

**Soil Organic Carbon Content and Stability, and Greenhouse Gas Emissions in Three  
Agroforestry Systems in Central Alberta, Canada**

by

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## Abstract

Western Canada's prairie region is extensively cultivated for agricultural production, which is a large source of greenhouse gas (GHG) emissions. Agroforestry systems are common land uses across Canada, which integrate trees into the agricultural landscape and could play a substantial role in sequestering carbon (C) and mitigating increases in atmospheric GHG concentrations. This thesis research quantified soil C storage and stability, and CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions in forest and herbland (areas without trees) components of three agroforestry systems (hedgerow, shelterbelt, and silvopasture) over two growing seasons (May through September in 2013 and 2014). The study evaluated 36 sites (12 hedgerows, 12 shelterbelts, and 12 silvopastures) in central Alberta, Canada, distributed along a soil/climate gradient of increasing moisture availability. Within each agroforestry system, the areas under forest consistently had greater total soil organic C (SOC) and SOC in most soil fractions separated by particle-size (up to 10 cm) and density (up to 30 cm) fractionation than in herbland areas. The C stored in this forest cover is more stable, so less of it is expected to be lost as CO<sub>2</sub> when the climate warms in the future. Soil CO<sub>2</sub> emission and temperature ( $r^2 = 0.53$ ,  $p < 0.01$ ) and CH<sub>4</sub> uptake and soil water content ( $r^2 = 0.38$ ,  $p < 0.01$ ) were significantly related in the studied land uses. Soil temperature and water content are dominant controls on N<sub>2</sub>O emissions, and together explained 71% of the variation in N<sub>2</sub>O emissions. Over the two seasons, forest soils had 3.4% greater CO<sub>2</sub> emission, 36% higher CH<sub>4</sub> uptake, and 66% lower N<sub>2</sub>O emission than adjacent herbland soils. As a result, forested areas had a smaller global warming potential (129) than their herbland counterpart (157 kg CO<sub>2</sub> ha<sup>-1</sup>) based on all three GHGs. Autotrophic respiration contributed more to total respiration in the

forest than in herbland ( $p < 0.01$ ), that, in turn, may be responsible for the high CO<sub>2</sub> emissions in the forest.

The SOC stock in the bulk soil (up to 30 cm) was greater in the silvopasture (201) than in either the hedgerow (178) or shelterbelt system (162 Mg C ha<sup>-1</sup>). Across particle-size fractions, SOC in the more stable fine fraction was in the order of: hedgerow > shelterbelt > silvopasture system. Similarly, the largest pool of SOC in the more stable heavy density fraction of both the 0-10 and 10-30 cm depth classes was in the shelterbelt (33 and 35 Mg ha<sup>-1</sup>, respectively), while the least SOC was in the silvopasture system (26 and 20 Mg ha<sup>-1</sup>, respectively). While ranked emissions of CO<sub>2</sub> were silvopasture > hedgerow > shelterbelt, soils in the silvopasture system had 15% greater CH<sub>4</sub> uptake and 44% lower N<sub>2</sub>O emission rates compared with the other two agroforestry systems. Silvopasture system can provide greater potential to induce soil C sequestration because it leads to a larger reduction in heterotrophic respiration ( $p = 0.03$ ) than the hedgerow and shelterbelt systems. Overall, opportunities appear to exist for enhancing soil C storage and stability, while reducing GHG emissions by retaining and establishing perennial vegetation, both forest and grassland, within agricultural landscapes.

## Preface

This thesis is an original work by Mark Baah-Acheamfour. Chapter 3, 4 & 5 of this thesis has been published as Baah-Acheamfour, M., Carlyle, C.N., Bork, E.W., Chang, S.X. "Trees increase soil carbon and its stability in three agroforestry systems in central Alberta, Canada," *Forest Ecology & Management*, vol. 328, 131-139 (2014), Baah-Acheamfour, M., Chang, S. X., Carlyle, C. N., Bork, E. W. "Carbon pool size and stability are affected by trees and grassland cover types within agroforestry systems of western Canada," *Agriculture, Ecosystem, & Environment*, vol. 213, 105-113 (2015), and Baah-Acheamfour, M., Carlyle, C. N., Lim, S.S., Bork, E. W., Chang, S. X. "Forest and grassland cover types reduce net greenhouse gas emissions from agricultural soils," *Science of the Total Environment*, doi:10.1016/j.scitotenv.2016.07.106 (2016), respectively. I was responsible for the data collection, analysis as well as manuscripts writing. Lim, S.S. assisted with data collection and contributed to manuscript edits. Chang, S.X., Bork, E. W., and Carlyle, C. N. were the supervisory authors and contributed to the editorial corrections of manuscripts.

M.B.-A.

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## List of Symbols and Abbreviations

BD: bulk density

BMPs: best management practices

C: carbon

CEC: cation exchange capacity

CEE: carbon equivalent emission

CH<sub>4</sub>: methane

CO<sub>2</sub>: carbon dioxide

ECD: electron capture detector

FID: flame ionization detector

GHG: greenhouse gas

GWP: global warming potential

N: nitrogen

N<sub>2</sub>O: nitrous oxide

Q<sub>10</sub>: temperature sensitivity

R<sub>A</sub>: autotrophic respiration

R<sub>H</sub>: heterotrophic respiration

R<sub>T</sub>: total respiration

SOC: soil organic carbon

SOM: soil organic matter

TCD: thermal conductivity detector

## **Chapter 1. General introduction**

### **1. Research background**

Canada's agricultural landscape is extensively cultivated for agricultural production, which is a large source of greenhouse gas (GHG) emissions, i.e., carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) (Liebig et al. 2005; Kebreab et al. 2006). In 2012, Canada contributed 552 Mt of CO<sub>2</sub>, 91 Mt of CH<sub>4</sub>, and 48 Mt of N<sub>2</sub>O, for a total of 699 Mt CO<sub>2</sub>-eq (carbon dioxide equivalent) (Environment Canada 2013). In the same year, agriculture-related GHG emissions were 56 Mt CO<sub>2</sub>-eq, which is about 8% of total national emissions (Environment Canada 2013).

