB or RF-SB sites.

A.3 Introduction

Afforested marginal soils have the potential to exchange significant amounts of the GHGs CO_2, CH_4, and N_2O (Kim et al., 2010; Evers et al., 2010), which are produced and/or consumed via microbial processes in the soil. However, the amount of soil-atmosphere gas exchange depends largely on soil physical factors—with soil moisture and temperature being important drivers of both the production/consumption and atmospheric exchange of GHGs (Davidson and Swank, 1986). Indeed, soil water content and temperature have a strong influence on the activity of both the soil microbial community and plant roots. Gas diffusivity, which varies inversely with soil water content affects soil aeration and the movement of gases in the soil, and thus indirectly controls the capacity of the soil to produce or consume CO_2, N_2O and CH_4 (Smith et al., 2003). Whereas CO_2 fluxes from afforested soils are usually much larger than the fluxes of CH_4 and N_2O, the latter two GHGs have global warming potentials (GWP) that, over a 100-year timeframe, that are 25 and 298 times that of CO_2, respectively (IPCC, 2007)—thus magnifying their potential impact on global radiative forcing.

Shelterbelts—consisting of one or more rows of trees and/or shrubs planted to provide protection from the wind—have been used for centuries to regulate environmental conditions in agricultural landscapes and provide a variety of economic, social, and environmental benefits that are valued by landowners and society (Mize et al., 2008). More recently, shelterbelts have been recognized for their potential to offset increasing concentrations of atmospheric CO_2 by storing photosynthetically fixed carbon (C) in woody biomass and soil organic matter (Sauer et al., 2007; Kort and Turnock, 1999; De Brauw, 2006). Tree plantings also have been reported to mitigate N_2O emissions and enhance CH_4 uptake in agricultural fields (Mutuo et al., 2005; Evers et al., 2010; Amadi et al., 2016). The role of shelterbelts in C sequestration and the mitigation of agricultural greenhouse gases (GHGs) is due mainly to the high rates of C accrual in biomass and soil through litter fall, entrapment of windblown sediments, and modification of the local microclimate and root activity (Zhang et al., 2012; Schoeneberger 2009; Kowalchuk and De Jong, 1995). As in forest soils, the establishment of shelterbelts results in the creation of an LFH layer on top of an organic matter-rich A-horizon (Anderson 1987; Peichl et al., 2010).

The role of soil water as a major control of GHG emissions from soils in the Canadian Prairies has been well studied (Hao et al., 2001; Liebig et al., 2005; Ellert and Janzen, 2008; Sainju et al., 2012); however, relatively little research has focused on understanding the dynamics of GHG emission from the organic matter-rich shelterbelt soils under elevated soil moisture conditions. In the agricultural landscape, it is not uncommon to encounter soil and/or management factors that promote elevated soil moisture conditions in the soil. For example, the short-term flooding of depressional areas in landscapes with variable topography (Wang et al., 2003) or the application of irrigation water in shelterbelts established along the borders of cropped fields. Elevated soil moisture coupled with warm temperatures favour soil microbial activity, which in turn, may alter the dynamics of soil GHG production/consumption/emission (Smith *et al*., 2003).

Nitrous oxide is produced within soils as a result of naturally occurring microbial processes; namely, nitrification and denitrification. Nitrifying bacteria are active under aerobic conditions and produce N_2O during the oxidation of ammonium (NH_4^+) to nitrate (NO_3^-). Denitrification, on the other hand, produces N_2O as an intermediate during the reduction of NO_3^- under anaerobic conditions. As a result, N_2O emission are positively correlated with factors that influence microbial activity—including nitrogen availability (Bouwman, 1996; Dobbie et al., 1999), soil water content (Corre et al., 1996; Dobbie et al., 1999), aeration status (Linn and Doran, 1984), and soil temperature (Dobbie and Smith, 2003; Yates et al. 2007). These factors are both spatially and temporally variable; consequently, soil-derived N_2O emissions also are inherently variable in both space and time (Yates et al., 2006; 2007). Indeed, soil N_2O emission patterns are often characterized by small areas ('hot-spots') and brief periods ('hot moments') that account for a high percentage of the total emissions (Groffman et al., 2009; Braker and Conrad, 2011; Butterbach-Bahl et al. 2013).

The foregoing discussion suggests that the magnitude of N_2O emissions from shelterbelt soils under elevated soil moisture conditions may rely mainly on soil concentrations of available N (i.e., NO_3^- and NH_4^+). For example, shelterbelts composed mainly of N_2-fixing trees (e.g., *Caragana arborescence*) may contain more available soil N relative to shelterbelts composed of non-N_2 fixing trees (e.g., *Pinus sylvestris* L.) (Vlassak et al., 1973; Albrecht and Kandji, 2003; Peichl et al., 2010; Moukoumi et al 2013). Consequently, under elevated soil moisture conditions, soil-derived N_2O emissions under N_2-fixing trees may be greater than those from soils with a more limited supply of available N (Malhi et al. 1990; Peichl et al., 2010).

Upland soils are natural sinks for atmospheric CH_4 due to oxidation processes facilitated by methanotrophic microbes under aerobic soil conditions (Suwanwaree and Robertson, 2005; Fowler et al., 2009). However, short-term increases in soil water content may increase soil CH_4

production, thereby reducing the soils' capacity for CH_4 uptake (Liu et al., 2006; Sainju et al., 2012). For example, in a study quantifying methane emissions from Canadian prairie and forest soils under short term flooding conditions, Wang and Bettany (1997) found increased CH_4 emission rates shortly after snowmelt in the spring and from low slope positions after rainfall in the summer. Reductions in CH_4 uptake also have been linked to increases in soil N availability (Bronson and Mosier, 1994; Sainju et al., 2012) due to competition between ammonia and methane oxidizing microbial communities (Hütsch et al., 1993). Consequently, shelterbelts that are characterized by increased available soil N (as is the case with N_2-fixing tree species) may have a lower capacity for CH_4 uptake—a condition that may be exacerbated under elevated soil moisture conditions.

Soil water content, particularly water-filled pore space (WFPS), also can influence rates of CO_2 emission. In dry soils, soil respiration may be limited by the slow diffusion of soluble C substrates in thin water films (Davidson et al., 2006). On the other hand, the addition of water to shelterbelt soils (either as precipitation or irrigation) can elicit substantial increases in total respiration—reflecting enhanced decomposition of the LFH layer and an increase in substrate availability (Davidson et al., 2006; Cisneros-Dozal et al., 2007). However, under saturated conditions, a large proportion of the pores are filled with water, thus restricting soil aeration and respiration, and slowing the diffusion of CO_2 to the soil-atmosphere interface. Consequently, CO_2 fluxes generally decrease under saturated conditions, though not necessarily by as much as when lack of water is the limiting factor (Smith *et al*., 2003).

Natural and anthropogenic modifications of soil hydrology in shelterbelt ecosystems may significantly alter the rates of production and consumption of GHGs. In Canada, roughly half of total greenhouse gas emissions from primary agriculture are linked to agricultural soils; therefore, any efforts to mitigate increasing atmospheric concentrations of GHGs from agriculture requires a thorough understanding of the hydrological influences on soil GHG exchange (Malhi et al; 2010; Helgason et al. 2005; Liebig et al. 2005). Furthermore, climate models have predicted that most areas in Temperate North America will probably experience the greatest alterations in precipitation under changing climate scenarios (IPCC, 2007); therefore, it is imperative to assess the impact of elevated soil moisture on GHG exchange in agro-ecosystems—including shelterbelts. In the present study, I monitored GHG emissions from shelterbelts that received irrigation water and compared them against non-irrigated sections of the same shelterbelts in the Prairie ecozone of

Saskatchewan, Canada. The objective of the study was to quantify and compare fluxes of CO_2, CH_4 and N_2O from shelterbelts under irrigated conditions to those from dryland (rainfed) shelterbelts.

A.4 Materials and Methods

A.4.1 Study site

The study was carried out at Canada Saskatchewan Irrigation Diversification Centre (CSIDC), in Outlook, SK. The CSIDC is located in the moist mixed grassland Ecoregion of Saskatchewan, Canada (51^o 29’ N, 107^o 03’ W), with an average annual air temperature of 12.5°C and cumulative annual precipitation of 278 mm during the April to October sampling season (based on 1981-2010 climate norms; Environment Canada, 2015). Average annual air temperatures during the 2013 and 2014 seasons were 16.1°C and 14.0°C, respectively; cumulative annual precipitation during the 2013 and 2014 sampling seasons were 180 mm and 326 mm, respectively. Soils at the site are classified as Orthic Dark Brown Chernozems, a mix of Asquith and Bradwell Association, with moderately sandy loam textures. They consist of well drained soils formed mainly in wind deposited sands and loamy lacustrine materials on a slightly undulating topography (Soil classification working group, 1998).

The study included three shelterbelts: a single row of caragana (C. *arborescence*) running north to south along the western border of a field planted to wheat in 2013 and canola and soybean in 2014 (C_N); a single row of Scots pine (*P. sylvestris* L.) running north to south along the western border of a field planted to soybean in 2013 and wheat in 2014 (SP_N) and a single row of Scots pine running east to west along the southern border of the same field (SP_E). Each shelterbelt consisted of an irrigated (IR) section and a rainfed (RF) section. Details of shelterbelt design and the characteristics of each shelterbelt are summarized in Table A.1.