Since agricultural land is a major source of GHG (Cole et al. 1993; Janzen et al. 1999), the adoption of best management practices (BMPs) in agriculture has enormous potential to reduce GHG emissions (Paustian et al. 1998; Desjardins et al. 2001; Duncan 2008). Planting of trees in the agricultural landscape such as in the form of agroforestry is one possible mechanism to increase carbon (C) capture in the agricultural landscape (Gordon and Thevathasan 2005; IPCC 2013). Agroforestry is a land management practice where trees and/or shrubs are deliberately combined with crops and/or livestock as a way of increasing diversity and sustainability (Schroeder 1994; Young 1997; Schoeneberger 2009). This approach to farming is reported to be an effective and low-cost method of reducing atmospheric CO<sub>2</sub> concentration (Albrecht and Kandji 2003; Montagnini and Nair 2004; Nair et al. 2009). The attractiveness of using agroforestry to increase C sequestration in agriculture rests on the notion that inclusion of woody plants in croplands and pasturelands would result in greater total above- and belowground C sequestration (Jose 2009). Incorporating trees via agroforestry systems within the agricultural landscape may not only increase total ecosystem C storage, but also reduce emissions of some

GHGs such as CH<sub>4</sub> and N<sub>2</sub>O from soils (Amadi et al. 2016). For example, deep-rooted trees can assimilate residual nitrate from the crop production area, thereby leaving less nitrate available for leaching losses and denitrification, as a result reducing N<sub>2</sub>O emissions (Kang et al. 1999; Dougherty et al. 2009; Evers et al. 2010). However, the amount of nitrate that can be captured by tree roots from beyond the crop rooting-zone depends on agroforestry system type, woody species composition, and tree stocking rate and spatial arrangement, and agronomic practices (Rowe et al. 2005; Rivest et al. 2010; Bergeron et al. 2011).

In Canada, agroforestry systems can take many forms, such as alley cropping, windbreak/shelterbelt, natural hedgerow, silvopasture, and riparian buffer systems (Table 1-1). While these systems are not unique to specific regions in Canada, they may be more widespread in some areas than others, depending on the nature and combination of regional soils and climates, and past land use history. Thevathasan et al. (2012) think shelterbelt areas across Canada (minus that in British Columbia) could be around 0.3 million km. Of this value, approximately 0.2 million km is projected to occur within the three Prairie Provinces of Alberta, Saskatchewan, and Manitoba. A more quantitative estimate of areas under shelterbelt at the province level was undertaken by Amichev et al. (2015) in Saskatchewan; total shelterbelt length (of any species) ranged from 322 to 45,231 km. Across the Canadian prairies, common contemporary agroforestry systems include shelterbelts, natural hedgerows, and silvopastures (Kort et al. 2014). The practice of growing short rotation woody crops such as hybrid poplars (*Populus* spp.) and willows (*Salix* spp.) in agroforestry for bioenergy production is contributing to the sustainability of farms especially in Quebec, Ontario, and the Prairie Region. Southern Ontario remains one of the few regions in Canada where some form of alley cropping systems is practiced (Oelbermann et al. 2004). Though agroforestry is practiced across Canada, it will be

difficult to reliably predict at this time the actual area of the Canadian agricultural landscape that is currently under agroforestry due to disparate data sources.

The system is also one of the few ecosystem services that could be eligible for payment under the various GHG emissions trading programs (Alberta Agriculture and Rural Development (AARD) 2015; Winans et al. 2016). Unfortunately, there are no direct economic incentives for landowners to establish or maintain agroforestry systems in Canada. In Alberta, for example, there are more trees taken out of the agricultural landscape than are planted. Many shelterbelt and hedgerow areas have been replaced with pipelines on the farms. This practice might presumably be influenced by the large monetary incentives that oil companies provide and the termination of federal funding for the Prairie Farm Rehabilitation Administration's (PFRA) shelterbelt program in 2013, through which some forms of agroforestry (e.g., shelterbelts) were promoted (PFRA 2000).

The C-trading market can be an effective mechanism to promote agroforestry by rewarding landowners through payment for C offsets (Crossman et al. 2012; Alam et al. 2014). However, to claim C credit, data collection to demonstrate the capacity of agroforestry systems as a C sink or their ability to reduce GHG emissions is required. Such information would also help develop future C offset policies that tie beneficial management practices to economic gains for landowners. Meanwhile, diverse field data to support claims that agroforestry systems can help increase C sequestration and mitigate climate change are limited across Canada in particular, resulting in different data gaps that need to be filled in order to promote the wider adoption of this unique land use system by landowners.

## 2. Research objectives

The overall goal of this thesis was therefore directed towards generating some of the underlying data needed to understand mechanisms conveying benefits of practicing agroforestry in western Canada, with particular attention to reducing GHGs. The specific objectives were to: (i) examine the impacts of forest and herbland (areas without trees) components of three agroforestry systems (hedgerow, shelterbelt, and silvopasture) on soil C and nitrogen (N) pool size and stability, by comparing C and N distribution in whole soils and three particle-size fractions (fine, medium, and heavy fraction). (ii) expand on the previous study by examining the effects of these land use systems on C and N distribution among three organic matter fractions (light, occluded, and mineral associated heavy fraction), (iii) quantify soil CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions over two growing seasons (May through September in 2013 and 2014) and examine the underlying environmental factors that regulate GHG emissions, and (iv) investigate the fractional contribution of autotrophic and heterotrophic components to total soil respiration in the studied land use systems. Data on autotrophic and heterotrophic respiration could help to infer which of the studied land use systems causes more C accumulation in the soil to be lost. Four experiments were conducted in this thesis research to test the following hypotheses:

- i. Within each agroforestry system, the amount of soil C and stability in areas under herbland will be lower than that under trees because long periods of tilled row-crop cultivation in herblands reduces the physical protection of soil organic matter from decomposition because of the destruction of soil structure.
- ii. Maintaining or establishing perennial vegetation in the form of agroforestry within the agricultural landscape could reduce soil CH<sub>4</sub> and N<sub>2</sub>O emissions due to a modification of

the soil micro-environment, though high root activity may cause CO<sub>2</sub> emissions to be greater in forested areas than herbland.

- iii. Within each agroforestry system, areas under trees could have greater autotrophic respiration but lower heterotrophic respiration compared to herbland. This response is projected to occur because of a greater supply of vegetative inputs (aboveground and roots) during the growing season as well as lower soil temperatures under perennial vegetation, which in turn slow down the microbial processes responsible for soil organic matter mineralization.
- iv. Emissions of CO<sub>2</sub> will be greater in the silvopasture than both hedgerow and shelterbelt systems because of the continuous presence of live vegetation from the combination of perennial forest and grassland cover types which enhances autotrophic respiration.

### **3. Thesis structure**

This thesis contains seven chapters. Chapter 1 (this chapter) provides a general introduction to agroforestry practices in Canada, and introduces the background of this thesis research. Chapter 2 reviews the literature on agroforestry potential to enhance C storage and mitigation of GHG emissions in Canada's agricultural landscapes. Chapter 3 & 4 evaluate the impact of agroforestry systems on soil C and its stability, by comparing C pools in soil particle-size and density fractions, respectively. Chapter 5 deals with the impact of agroforestry systems on CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions, and the relationship of soil GHG emissions with soil temperature or water content. Chapter 6 addresses the contribution of autotrophic and heterotrophic components to total soil respiration as affected by different agroforestry systems. Chapter 7 summarizes key

findings of this thesis research and provide general conclusions. In addition, suggestions for future research needs are given in Chapter 7. Chapters 2 to 6 each forms a manuscript which has either been published (3, 4 & 5), is under review (2) or will be submitted for publication (6).

**Table 1-1.** Major agroforestry practices in Canada and their main ecological functions

Agroforestry system	Brief description	Ecological functions
Alley cropping	Cultivation of crops between rows of trees	Enhance or diversify farm products Reduce water runoff and erosion Protect growing plants Decrease nutrient or chemical loss Increase carbon storage in plant biomass and soils
Windbreak/ Shelterbelt	Strips of planted trees and shrubs along the margins of agricultural lands to reduce wind speed	Enhance crop yield Protect wind-sensitive crops and structures Enhance crop and animal production Control erosion Distribute snowfall Increase carbon storage in plant biomass and soil
Hedgerow	Naturally growing trees, shrubs, and underlying herbaceous vegetation along the margins of agricultural lands	Increase biodiversity Create wildlife habitat Distribute snowfall Increase carbon storage in plant biomass and soil
Silvopasture	Trees growing irregularly or planted in a systematic pattern on rangeland or pastures	Provide diversification of crops in time and space Create wildlife habitat Increase carbon storage in plant biomass and soil
Riparian forest buffer	A combination of trees, shrubs, and grass buffers on the banks of streams, rivers, wetlands and lakes	Provide economic diversification either through plant production or recreational fees Ameliorate non-point source pollution from adjacent land-use Protect watershed and stream banks Enhance aquatic and terrestrial habitats Filtering nutrient runoff

## References

- AARD. 2015. Agricultural carbon offsets: Information for Alberta's offset market [Online]  
Available from <http://worldwidewords.agriculture.alberta.ca/agcarbonoffsets> [accessed  
10 November 2015].
- Alam, M., A. Oliver, A. Paquette, J. Dupras, J. Reveret and C. Messier. 2014. A general  
framework for the quantification and valuation of ecosystem services of tree-based  
intercropping systems. *Agrofor. Syst.* 88: 679–691.
- Albrecht, A. and S.T. Kandji. 2003. Carbon sequestration in tropical agroforestry systems. *Agric.  
Ecosyst. Environ.* 99: 15–27.
- Amadi, C.C., K.C. Van Rees and 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.
- Amichev, B.Y., M.J. Bentham, D. Cerkowniak, J. Kort, S. Kulshreshtha, C.P. Laroque, J.M.  
Piwowar and K.C. Van Rees. 2015. Mapping and quantification of planted tree and shrub  
shelterbelts in Saskatchewan, Canada. *Agrofor. Syst.* 89: 49–65.
- Bergeron, M., S. Lacombe, R.L. Bradley, J. Whalen, A. Cogliastro, M. Jutras and P. Arp. 2011.  
Reduced soil nutrient leaching following the establishment of tree-based intercropping  
systems in eastern Canada. *Agrofor. Syst.* 83: 321–330.
- Cole, C.V., K. Flatch, J. Lee, P. Sauerbeck and B. Stewart. 1993. Agriculture source and sinks of  
carbon. *Water Air Soil Poll.* 70: 111–122.

- Crossman, N.D., B. Burkhard and S. Nedkov. 2012. Quantifying and mapping ecosystem services. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manage.* 8: 1–4.
- Desjardins, R., S. Kulshreshtha, B. Junkins, W. Smith, G. Grant and M. Boehm. 2001. Canadian greenhouse gas mitigation options in agriculture. *Nutr. Cycl. Agroecosyst.* 60: 317–326.
- Dougherty, M.C., T.N Thevathasan, A.M. Gordon, L. Lee and J. Kort. 2009. Nitrate and *Escherichia coli* NAR analysis in tile drain effluent from a mixed tree intercrop and monocrop system. *Agric. Ecosyst. Environ.* 131: 77–84.
- Duncan, K. 2008. Agricultural practices that reduce greenhouse gases (GHGs) and generate co-benefits. *Environ. Toxicol.* 110: 61–69.
- Environment Canada. 2013. Canada's emission trend. [online] <http://www.ec.gc.ca/indicateurs-indicators/default.asp?lang=en&n=CCED3397-1> [accessed 18 May 2014].
- Evers, A.K., A. Bambrick, S. Lacombe, M.C. Dougherty, M. Peichl and Gordon, N.V Thevathasan, J. Whalen and R.L. Bradley. 2010. Potential greenhouse gas mitigation through temperate tree-based intercropping systems. *Open Agric. J.* 4: 49–57.
- Gordon, A.M. and T.N Thevathasan. 2005. How much carbon can be stored in Canadian agroecosystems using a silvopastoral approach? In: Mosquera –Losada MR., McAdams J., Rigueiro-Rodriquez A. (eds.) *Silvopasture and Sustainable Land Manage*, 1<sup>st</sup> edn. CABI Publishing, Wallingford, 210-217 pp.