The irrigated sections of the Scots pine shelterbelts (SP_N-IR and SP_E-IR) received a total of 62.5 mm and 37.5 mm of irrigation water in 2013 and 2014, respectively; the irrigated section of the caragana shelterbelt (C_N-IR) received a total of 50 mm and 75 mm of additional water in 2013 and 2014, respectively. The rainfed sections of the shelterbelts (C_N-RF, SP_N-RF and SP_E-RF) served as a reference, representing the status of GHG exchange under normal precipitation regimes.

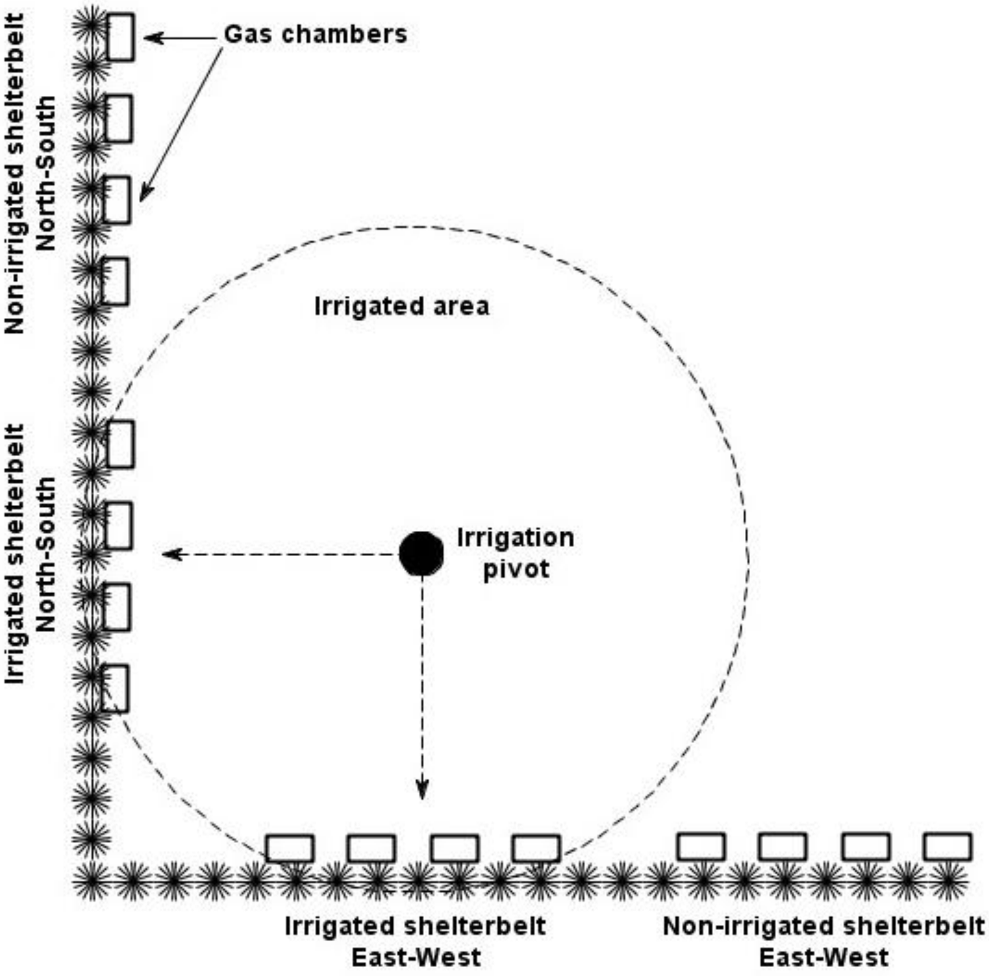

Fig. A.1 Experimental layout for studying changes in GHG fluxes in irrigated and rainfed shelterbelts at Outlook SK.

Fig. A.2 On-going irrigation activity on the Scotspine N-S irrigated site

Table A.1. Characteristics of shelterbelts at the three study sites at CSIDC Outlook, Saskatchewan, Canada.

Site Characteristic	Study sites		
Site name	Caragana N-S	Scots pine N-S	Scots pine E-W
Age at start of this study (years)	36	18	21
Number of rows	1	1	1
Mean tree height (m)	6	11	12.5
Mean DBH (cm)	6.8	27.7	30.5
Tree spacing	1	2.5	2.5
Total length of shelterbelt (m)	750	200	435
Soil C (0-30 cm) (Mg ha-1/%)	93.0/2.60	57.1/1.55	60.5/1.66
Soil N (0-30 cm) (Mg ha-1/%)	7.60/0.21	4.56/0.12	4.64/0.13
Soil NH_4-N (µg N g soil^{-1})‡	1.10 (0.17)^{y}	0.70 (0.15)	0.71 (0.18)
Soil NO_3-N (µg N g soil^{-1})†	0.65 (0.14)	0.29 (0.10)	0.32 (0.11)

† Average of values of available N measured on three separate dates (i.e. July 2013, June 2014 and October 2014) during the study period.

‡ Numbers in parenthesis represent standard deviation

A.4.2 Greenhouse Gas Sampling and Analysis

During the fall of 2012, the bases of four rectangular (22-cm × 45.5-cm) gas chambers were installed in both the irrigated and rainfed sections of each of the shelterbelts. The chamber bases were installed along a transect at the center of each shelterbelt (at a spacing of *ca*. 20-m between chambers) and were set into the soil to a depth of 5 cm. Gas samples were collected by attaching a flux chamber (height = 10 cm; Amadi et al. 2016) to the base and withdrawing 20-mL gas samples as soon as the chamber was in place (t_0) and again 20 min later (t_{20}). All green vegetation within the chambers was removed prior to sampling. Gas samples were collected using a 20-mL polypropylene syringe (Monoject^{TM}, Luer lock fitting) fitted with a 25-gauge needle; injected into pre-evacuated 12-mL Exetainer® vials (LabCo Inc., High Wycombe, UK) fitted with butyl rubber stoppers (Rochette & Bertrand, 2003); and returned to the Department of Soil Science at the University of Saskatchewan in Saskatoon for analysis.

Gas sampling started in spring 2013, at the start of the spring thaw, and continued through to soil freeze-up in the late fall in both 2013 (May 10th to October 9th) and 2014 (April 22nd to October 15th). Disturbance of the soil and litter deposits underneath the shelterbelts was minimized during chamber installation and all chamber bases remained in place throughout the two-year sampling period. Gas sampling occurred twice per week during the spring snowmelt period, after which sampling intensity was reduced to once per week throughout the summer and then to once every two weeks during the fall. Immediately following gas sampling, soil moisture and temperature measurements were collected around the gas chambers at a depth of 10 cm. Soil temperature was measured using a stem-style digital thermometer (Reed PS100, Brampton, ON) whereas soil moisture measurements were taken using digital soil moisture meter (HydroSense, Campbell Scientific, Inc., Logan, UT).

At the end of each sampling day, gas samples were transported to the laboratory for analysis using a gas chromatograph (Bruker 450 GC, Bruker Biosciences Corporation, USA) equipped with a thermal conductivity detector (TCD), flame ionizer detector for (FID) and electron capture detector (ECD) for CO_2, CH_4 and N_2O measurements, respectively (Farrell and Elliot, 2008). Gas samples were introduced into the chromatograph using a CombiPAL auto-sampler (CTC Analytics AG, Switzerland) and data processing was completed using the Varian Star Chromatography Workstation (ver. 6.2) software. Daily gas fluxes were estimated by fitting a linear regression equation to the concentration *vs.* time data, with the daily flux calculated as the slope of the regression line. Cumulative fluxes were calculated using an area-under-the-curve (AUC) analysis of the gas production *vs*. time curves (Hintze, 2009; Engel *et al*., 2010).

A.4.3 Soil Sampling and Analysis

During June 2014, four 3.2 cm (i.d.) soil cores (0–100 cm bgs) were collected from within the irrigated and rainfed sections of each shelterbelt and divided into 0–30 and 30–100 cm depth increments. The soil samples were air dried, crushed and ground with a rolling pin to break aggregates; all visible roots were removed and a subsample of soil (~150 g) was passed through a 2-mm sieve. A 20-g subsample of the air dried soil was placed on a ball grinder for five minutes to create a fine powder (< 250 µm) for Total N (TN) and soil organic carbon (SOC) analyses. Bulk density samples were collected using a hand-held core sampler (i.d. = 5.4 cm, height = 3 cm), which were then weighed and dried at 105°C for 24 h. Soil samples (0–30-cm) collected in July 2013, June 2014, and October 2014 were used to monitor soil nitrate N (NO_3-N) and ammonium N (NH_4-N) concentrations, and were treated as described above prior to analysis.

Soil organic C was determined using a LECO C632 Carbon analyzer (Wang and Anderson, 1998), following a 12*M* HCl pretreatment to remove all inorganic C. Total N was determined by dry combustion using LECO TruMac CNS analyzer (Figueiredo, 2008). Total inorganic N (NO_3^--N + NH_4^+-N) were determined using 2.0*M* KCl extraction (Maynard *et al*., 2008) and analyzed colorimetrically (Technicon Autoanalyzer; Technicon Industrial Syatems, Tarrytown, NY, USA). Soil particle size was determined using a modified pipette method (Indorante *et al*., 1990). Measurement of soil pH in water (1:1 paste; Hendershot *et al*., 2008) was completed using a Beckman 50 pH Meter (Beckman Coulter, Fullerton, CA, USA).