- Janzen, H.H., R.L. Desjardins, R. Asselin and B. Grace. 1999. The health of our air: towards sustainable agriculture in Canada. Available from Agriculture and Agri-Food Canada. Catalogue No. A53-1981/1998E. 98 pp.
- Jose, S. 2009. Agroforestry for ecosystem services and environmental benefits: an overview. *Agrofor. Syst.* 76: 1–10.
- Kang, B.T., F.E. Caveness, G. Tian and G.O. Kolawole. 1999. Long-term alley cropping with four hedgerow species on an Alfisol in southwestern Nigeria – effect on crop performance, soil chemical properties and nematode population. *Nutr. Cycl. Agroecosyst.* 54: 145–155.
- Kebreab, E., K. Clark, C. Wagner-Riddle and J. France. 2006. Methane and nitrous oxide emissions from Canadian animal agriculture: A review. *Can. J. of Anim. Sci.* 86: 135–158.
- Kort, J., J. Richardson, R. Soolanayakanahally and W. Schroeder. 2014. Innovations in temperate agroforestry: the 13th North American Agroforestry Conference. *Agrofor. Syst.* 88: 563–567.
- Liebig, M., J. Morgan, J. Reeder, B. Ellert, H. Gollany and G. Schuman. 2005. Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada. *Soil Tillage Res.* 83: 25–52.
- Montagnini, F. and P.K.R. Nair. 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. *Agrofor. Syst.* 61-2: 281–295.

- Nair, P.K.R., B.M. Kumar and V.D. Nair. 2009. Agroforestry as a strategy for carbon sequestration. *J. Plant Nutr. Soil Sci.* 172:10–23.
- Oelbermann, M., P.R. Voroney and A.M. Gordon. 2004. Carbon sequestration in tropical and temperate agroforestry systems: a review with examples from Costa Rica and southern Canada. *Agric. Ecosys. Environ.* 104: 359–377.
- Paustian, K., C.V. Cole, D. Sauerbeck and N. Sampson. 1998. CO<sub>2</sub> mitigation by agriculture: An overview. *Clim. Change* 40: 135–162.
- PFRA. 2000. *Prairie agricultural landscapes: a land resource review*. ISBN 0–622–28574–3. Minister of Public Works and Government Services 2000, 179 pp
- Rivest, D., A. Cogliastro, R.L Bradley and A. Olivier. 2010. Intercropping hybrid poplar with soybean increases soil microbial biomass, mineral N supply and tree growth. *Agrofor. Syst.* 80: 33–40.
- Rowe, E C., M. Van Noordwijk, D. Suprayogo and G. Cadisch. 2005. Nitrogen use efficiency of monoculture and hedgerow intercropping in the humid tropics. *Plant and Soil* 268: 61–74.
- Schoeneberger, M.M. 2009. Agroforestry: working trees for sequestering carbon on agricultural lands. *Agrofor. Syst.* 75: 27–37.
- Schroeder, P. 1994. Carbon storage benefits of agroforestry systems. *Agrofor. Syst.* 27: 89–97.
- Sharrow, S.H. and S. Ismail. 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. *Agrofor. Syst.* 60: 123–130.

- Thevathasan, N.V., A.M. Gordon, R. Bradley, et al. 2012. Agroforestry research and development in Canada: The way forward. In: Nair PKR, Garrity DP (eds.), *The Future of Global Land Use*, Dordrecht, The Netherlands, 247–283 pp.
- Young, A. 1997. *Agroforestry for soil management*, 2<sup>nd</sup> ed. CAB International, Wallingford, UK, 330 pp.

## **Chapter 2. Agroforestry enhance C storage and mitigation GHG emissions in Canada's agricultural landscapes\***

### **1. Review of C sequestration and GHG emissions in agroforestry systems**

Best management practices are needed to reduce greenhouse gas (GHG) emissions from agricultural lands, which are a significant source of GHG (Paustian et al. 1998; Kebreab et al. 2006). Agroforestry systems are common features in Canada's agricultural landscape and could play an important role in storing carbon (C), reducing GHG emissions and contributing other ecosystem services (Montagnini and Nair 2004; Jose 2009). Despite mechanisms in some Canadian jurisdictions to reward landowners for practices that reduce GHG, agroforestry does not qualify due in part to a lack of data that would support C offset policies.

In this chapter, I reviewed studies that quantify C stores and/or GHG emissions in agroforestry systems in Canada, with the aims of assessing the benefit of planting or retaining perennial vegetation in agriculture and drawing lessons that can better direct future research in this area. We obtained most original (primary) research published on the subject by searching in CAB Abstracts, Biological Abstracts, Google Scholar, and Web of Science using keywords "agroforestry", "carbon sequestration", "greenhouse gas emissions" "carbon dioxide", "methane", "nitrous oxide", "biomass plantation".

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\* A version of this chapter is under review:

Baah-Acheamfour, M., Chang, S. X., Carlyle, C. N., Bork, E. W. Review - Agroforestry enhance carbon sequestration and mitigate GHG emissions in Canada's agricultural landscape. In review for The Forestry Chronicle Manuscript ID TFC-2016-0007

Often, agroforestry systems are examined by their two components, the treed component, which I will refer to as *forest*, and the *herbland*, which may be an annual crop, a buffer zone or pasture, all of which are dominated by herbaceous plants. To be included for analysis in this review, studies had to report on C stores and/or GHG emissions and contrast between the forest and herbland areas of the system. The practice of planting fast-growing woody species on marginal agricultural land (i.e., biomass plantation) for bioenergy production is an important aspect of agroforestry in Canada that I considered in this review, although this might be referred to as afforestation in other areas. Consequently, data from 28 research papers, including 12 from biomass plantation were extracted and categorized according to the type of agroforestry system studied. To conduct the synthesis, some assumptions were made. For studies that reported soil C concentration without bulk density values, bulk densities and soil C stocks were calculated following Eqn. [1] (Post and Known 2000):

$$BD = \frac{100}{\frac{\%OM}{0.244} + \frac{100 - \%OM}{1.64}} \quad [1]$$

where BD is bulk density ( $\text{g cm}^{-3}$ ). Eqn. [1] assumes that 58% of organic matter (OM) is C (Mann 1986). We considered this approach since the number of studies that report soil C stock in agroforestry systems in Canada is small. Overall, there were insufficient data to allow for a more comprehensive analysis such as a meta-analysis to be performed. This review will cover C stored in vegetation, the size of soil C pools, the potential to reduce  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$  emissions in agroforestry systems in Canada.

## 2. Carbon stored in vegetation in agroforestry systems

Carbon sequestration involves the removal of C from the atmosphere into long-lived global pools (including oceans, vegetation, and soils) through physical or biological processes (Jose 2009). Among the various agroforestry systems in Canada, the most widely studied are the alley cropping systems (Table 2-1). Some studies on this system have been done in eastern Canada, where research has been conducted to understand the role of trees on farmland. Lately, other systems such as hedgerows, shelterbelts and silvopastures have been studied in western Canada (Kort and Turnock 1999; Kort et al. 2008; Baah-Acheamfour et al. 2014, 2015; Banerjee et al. 2015; Amadi et al. 2016).

Early research on alley cropping reported that large quantities of C were stored in the rows of trees when fast-growing tree species such as hybrid poplars (*Populus* spp.) were used. In Guelph, Ontario, for example, Peichl et al. (2006) examined the ecological advantages of integrating trees into farmland by calculating the total tree C stocks of hybrid poplar and Norway spruce (*Picea abies* (L.) Karst) within an alley cropping system. For comparison, sampling was also undertaken in annual cropland of this system planted with barley (*Hordeum vulgare* L.). At 13 years of age, the total mean tree C stock of hybrid poplar (18.7 Mg C ha<sup>-1</sup>) was more than twice that found in spruce (9.2 Mg C ha<sup>-1</sup>). Additionally, the mean C stock in biomass of both hybrid poplar and Norway spruce was 81 and 62% greater, respectively, than that in the annual cropland (3.5 Mg C ha<sup>-1</sup>). Working at the same location, Wotherspoon et al. (2014) quantified C stocks of five tree species commonly used in alley cropping systems with barley or soybean (*Glycine max* (L.) Merr) and found that mean biomass C storage (straw biomass plus grain yield) in areas occupied by the annual crops was 3.3 Mg C ha<sup>-1</sup>, well below the 13, 15, 16, 16, and 27

Mg C ha<sup>-1</sup> found in Norway spruce, black walnut (*Juglans nigra* L.), red oak (*Quercus rubra* L.), eastern white cedar (*Thuja occidentalis* L.), and hybrid poplar, respectively, 25 years after establishment. Hybrid poplar trees continued to store more C than similarly aged spruce 25 years after establishment, thereby demonstrating the high sequestration potential of hybrid poplars. However, hybrid poplars declined in above- and belowground tree C sequestration rates from 1.4 Mg C ha<sup>-1</sup> year<sup>-1</sup> 13 years after establishment (Peichl et al. 2006) to 1.1 Mg C ha<sup>-1</sup> year<sup>-1</sup> 25 years after establishment (Wotherspoon et al. 2014). A key management implication of both studies is that intercropping with fast growing tree species such as hybrid poplar can result in large short-term C sequestration, whereas the planting of slower growing conifers will contribute to long-term C storage.

Borden et al. (2014) used ground penetrating radar (GPR) to estimate C storage in coarse root biomass in a 25-year-old alley cropping system in southern Ontario. The alley cropping systems include five tree species (hybrid poplar, black walnut, red oak, Norway spruce, and eastern white cedar) planted in rows spaced 15 m apart, with maize (*Zea mays* L.), barley, soybean, and winter wheat (*Triticum aestivum*) planted in rotation in the alley. The mean C stock of tree root systems up to 90 cm deep was 2.9 Mg C ha<sup>-1</sup>, with all but two of the tree species (i.e., Norway spruce and eastern white cedar) had their largest root C in the surface 0-40 cm of the soil. In 3-year-old willows of three varieties (*Salix dasyclados*—SV1, *Salix miyabeana*—SX67 and *Salix purpurea*—9882-41) planted between rows of 21-year-old mixed tree species (predominantly black walnut with some red oak, white ash (*Fraxinus Americana* L.) and black locust (*Robinia pseudoacacia* L.) in Guelph, Ontario, Cardinael et al. (2012) also observed higher biomass yield and leaf litter input in the willow-tree alley cropping system (8.8 Mg C ha<sup>-1</sup>) than in adjacent agricultural land (5.8 Mg C ha<sup>-1</sup>).

Fortier et al. (2015) carried out a field experiment to quantify vegetation C (above- and belowground) in herbaceous, hybrid poplar, and woodlot riparian buffer systems across four agricultural sites in southern Quebec. The hybrid poplar buffers were in their 9<sup>th</sup> growing season, whereas woodlots varied in tree species and age. The four woodlot buffers include a: (i) 200-year-old eastern hemlock (*Tsuga canadensis* L.) stand (ii) 73-year-old eastern white cedar stand; (iii) 27-year-old grey birch (*Betula populifolia* Marsh.) stand, and (iv) 54-year-old sugar maple (*Acer saccharum* Marshall) stand. Vegetation C was up to 4, 110, and 160 Mg C ha<sup>-1</sup> in the herbaceous, hybrid poplar, and woodlot buffers, respectively. When total root biomass C (up to 60 cm depth) was estimated, it was found to be greater in the hybrid poplar than both the herbaceous and the 27-year-old grey birch woodlot buffers. Interestingly, hybrid poplar buffer system showed a high biomass yield from year six (8.9 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup>) to year nine (15.1 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup>) than the other buffer systems (Fortier et al. 2010; 2013a). The results from all the above studies suggest the establishment of hybrid poplar, in replacement of herbaceous buffer in agriculture has great potential to increase ecosystem C stock over a short time period. In a related study, Oelbermann et al. (2015) studied the C sequestration benefit of replacing herbaceous buffer with woody species or perennial grasses. After 25 years of replacing an agriculturally degraded riparian zone with native tree species, autumnal litterfall was greater in rehabilitated areas with native species than an undisturbed naturally forested riparian zone (> 100 m) located 320 m upstream from the rehabilitated site. The amount and quality of litter inputs play an important role in determining nutrient cycling and ecosystem soil C. Replacement of herbaceous buffer with woody buffer in the agricultural riparian zone could increase litter amount and lower its quality, which in turn, affect the duration of C sequestration in soils (Oelbermann and Gordon 2000).