A.4.4 Statistical analysis

Differences in gas exchange and soil properties in the irrigated and rainfed sections of the shelterbelts were analyzed using the PROC MIXED SAS 9.4 for RCBD with the treatments (irrigation vs. rainfed) as a fixed effect and block (shelterbelts) as a random effect (SAS Institute Inc, 2013). Degrees of freedom were approximated by the method of Kenward-Roger (ddfm = kr). Treatments were compared using the Fisher Protected Least Significance Difference (LSD) method. Significance was declared at $P < 0.05$.

A.5 Results

A.5.1 Soil and environmental conditions

Average annual air temperatures at the CSIDC site were warmer than the long-term norms in both 2013 and 2014 (+3.6°C and +1.5°C, respectively). At the same time, cumulative annual precipitation at the site was about 35% lower (-98 mm) than normal in 2013, and 17% greater (+48 mm) than normal in 2014. The addition of irrigation water, increased the total water input in the caragana site (C_N-IR; Fig. A.1a) by 28% in 2013 and by 23% in 2014. Likewise, irrigation in the Scots pine shelterbelts (SP_N-IR and SP_E-IR; Figs. A.2a and A.3a, respectively) increased the total water inputs by 35% and 12% in 2013 and 2014, respectively.

Prior to the addition of irrigation water (i.e., from April to June) there were no significant differences in soil moisture between the irrigated and rainfed sections of any of the shelterbelts in 2013 (P = 0.368) and 2014 (P = 0.765). However, the addition of irrigation water resulted in varying levels of soil water content in all three shelterbelts (Figs. A.1b– A.3b). For example, between the months of July and October 2013, volumetric soil water content in the irrigated sections of the shelterbelts was 43% to 45% greater than in the rainfed sections of the respective shelterbelts. Likewise, in 2014, soil water content was 38% to 47% greater in the irrigated sections of the shelterbelts than in the rainfed sections.

In general, soil temperature in the IR-SB followed seasonal trends similar to those in the RF-SB (Figs. A.1c– A.3c) and did not respond to irrigation-induced changes in soil moisture. Indeed, across sites, there was no significant difference in mean soil temperature between the IR-SB and RF-SB sites in either 2013 (P = 0.746) or 2014 (P = 0.886).

Soil organic C content in the upper soil horizons (0–30 cm) tended to be greater (P = 0.0767) in the IR-SB than in the RF-SB (Table A.2). There were no significant differences (P > 0.05) in TN, bulk density, soil pH and C:N ratio between the IR-SB and RF-SB sites. Similarly, soil NH_4-N and NO_3-N concentrations measured in July 2013, June 2014 and October 2014 did not show significant differences (P > 0.05) between the IR-SB and the RF-SB sites (Table A.3).

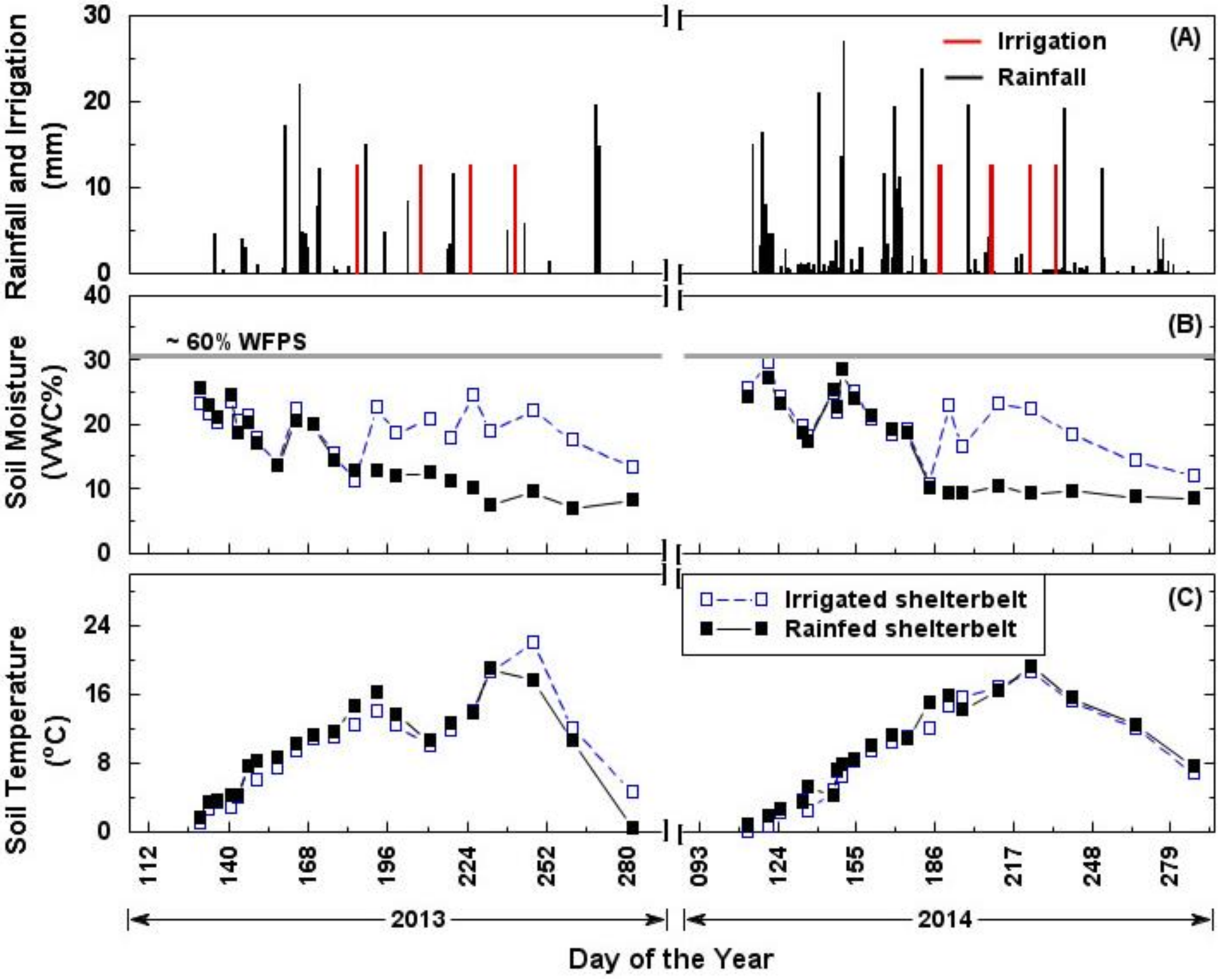

Fig. A.3 Rainfall plus irrigation (A), soil water content (B) and soil temperature (C) measured at the Caragana (N-S) shelterbelt during 2013 and 2014. Volumetric Water Content (VWC%; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both irrigated (open squares) and rainfed (filled squares) sites of the shelterbelt. The grey bar on Panel B represents the approximate range where water filled pore space (WFPS) is at 60%.