In mature shelterbelts in Saskatchewan, the total biomass C storage was 3.8, 7.2, and 15.2 Mg C ha<sup>-1</sup> in shelterbelts made up of coniferous and deciduous trees, and perennial shrubs, respectively (Kort and Turnock 1999). Mean aboveground biomass C storage across shelterbelts was 4.1 Mg C ha<sup>-1</sup> which was more than belowground biomass C storage. While there are a limited number of publications on C storage and GHG mitigation potential of agroforestry systems in Canada, these preliminary results highlight the potential role of shelterbelts in maintaining or increasing C storage in Canadian agroecosystems.

Pinno and Belanger (2008) also examined the long-term effects of growing trees on ecosystem and soil C pools in a pastureland dominated by native grasses (e.g., *Agropyron* spp. and *Bromus* spp.). There was a significant gain in vegetation C after growing white spruce (*Picea glauca* (Moench) Voss) and Siberian larch (*Larix sibirica* Lebed) for 50 years (104 Mg C ha<sup>-1</sup>) than in adjacent herbland (4.5 Mg C ha<sup>-1</sup>). Most of the C gains in the forested areas of the silvopasture came from the aboveground compartment, while much of C allocation (around 78%) in the herbland went towards the development of structural roots (Pinno and Belanger 2008). A southern Ontario study also concluded incorporating trees into managed pastures substantially increased the C sequestration capacity (Gordon and Thevathasan 2005). The biomass C sequestration reported in this study varied from 0.3 Mg C ha<sup>-1</sup>year<sup>-1</sup> in pasturelands to 0.6 Mg C ha<sup>-1</sup>year<sup>-1</sup> in adjacent grazed mixedwood forest. Results also indicated that there was a marked trend towards greater C sequestration with the introduction of fast-growing tree species such as hybrid poplars and willows. Gordon and Thevathasan (2005) surmised that the total GHG emissions of the Canadian agricultural sector could potentially be offset by as little as 6.4 million ha of land managed under a silvopasture system. Currently, there is 20 million ha of rangeland and pastureland grazed by livestock in the Canadian prairie, much of which already

has trees and shrubs. Around 51% of that area is in Alberta, 39% in Saskatchewan, with the remaining 10% in Manitoba (Statistics Canada 2011). Nair et al. (2009) reported that the C sequestration potential of silvopasture systems in temperate North America varied from a low of 1.8 to a high of 3.3 Mg C ha<sup>-1</sup> year<sup>-1</sup>. Using an intermediate level of C sequestration (2.3 Mg C ha<sup>-1</sup> year<sup>-1</sup>), the silvopasture area of 20 million ha in the Canadian prairies alone would represent a C sequestration potential of 2.3 Pg of C for the first 50 years.

Measurements of ecosystem C within silvopastures are influenced by the lack of reliable estimates of the amount of grass and forest understory vegetation consumed by grazing animals. It is believed that a large amount of biomass is removed annually through grazing and that such biomass removals constitute a significant loss of C from silvopasture systems (Follett and Kimble 2000). For example, sheep consumed a total of 22 Mg ha<sup>-1</sup> of forage in silvopasture and deposited 7 Mg ha<sup>-1</sup> of manure in a study conducted in Oregon (Sharrow and Ismail 2004). In Canada, there is an abundance of information on the impact of grazing on ecosystem C in grasslands (e.g., Naeth et al. 1991; Baron et al. 2002), but studies on these responses in silvopasture systems are limited and inconclusive. In the absence of comprehensive studies on animal grazing impacts, we believe that it is reasonable to accept C data from this agroforestry system as only a partial expression of the overall C sequestration potential. Further, methodological difficulties in the estimation of C stock in biomass and the extent of ecosystem C storage are present in all agroforestry systems (Udawatta and Jose 2011). Using the available biomass data from all agroforestry systems that are less than 30 years, I estimated these systems to possess an average biomass C (above- and belowground) storage potential of approximately 28.3 Mg C ha<sup>-1</sup>. Of this, 23.2 Mg C ha<sup>-1</sup> could come from the area occupied by trees and understory vegetation alone, with the balance derived from neighboring herblands (Fig. 2-1). The

estimate of 28.3 Mg C ha<sup>-1</sup> represents almost 45% of the mean vegetation C found in temperate agroforestry systems (Schroeder 1994; Montagnini and Nair 2004; Nair 2011; Udawatta and Jose 2011); the remainder of which is predominantly stored in the soil. Further, biomass C stored in agroforestry systems is subject to a number of variables such as soil, climate, and management practices. Large variability may exist among management practices and sampling methodologies. For example, according to Wotherspoon et al. (2014), the branches from all tree species in the alley cropping system they studied were mostly pruned to a height of 4-5 m from the ground every 5 years. Although the authors did not specify the fate of these crown prunings, in an earlier review of C sequestration in temperate agroforestry systems, Oelbermann et al. (2006) indicated that crown prunings are often not applied to the neighboring herblands of the agroforestry systems due to slow decomposition rate in temperate conditions. Albeit a reduction in aboveground biomass, crown pruning could constitute a significant source of organic matter to the soil ecosystem if not taken offsite but chipped and spread within the tree rows instead.

### **3. Soil C stored in agroforestry systems**

Carbon stored in above- and belowground biomass is susceptible to loss with fire and other surface disturbances. Soil represents a mechanism for short to long-term C storage, and contains more C than all terrestrial vegetation and the atmosphere combined (Paustian et al. 1998; Watson et al. 2002). With an estimated 2400 Pg of C (up to 2 m deep), the soil organic C (SOC) pool is 3.1 times that of the atmospheric pool (770 Pg) and 4 times the size of the vegetation pool (610 Pg) (Lal 2001).

Trees and understory vegetation within agroforestry systems alter the above- and belowground total productivity of agro-ecosystems, modify rooting depth and root distribution, and increase organic matter input to the soil from litterfall, all of which may in turn alter local ecological conditions (Jose 2009). Increases in organic matter promote processes that lead to soil organic C (SOC) accumulation such as humification, soil aggregation, and re-distribution of C into lower soil layers by deep-rooted trees (Nair 2011). Almost all past studies on agroforestry systems across Canada demonstrate the importance of trees in soil C accumulation (Table 2-2); however, estimates of the effects of trees on SOC accumulation in different agroforestry systems varied widely (Appendix 2-1). For example, Peichl et al. (2006) compared SOC in areas occupied by hybrid poplar and Norway spruce trees within an alley cropping system with that in the annual cropland of the system. Carbon stocks in the top 20 cm of soil after 13 years showed that the annual cropland had 13.5 and 1.5 Mg C ha<sup>-1</sup> less than the hybrid poplar and Norway spruce sites, respectively. Working at the same location 21 years after tree establishment, Bambrick et al (2010) further showed that soil (0-20 cm) within the annual cropland continued to have 5.9 Mg C ha<sup>-1</sup> less than soils under the hybrid poplar, with no difference between the latter and areas occupied by spruce. Further, while Wotherspoon et al. (2014) showed that SOC stocks were greater in the topsoil (0-20 cm) of the annual cropland of the system compared to areas occupied by both the hybrid poplar and Norway spruce after 25 years, when C stocks were assessed at 0-40 cm depth, the areas occupied by both the hybrid poplar and Norway spruce within an alley cropping system continued to show greater total SOC relative to the annual cropland. Wotherspoon et al. (2014) attributed this finding to the high SOC found within the 20-40 cm soil layer in the area occupied by trees. These results support the contention that tree-based systems contain more C in deeper soil layers than conventional agricultural systems,

presumably because of high input of organic matter associated with decomposing tree roots (Haile et al. 2008). Carbon stored in deeper soil layers is considered stable due in part to a lack of supply of fresh organic matter which stimulates decomposition (Fontaine et al. 2007). The study by Pinno and Belanger (2008) to explain C levels in the mineral soil at different depths of a 50-year-old silvopasture revealed trees could help store C in deep soil layers. After 50 years of integrating both white spruce and Siberian larch in pastureland, the authors recorded a significant shift from soil C accumulation being driven by root litter quality in the Ah horizons to being driven by root litter quantity in the B horizons in the forest portion of the silvopasture. The root extension study by Borden et al. (2014) in an alley cropping system in eastern Canada focused on 0-90 cm depth and is also useful for understanding the importance of woody species in C storage in deeper soil layers. Key questions include to what degree individual plant species impact C storage in the soil profile, and whether planting of fast-growing species such as hybrid poplars has advantages over slow-growing species, such as Norway spruce? Research on this subject could be useful in the context of selecting woody species of practical significance and developing agroforestry systems aimed at increasing long-term soil C storage. Large amounts of root biomass in the surface soil (0-40 cm) are usually characteristic of fast-growing species used in agroforestry systems. Repeated pruning and wide spacing may promote proliferation of roots near the surface and such a trait is usually undesirable in terms of depositing C deeper into the soil profile. However, a root system that can extend deep into the soil profile would in theory be advantageous in storing C and capturing nutrients that might otherwise be lost beyond the crop rooting zone.

Several studies have also reported no differences in soil C between forest and herbland components of agroforestry. For example, Oelbermann et al. (2006) found no differences in SOC

in the shallow soil layer (0-40 cm) between areas occupied by hybrid poplar trees and adjacent areas without trees. Levels of SOC stored in the area occupied by trees of a 13-year-old alley cropping system in southern Canada were 102 and 125 Mg C ha<sup>-1</sup> for the 0-20 and 0-40 cm depth, respectively, compared to 98 and 120 Mg C ha<sup>-1</sup> year<sup>-1</sup>, respectively, in the annual cropland. Although the alley cropping system studied comprised of 13-year-old hybrid poplar trees, Oelbermann et al. (2006) opined that this timeframe may have been too short to detect any significant differences in SOC. To confirm the above explanation, Oelbermann and Voroney (2011) used the CENTURY soil organic matter model to evaluate how the alley cropping system (at year 13) and adjoining cropped field will affect SOC stocks over a 100 year period. The model predicted SOC in the system would increase steadily whereas that in the cropped area would show a decline. Hybrid Poplar has a relatively short lifespan; the plant may only reach 30 to 50 years of age maximum before senescence. An alley cropping system with hybrid poplar could become a net source of C after this age without any management intervention such as coppicing. In a recent study of a 9-year-old hybrid poplar alley cropping system, Winans et al. (2014) observed no significant differences in SOC stocks (0-5 cm depth) between the area with trees of the system and adjacent annual cropland. However, the latter stored more C than the area with trees within the 0-30 cm soil layer. They concluded that while deep rooting trees allowed more C to be deposited in deeper soil layers, tillage-induced burial of crop residues at the bottom of the plow-pan may also help sequester C below 20 cm. The afore-mentioned results highlight the inconsistency and uncertainty in how trees in agroforestry systems impact SOC in agroecosystems. It is not clear whether these inconsistencies are caused by environmental or management factors, or simply by an artifact of sampling methodology. The effects of sampling

methods on SOC have been demonstrated by Wotherspoon et al. (2015) in an alley cropping system in southern Canada.