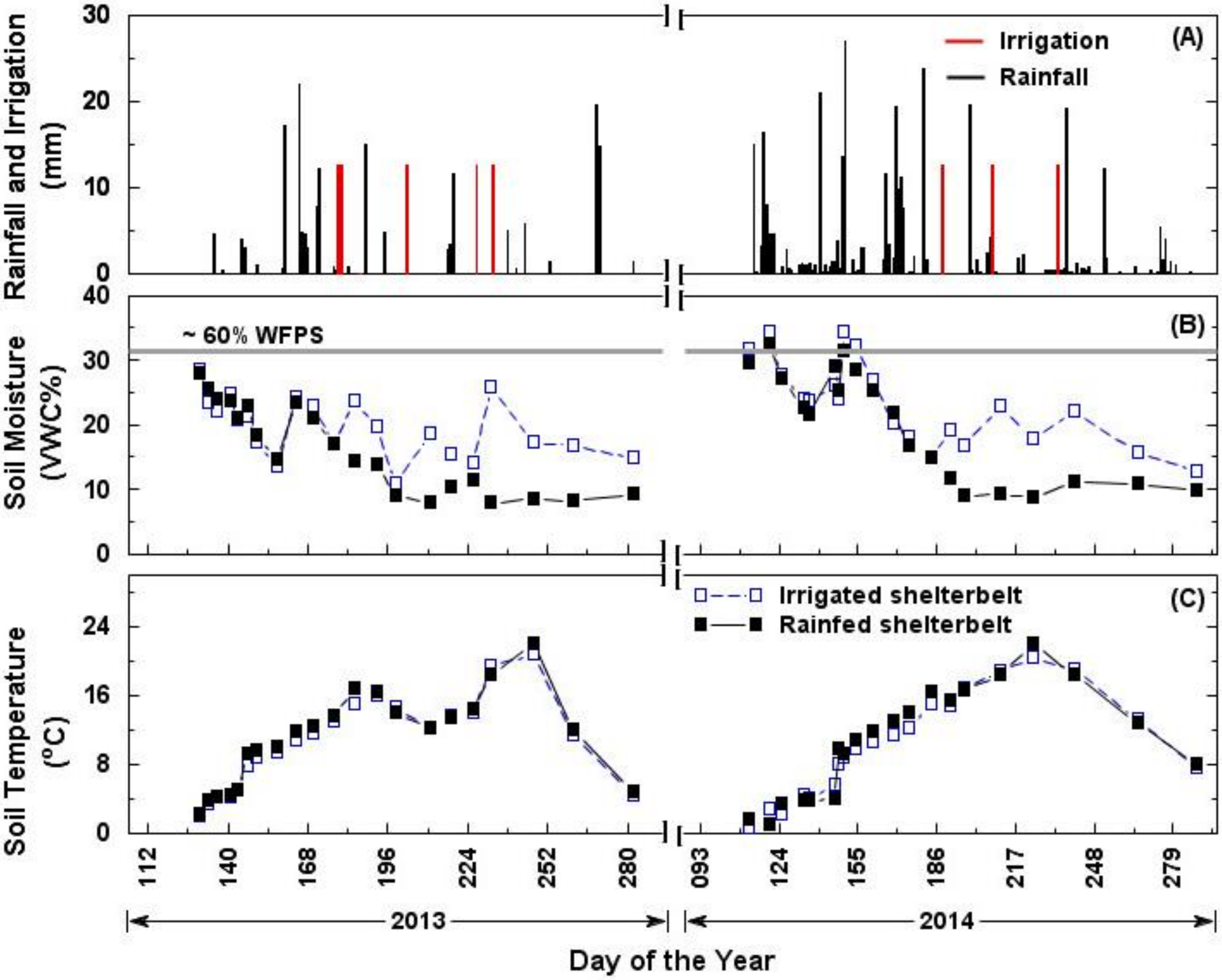

Fig. A.4 Rainfall plus irrigation (A), soil water content (B) and soil temperature (C) measured at the Scots pine (N-S) shelterbelt during 2013 and 2014 sampling seasons. Volumetric Water Content (VWC%; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both irrigated (open squares) and rainfed (filled squares) sites of the shelterbelt. The grey bar on Panel B represents the approximate range where water filled pore space (WFPS) is at 60%.

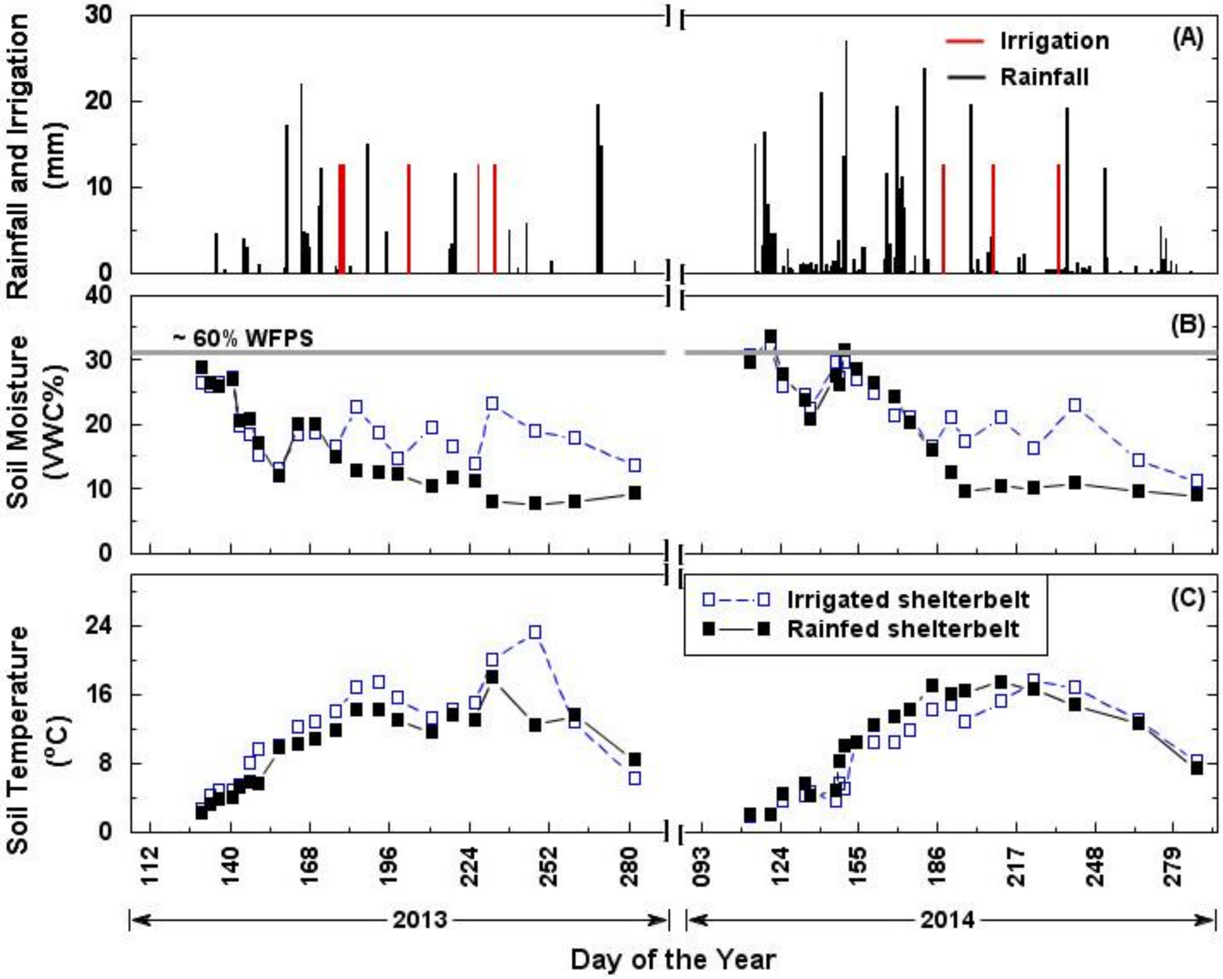

Figure A.5 Rainfall plus irrigation (A), soil water content (B) and soil temperature (C) measured at the Scots pine (E-W) shelterbelt during 2013 and 2014 sampling seasons. Volumetric Water Content (VWC%; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both irrigated (open squares) and rainfed (filled squares) sites of the shelterbelt. The grey bar on Panel B represents the approximate range where water filled pore space (WFPS) is at 60%.

Table A.2. Soil chemical and physical properties in irrigated and rainfed shelterbelts and across tree species at CSIDC Outlook, Saskatchewan, Canada.

Soil property	IR-SB†	RF-SB[z]	p-value
SOC (Mg ha^{-1})	74.4 (21.0)‡	66.1 (18.6)	0.0767
TN (Mg ha^{-1})	5.8 (1.56)	5.4 (1.69)	0.2034
Bulk density (Mg m^{-3})	1.21 (0.08)	1.23 (0.05)	0.4954
Soil pH	7.07 (0.18)	7.04 (0.18)	0.612
C:N ratio	12.85 (1.52)	12.45 (2.11)	0.6126

† Site names: IR-SB, irrigated shelterbelts; RF-SB, rainfed shelterbelts.

‡ Numbers in parenthesis represent standard deviation

Table A.3. Available soil ammonium nitrogen (NH_4-N) and nitrate nitrogen (NO_3-N) within 0-30 cm soil depth measured in July 2013, June 2014 and October 2014 across irrigated and rainfed shelterbelt sites in CSIDC.

Soil property	IR-SB†	RF-SB‡	p-value
July-2013			
NH_4-N (µg N g soil^{-1})	0.83 (0.38)‡	0.89 (0.28)	0.533
NO_3-N (µg N g soil^{-1})	0.44 (0.25)	0.39 (0.19)	0.4356
June-2014			
NH_4-N (µg N g soil^{-1})	0.99 (0.27)	0.97 (0.22)	0.8307
NO_3-N (µg N g soil^{-1})	0.54 (0.23)	0.54 (0.29)	0.9506
October-2014			
NH_4-N (µg N g soil^{-1})	0.68 (0.30)	0.65 (0.26)	0.7093
NO_3-N (µg N g soil^{-1})	0.30 (0.18)	0.31 (0.17)	0.6416

† Site names: IR-SB, irrigated shelterbelts; RF-SB, rainfed shelterbelts.

‡ Numbers in parenthesis represent standard deviation

A.5.2 Soil N_2O, CH_4 and CO_2 exchange

Daily soil N_2O fluxes during the two year study period ranged from -2.3 to +16.5 g N_2O-N ha^{-1} d^{-1}, with negative values indicating uptake (Fig. A.4). In general, daily N_2O emissions were greater in the irrigated section of the caragana shelterbelt than in the rainfed section—especially during the months when irrigation water was applied (Fig. A.4a). Conversely, daily soil N_2O emissions in the Scots pine shelterbelts were relatively unaffected by elevated soil moisture conditions in the irrigated sections of the shelterbelts (Figs. A.4b & A.4c). Cumulative N_2O emissions were significantly ($P = 0.0081$) greater in the IR-SB sites during 2013, but no significant difference in cumulative N_2O emissions was observed in 2014. However, when summed across the 2-year study period, cumulative N_2O emissions were significantly ($P = 0.0161$) greater under the IR-SB than in the RF-SB sites (Table A.4).

Both CH_4 emission and consumption were observed during the study period, with rates ranging from -10 to +55 g CH_4-C ha^{-1} d^{-1} (Fig. A.5)—with negative values indicating uptake. Daily CH_4 fluxes did not show any clear seasonal pattern but did appear to vary with changes in soil water content. For example, CH_4 uptake was generally greater in the rainfed section of the caragana shelterbelt than in the irrigated section (Fig. A.5a). However, this trend was not observed in the Scots pine shelterbelts (Figs. A.5b & A.5c). Large, transient CH_4 emission events were observed in 2013 at all three shelterbelts, and under both irrigated and rainfed conditions. For example, a single large CH_4 emission event at SP_N-IR during August 2013 contributed 72% of the total annual CH_4 emission from this site (Fig. A.5b); at the same time, there was no corresponding peak at the SP_N-RF site, or at any of the other sites (Figs. A.5a & A.5c). Regardless of these emission events, however, soils under all the shelterbelts were small net sinks for CH_4 (Table A.4). In 2013, cumulative CH4 oxidation in the rainfed shelterbelts was significantly ($P = 0.0067$) greater than in the IR-SB, though no significant ($P = 0.4621$) effect on CH_4 oxidation was found between the IR-SB and RF-SB sites in 2014 (Table A.4). Summed across the entire two-year study period, the sink potential (i.e., cumulative CH_4 uptake) of the IR-SB was significantly lower ($P = 0.0342$) than the RF-SB sites (Table A.4).

Throughout the study period, daily CO_2 fluxes in both the IR-SB and RF-SB sites ranged from 0 to 50 kg CO_2-C ha^{-1} d^{-1} (Fig. A.6). Moreover, daily CO_2 fluxes followed similar seasonal trends as soil temperature; i.e., with the highest fluxes occurring during periods of high soil temperature (June to August) and the lowest fluxes occurring during periods of low soil temperature (typically during the early spring and late fall). In 2013, daily CO_2 fluxes in the irrigated sections of all three shelterbelts followed seasonal patterns similar to those observed in the rainfed sections; moreover, irrigation did not appear to stimulate CO_2 emissions in the irrigated sections in during the season (Fig. A.6). Consequently, there was no significant ($P = 0.949$) difference in cumulative CO_2 emissions between the IR-SB and RF-SB sites in 2013 (Table A.4). Conversely, irrigation had a significant effect on the daily CO_2 emissions in 2014, with emissions generally being greater under irrigation during the growing season (Fig. A.6). During 2014, therefore, cumulative CO2 emissions were significantly greater in the IR-SB than in the RF-SB sites (Table A.4). Moreover, summed across both study years cumulative CO_2 emissions in the IR-SB were marginally greater ($P = 0.0634$) than those in the RF-SB (Table A.4).

A.5.3 Relationship between soil gas exchange and soil temperature and moisture

There was a strong positive relationship between daily CO_2 flux and soil temperature in both the IR-SB ($r = 0.59$, $P < 0.001$) and RF-SB ($r = 0.53$ $P < 0.001$) sites. Significant correlations also were observed between the daily CO_2 flux and soil water content in both the IR-SB ($r = 0.30$, $P < 0.001$) and RF-SB (0.44 $P < 0.001$) sites. Whereas soil temperature was negatively correlated with the daily CH_4 flux in both the IR-SB ($r = -0.35$, $P < 0.001$) and RF-SB (-0.30 $P < 0.001$) sites, there was a positive correlation between soil moisture and the daily CH_4 flux ($r = 0.28$, $P = 0.002$ and $r = 0.46$, $P < 0.001$ for the IR-SB and RF-SB sites, respectively). No significant effects of soil temperature or soil moisture on N_2O fluxes were observed in either the IR-SB or RF-SB sites.

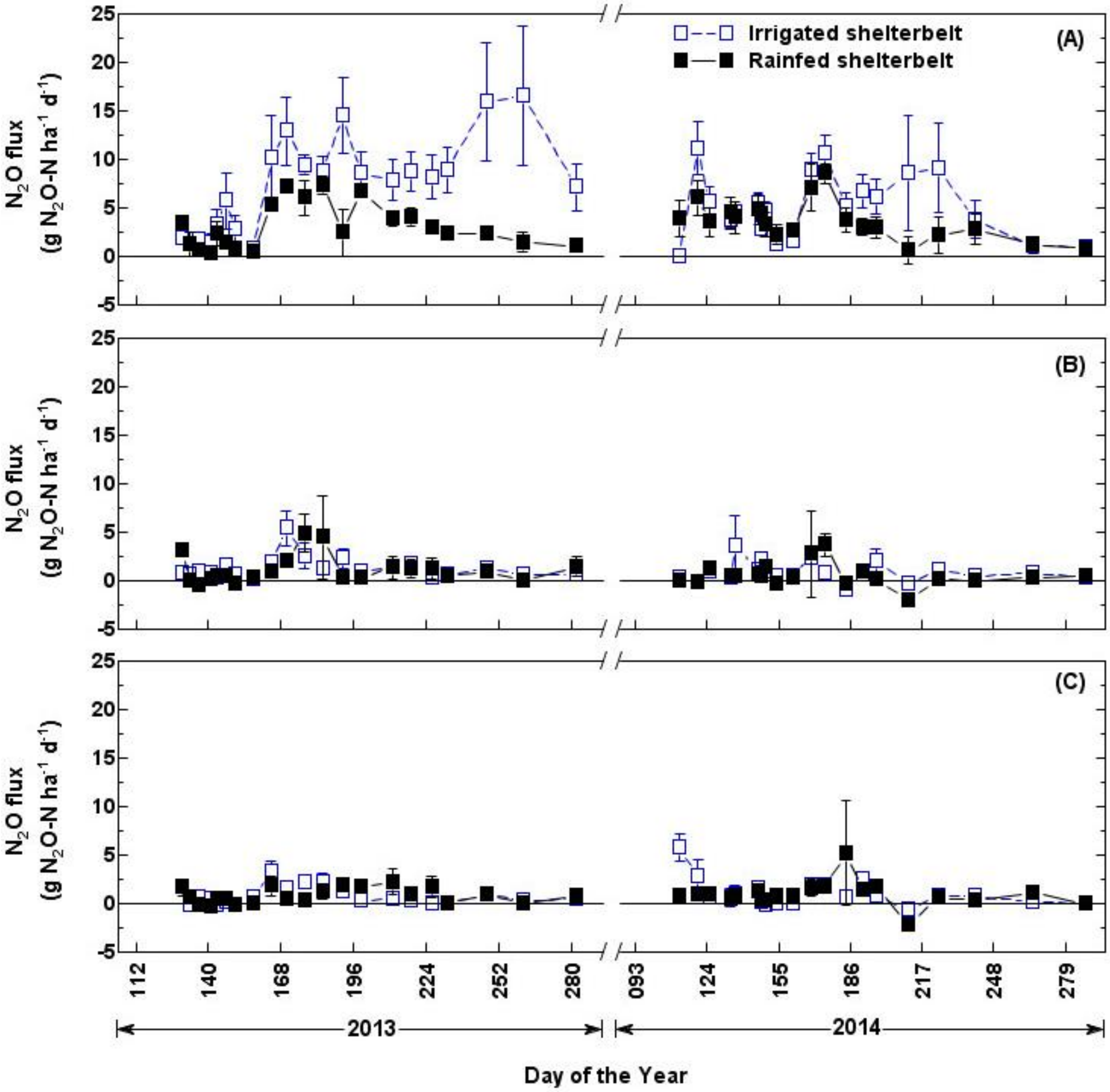

Fig. A.6 Daily soil N_2O fluxes (g N_2O-N ha^{-1} d^{-1}) in the irrigated (open squares) and rainfed (filled squares) sites of the shelterbelts: (A) Caragana (N-S), (B) Scots pine (N-S) and (C) Scots pine (E-W) during 2013 and 2014 study periods. Error bars represent standard error.

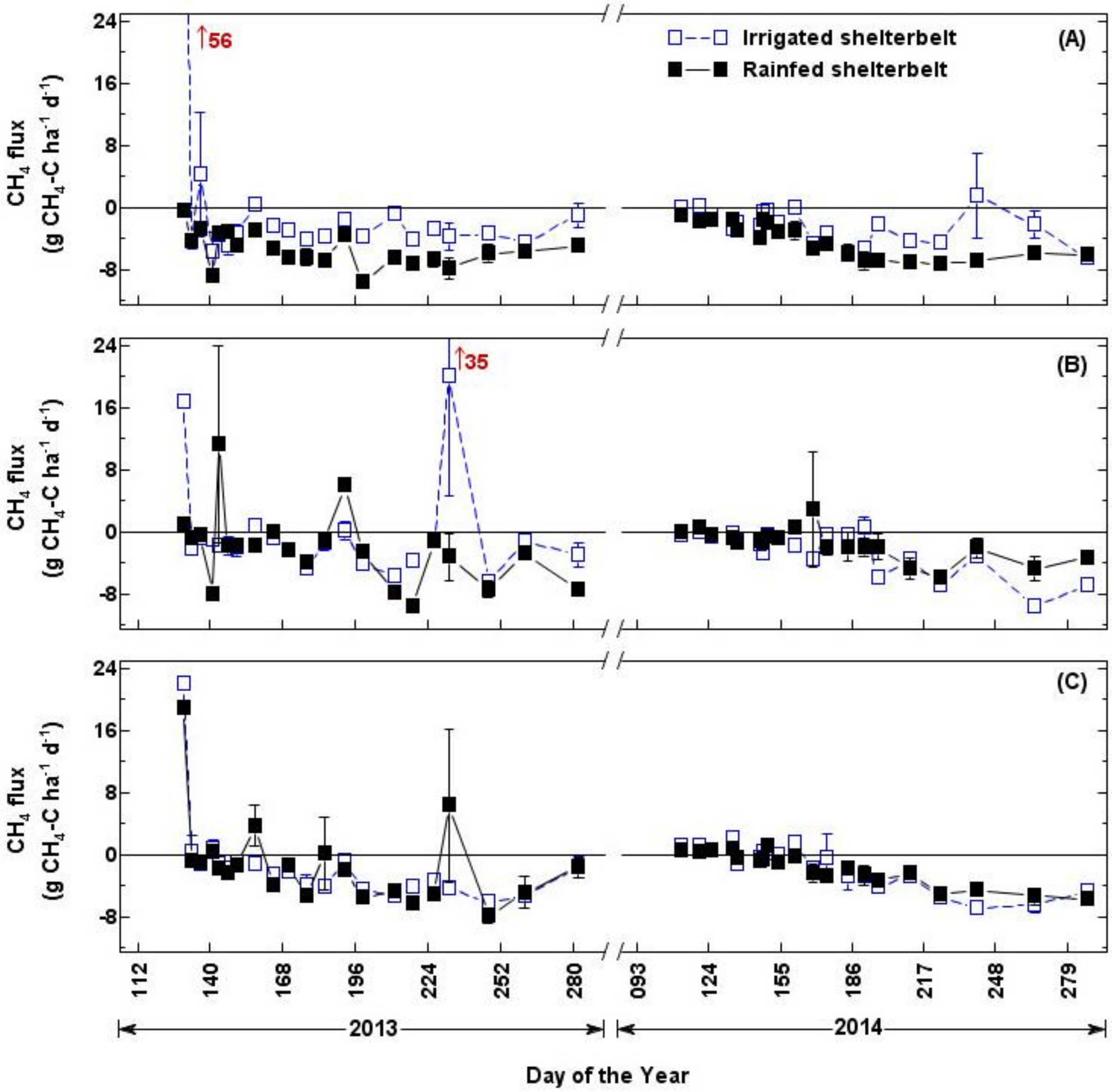

Fig. A.7 Daily soil CH$_4$ fluxes (g CH$_4$-C ha^{-1} d^{-1}) in the irrigated (open squares) and rainfed (filled squares) sites of shelterbelts: (A) Caragana (N-S), (B) Scots pine (N-S) and (C) Scots pine (E-W) during 2013 and 2014 study periods. Error bars represent standard error. Negative numbers indicate uptake.

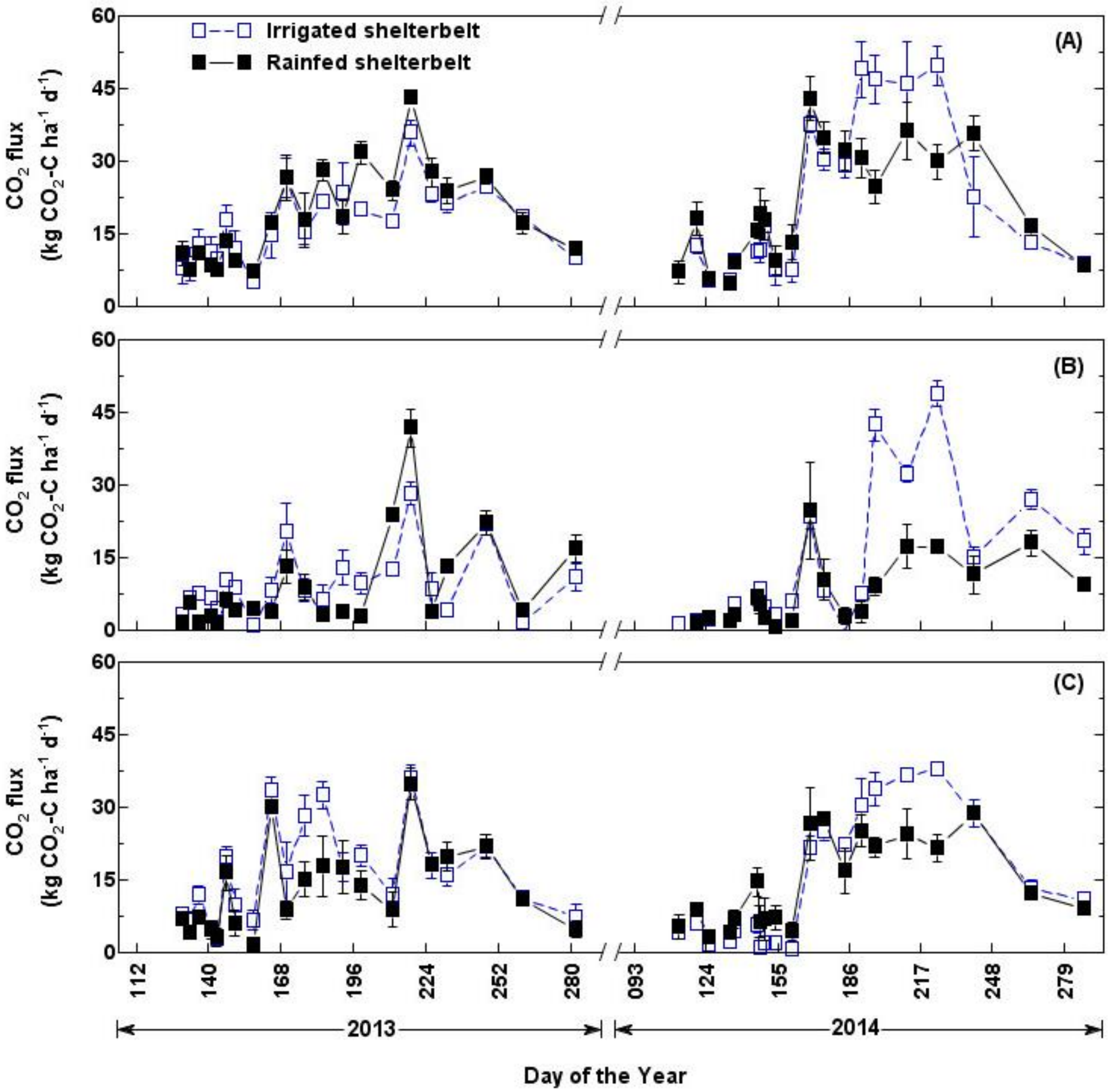

Fig. A.8 Daily soil CO$_2$ fluxes (kg CO$_2$-C ha^{-1} d^{-1}) in the irrigated (open squares) and rainfed (filled squares) sites of shelterbelts: (A) Caragana (N-S), (B) Scots pine (N-S) and (C) Scots pine (E-W) during 2013 and 2014 study periods. Error bars represent standard error.

Table A.4. Cumulative fluxes of N_2O (kg N_2O-N ha^{-1}), CH_4 (kg CH_4-C ha^{-1}) and CO_2 (kg CO_2-C ha^{-1}) in irrigated and rainfed shelterbelts during 2013 and 2014 study periods, and for the sum of both years.

Gas flux	IR-SB†	RF-SB	p-value
N_2O flux (kg N_2O-N ha^{-1})			
2013	0.57 (0.64)‡	0.25 (0.19)	0.0081
2014	0.36 (0.39)	0.24 (0.32)	0.204
Total flux	0.93 (1.00)	0.49 (0.47)	0.0161
CH_4 flux (kg CH_4-C ha^{-1})			
2013	-0.30 (0.24)	-0.59 (0.28)	0.0067
2014	-0.55 (0.15)	-0.62 (0.26)	0.4621
Total flux	-0.85 (0.29)	-1.20 (0.51)	0.0342
CO_2 flux (Mg CO_2-C ha^{-1})			
2013	2.32 (0.72)	2.30 (0.73)	0.949
2014	3.52 (0.54)	2.88 (1.14)	0.0158
Total flux	5.83 (1.10)	5.18 (1.80)	0.0634

† Site names: IR-SB, irrigated shelterbelts; RF-SB, rainfed shelterbelts.
‡ Numbers in parenthesis represent standard deviation

A.6 Discussion

The lack of a significant difference in seasonal trends in soil temperature in the IR-SB and RF-SB sites can be attributed to the effects of shading from the tree canopies. That is, within all three shelterbelts, the canopies prevented penetration of direct solar radiation to the soil underneath the trees—creating an almost uniform soil temperature in the shelterbelts regardless of whether or not they received irrigation water. However, the rainfed sections of the shelterbelts were characterized by severe drying (< 15% VWC) from late-July to mid-October in both 2013 and 2014. Conversely, soil water contents in the IR-SB remained above 15% VWC (with VWCs as great as 26%—equivalent to 50% WFPS) throughout the growing season and early fall. Whereas elevated soil moisture conditions are expected to increase the potential for GHG emissions (i.e., greater N_2O and CO_2 emissions, and lower CH_4 uptake) (Mosier et al., 2006), my results show that in shelterbelt ecosystems under similar soil conditions, the GHG emission response to elevated soil moisture conditions varied with the dominant tree species in the shelterbelt (Fig. A.4 – A.6).

Nitrous oxide is perhaps the most important GHG affected by elevated soil moisture conditions, since even small emissions can result in a large impact in terms of radiative forcing. The range of daily N_2O fluxes measured in this study (-2.3 to 16.5 g N_2O-N ha^{-1} d^{-1}) is similar to that reported in forested soils (-4 to 7 g N_2O-N ha^{-1} d^{-1}) in the Mid-Boreal Upland Ecoregion of Central Saskatchewan (Matson et al., 2009). Whereas elevated soil moisture conditions stimulated emissions of N_2O in the caragana shelterbelt, a similar effect was not observed in the Scots spine shelterbelts—where N_2O fluxes follow a pattern that Brumme et al. (1999) and Yates et al. (2006) refer to as “background” emissions (Fig. A.4). Consequently, the observation that cumulative N2O emissions was greater in the IR-SB compared to the RF-SB sites may have been arrived at due to the dominant N2O response to irrigation in the caragana shelterbelt. The high standard deviation values associated with the cumulative N2O data agrees with this observation. The lack of N_2O response to elevated soil moisture in the Scots pine shelterbelts (Fig A.4b-c) was likely a result of N limitations, which are characteristic of most soils under coniferous tree plantings (Foster and Bhatti, 2006). Indeed, N limitations in coniferous forest soils have been reported to reduce N_2O production (Peichl et al., 2010). As well, when N was added to laboratory incubations of soils collected from coniferous forests, the potential for N_2O emissions increased significantly (Ullah et al., 2008). These studies—together with the available N data from the CSIDC sites, which demonstrated that the available N (NO_3^- + NH_4^+) content of the soils was consistently greater under caragana than under Scots pine (Table A.1),support my supposition that the lack of an N_2O response to irrigation in the Scots pine shelterbelts was most likely due to an N limitation. At the same time, greater N availability in the soils under caragana is likely responsible for the observed N_2O response to elevated soil moisture in the C_N-IR site compared to the C_N-RF site (Table A.1; Fig A.4a). Caragana has been shown to fix about 75–85% of its N from the atmosphere (Moukoumi et al., 2013)—returning about 20–60 kg N ha^{-1} to the soil in the litter (Issah et al., 2014). Thus, inputs to the soil of N derived from N_2-fixation can exceed plant N requirements, resulting in enhanced N_2O emissions.

Aside from a few days in late-April and early-June 2014, soil water content rarely exceeded 60% WFPS (i.e., ~32% VWC)—even during periods of active irrigation (Figs. A.1b– A.3b). This suggests that nitrification was the predominant driver of N_2O production in the shelterbelts. Other studies have reported nitrification as the major driver for N_2O production, especially in dry, well drained soils such as those at the CSIDC site (Rosenkranz et al., 2006; Chapuis-Lardy et al., 2007). Several studies have shown a strong relationship between N_2O emissions and both soil water content and soil temperature (Smith et al., 2003; Ball et al., 2007); however, no such relationships were found either in the IR-SB or RF-SB sites in this study. I suspect that the lack of correlation reflects the low concentrations of available N in the soil, which resulted in very low N_2O emissions, especially in the Scots pine shelterbelts. Similar findings were reported by Peichl et al. (2010).

Daily CH_4 fluxes measured in this study were in the range Matson et al. (2009) reported for forested soils in the Mid-boreal Upland Ecoregion of Central Saskatchewan (i.e., -39 to +25 g CH_4-C ha^{-1} d^{-1}). Across the entire study period, irrigation reduced the capacity for atmospheric CH_4 uptake by soils in the shelterbelts (Table A.4), although there was no clear response to irrigation in the Scots pine shelterbelts (Fig A.5b-c). The correlation analysis supports this observation, indicating decreasing CH_4 uptake (i.e., oxidation) with increasing soil moisture.

Although soil moisture certainly played a large role in moderating CH_4 consumption (Mosier et al., 2006; Liebig et al., 2005), it is possible that CH_4 consumption in soils under the caragana also was impacted by the inhibitory effect of NH_4^+ on CH_4 oxidation (Bronson and Mosier, 1994; Sainju et al., 2012). For example, Suwanwaree and Robertson (2005) reported that CH_4 oxidation in forest and successional soils was significantly suppressed for several weeks after a one-time addition of NH_4NO_3 fertilizer.

Daily CO_2 fluxes measured in this study were similar to those reported by Peichl et al. (2010) for forested soils in southern Ontario (i.e., 2 to 50 kg CO_2-C ha^{-1} d^{-1}). In general, CO_2 fluxes were driven mainly by trends in soil temperature, which may have exerted a controlling effect on both soil microbial respiration and root respiration (Robertson et al., 2000). This was supported by the correlation analysis—both in the present study and the study by Peichl et al. (2010)—which indicated a strong link between the daily CO_2 flux and soil temperature. In addition to this overarching trend, increased soil moisture resulting from above-normal precipitation and the application of irrigation water also appeared to stimulate CO_2 production/emission in 2014. However, irrigation had no significant effect on CO_2 emissions during the much drier 2013 season. Nevertheless, when examined across the entire study period, I found a strong correlation between CO_2 flux and soil water content. This agrees with previous studies that have demonstrated an increase in aerobic soil respiration in response to an increase in soil moisture and a positive linear relationship between soil moisture and CO_2 emissions (Ellert and Janzen, 2008; Jabro et al., 2008; Trost et al., 2013). Despite the lack of response in CO_2 emissions during the period of irrigation water application in 2013, my data indicate that cumulative CO_2 emissions, summed across the two-year sampling seasons, were greater in the IR-SB than in the IR-SB sites. (Table A.4).

A.7 Conclusion

Not surprisingly, GHG dynamics in the shelterbelts were affected by elevated soil moisture conditions resulting from the application of irrigation water. In general, N_2O and CH_4 emissions from shelterbelt soils under Scots pine were relatively unaffected by the changes in soil moisture status that accompanied the application of irrigation water. Moreover, the Scots pine shelterbelts were only a very minor source of N_2O and were a small net sink for atmospheric CH_4. In terms of soil-derived CO_2 emissions, the Scots pine shelterbelts have the potential to emit more CO_2 in response to irrigation, though this effect appeared to be dependent on the total amount of water received as both precipitation and irrigation. That is, a positive response to irrigation appears to have depended on the ability of the trees to access water from outside the canopy and a subsequent increase in root and microbial respiration.

Greenhouse gas dynamics in the caragana shelterbelt were considerably different than those in the Scots pine shelterbelts. For example, N_2O emissions response to irrigation water additions were much greater under caragana than they were under Scots pine Likewise, the CH_4 sink potential of the soils under caragana was greater than that of the soils under Scots pine, though the sink potential of the caragana soils was greatly diminished under irrigated conditions. In both instances, these responses are believed to be related the much greater concentrations of available N and organic carbon in the soils developed under caragana. That is, caragana can fix considerable amounts of atmospheric N_2 in its leaves and this N is then returned to the soil as litterfall, which is then incorporated into the various soil-N pools. This includes the available N pool, which is subject to microbial transformations (nitrification and denitrification) that can yield N_2O. In addition, C inputs from litterfall are much greater for caragana than they are for Scots pine, and these inputs provide a fairly readily decomposable substrate for the soil microbial community—supporting increased microbial respiration and providing a C and energy source for denitrifying microorganisms. Thus, it is the confluence of N, C and water inputs in the irrigated caragana that drive the changes in GHG dynamics in this system.

My results suggest that shelterbelts comprised of coniferous tree species—which are generally characterized by low available soil N—may benefit from intercropping with N_2-fixing tree species. Such a relationship could increase the overall N use efficiency of the shelterbelt (i.e., with the conifers utilizing some of the N fixed by the caragana) while reducing the amount of soil N available for transformation to N_2O. Further research is needed in this area to improve the effectiveness of shelterbelts as a mechanism to mitigate agricultural GHG emissions.

APPENDIX B. LOCATIONS OF GREENHOUSE GAS SAMPLING SITES IN THE BOREAL AND PRAIRIES ECOZONES OF SASKATCHEWAN, CANADA

B.1 Map of Study Sites

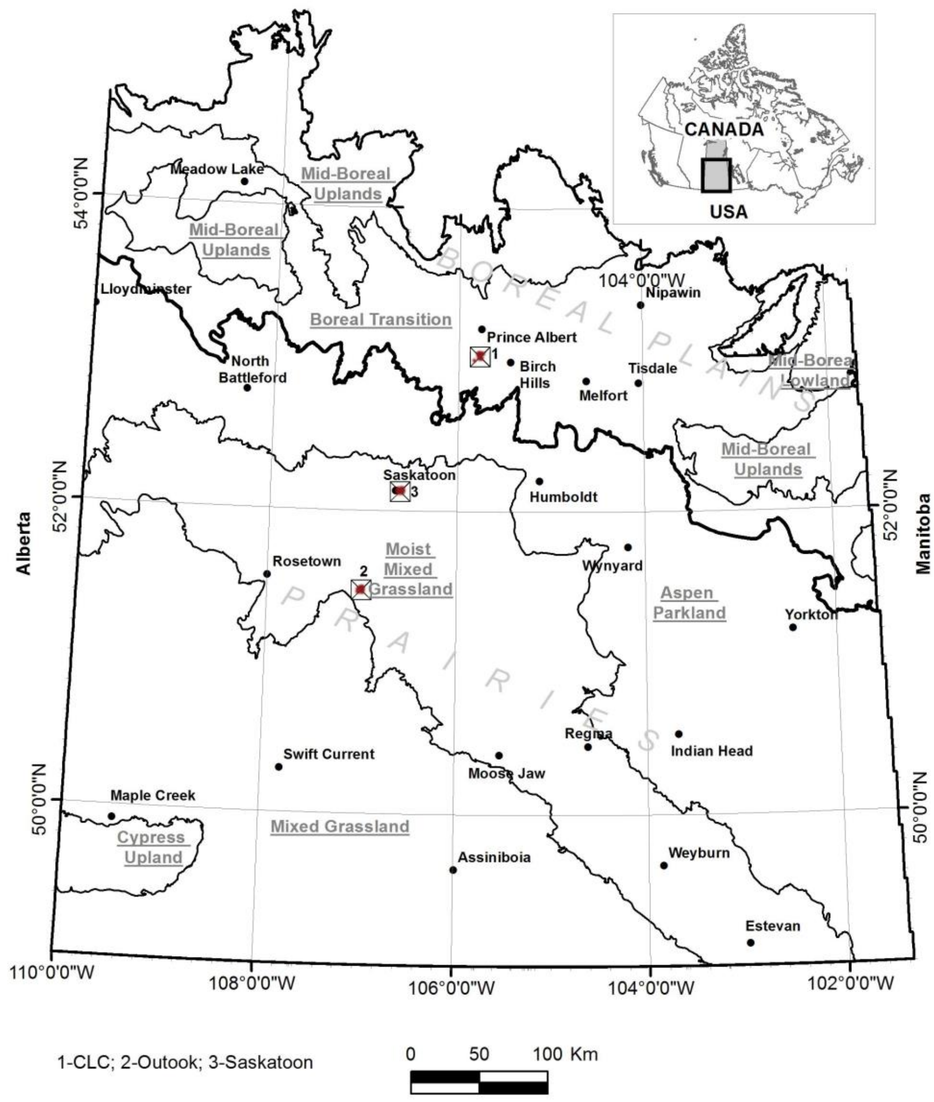

Fig. B.1 Map of study sites at Prince Albert (1), Outlook (2), and Saskatoon (3). Map courtesy of Dr. Beyhan Amichev

APPENDIX C. SITE PHOTOS AND EXPERIMENTAL LAYOUT FOR COMPARING GHG FLUXES IN SHELTERBELTS AND CROPPED FIELDS (CHAPTER 3)

C.1 Experimental Layout (Chapter 3)

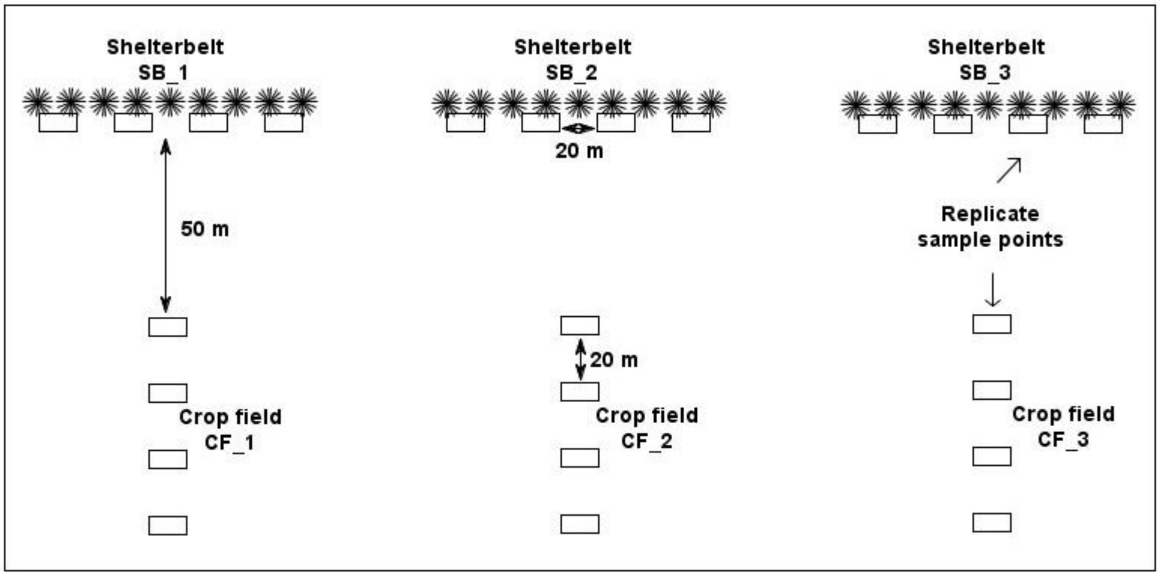

Fig. C.1 Typical experimental layout for comparing soil-derived GHG fluxes between shelterbelts and cropped fields in each study location (Outlook, Saskatoon and Prince Albert).

C.2 Study Site at Outlook, SK

Fig. C.2 Comparison of soil-derived GHG fluxes between shelterbelts and cropped fields at Canada Saskatchewan irrigation Diversification centre, Outlook SK

C.3 Study Site at Saskatoon, SK

Fig. C.3 Comparison of soil-derived GHG fluxes between shelterbelts and cropped fields at the University of Saskatchewan Horticulture Research Facility, Saskatoon SK.

C.4 Study Site at Prince Albert, SK

Fig. C.4 Comparison of soil-derived GHG fluxes between shelterbelts and cropped fields at the Conservation Learning Centre, near Prince Albert SK

APPENDIX D. SITE PHOTOS AND EXPERIMENTAL LAYOUT FOR MONITORING GHG FLUXES AT VARIOUS DISTANCES AWAY FROM A SHELTERBELT (CHAPTER 4)

D.1 Experimental Layout (Chapter 4)

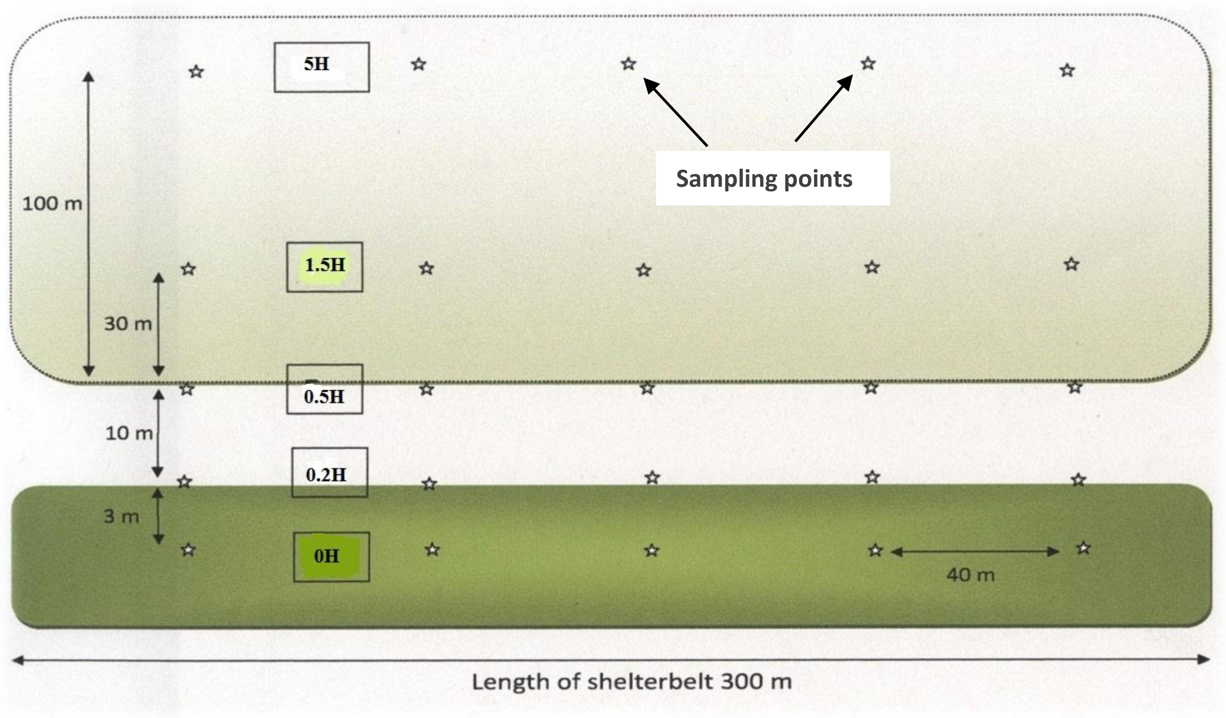

Fig. D.1 experimental layout for monitoring GHG fluxes at various distances away from a shelterbelt at the Conservation Learning Centre, near Prince Albert SK

D.2 Study Site at Prince Albert, SK

Fig. D.2 Site for monitoring GHG fluxes at various distances away from a shelterbelt at the Conservation Learning Centre, near Prince Albert SK