#### **4. Potential for reduction of soil CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions**

The importance of planting or retaining woody species in agriculture, to help mitigate GHG emissions, has been demonstrated by several studies in eastern and western Canada. For example, Peichl et al. (2006) reported a higher soil CO<sub>2</sub> emission (5.2 g CO<sub>2</sub>-C m<sup>-2</sup> day<sup>-1</sup>) in areas occupied by a 13-year old hybrid poplar compared to the annual cropland of the system (3.3 g CO<sub>2</sub>-C m<sup>-2</sup> day<sup>-1</sup>). The authors further quantified individual C fluxes of assimilation, soil respiration, and C leaching, as well as C lost to barley harvest, to assess the effectiveness of C sequestration in each system. Thirteen years after establishment, the hybrid poplar showed a net CO<sub>2</sub> sink of 13.2 Mg CO<sub>2</sub>-C ha<sup>-1</sup> year<sup>-1</sup>, whereas the cropland emitted 2.9 Mg CO<sub>2</sub>-C ha<sup>-1</sup> year<sup>-1</sup> into the atmosphere. In a related study, CO<sub>2</sub> emissions were measured in five alley cropping systems (hybrid poplar, black walnut, Norway spruce, and eastern white cedar) and an adjoining soybean cropping system (Wotherspoon et al. 2014). Measurements were done between June and October 2012, and in May 2013 for a 24-h period. Mean CO<sub>2</sub> emissions over the entire sampling period from the hybrid poplar, black walnut, Norway spruce, and white cedar plots were 6.2, 5.9, 5.8, 5.6 g CO<sub>2</sub>-C m<sup>-2</sup> day<sup>-1</sup>, respectively. The mean CO<sub>2</sub> emission from the annual cropland was 4.9 g CO<sub>2</sub>-C m<sup>-2</sup> day<sup>-1</sup>. When comparing CO<sub>2</sub> emission rates at various distances from the tree row to the annual cropland, CO<sub>2</sub> emission was always greatest closest to the tree row at 0 m (7.7 to 5.6 g CO<sub>2</sub>-C m<sup>-2</sup> day<sup>-1</sup>), and lowest at 6 m (3.5 to 4.6 g CO<sub>2</sub>-C m<sup>-2</sup> day<sup>-1</sup>). Wotherspoon et al. (2014) also quantified the net CO<sub>2</sub> flux for all systems taking into account C assimilation by

trees, litterfall C input, root turnover, crop C input and output, and C leaching loss. For hybrid poplar, red oak, black walnut, Norway spruce and eastern white cedar systems the net fluxes were 2.1, 1.6, 0.8, 1.8 and 1.4 Mg CO<sub>2</sub> ha<sup>-1</sup> year<sup>-1</sup>, respectively. Net CO<sub>2</sub> flux for the annual cropland was -1.2 Mg CO<sub>2</sub>-C ha<sup>-1</sup> year<sup>-1</sup>. Greater CO<sub>2</sub> emissions within tree rows in alley cropping systems compared to the adjacent croplands were attributed to high root respiration in the former (Wotherspoon et al. 2014). However, greater C assimilation within the tree rows compensated for high soil respiration, which resulted in net sequestration of C compared to the annual cropland of the system. Agroforestry could be a net source of CO<sub>2</sub> depending on the components (e.g., tree, crops, and livestock) and the type of management practices used. Mechanized activities (fossil fuel combustion), biomass burning, tillage, harvesting, manuring, and livestock production can all increase CO<sub>2</sub> emissions, thus making a given land use a potential C source (Dixon 1995).

Soil mineral nitrogen and N<sub>2</sub>O emissions increased following nitrogen-based fertilizer application in an 8-year-old willow plantation system in southern Canada (Lutes et al. 2016). Up to 22 µg N<sub>2</sub>O-N m<sup>-2</sup> ha<sup>-1</sup> was emitted from fertilized area compared to 26 g N<sub>2</sub>O-N µg m<sup>-2</sup> ha<sup>-1</sup> in the unfertilized area. There has been considerable interest in the use of agroforestry to recover nutrients, such as nitrate, that have a greater likelihood of being leached, thereby contributing to both the reduction of groundwater contamination and N<sub>2</sub>O emissions (e.g., Thevathasan and Gordon 1997; Thevathasan et al. 2004; Evers et al. 2010; Bergeron et al. 2011). Approximately 2.4% of the leached nitrogen is lost as N<sub>2</sub>O (Kaiser et al. 1998). Maintaining trees within agricultural landscape have led to the reduction of nitrogen leaching, which indirectly lowered soil N<sub>2</sub>O emissions by about 0.7 kg N<sub>2</sub>O ha<sup>-1</sup> year<sup>-1</sup> compared to herbland areas (Thevathasan et al. 2004). One explanation is that deep rooting trees can assimilate residual nitrate left from

nitrogen fertilizer applications, thereby leaving less available for denitrification and subsequently reducing N<sub>2</sub>O emissions (Thevathasan et al. 2004; Dougherty et al. 2009; Evers et al. 2010; Bergeron et al. 2011).

Maintaining trees in agricultural landscapes may not just minimize nutrient losses but also reduce the need for application of nitrogen-based fertilizers to soils. Reduction in nitrogen-based fertilizer application is considered an important approach to mitigating agricultural N<sub>2</sub>O emissions (IPCC 2000, 2013). Based on a modelled nitrogen-cycling study conducted with data from a fast-growing hybrid poplar-based alley cropping system, Thevathasan et al. (2004) concluded that around 20 kg ha<sup>-1</sup> of nitrogen-based fertilizer could be saved as a result of additional nitrogen input from nitrogen-fixing trees. Two between row-spacings (12.5 and 15 m) and two within row-spacings (3 m or 6 m) were used in conjunction with all possible combination of agricultural crops (soybean or barley) and biological nitrogen-fixing trees, including sea buckthorn (*Hippophae rhamnoides* L.) and gray alder (*Alnus incana* (L.) Moench).

The presence of woody legumes impacts the overall nitrogen balance of agroforestry systems in which they are included by (i) increasing nitrogen input from nitrogen-fixing trees, (ii) enhancing the availability of nitrogen resulting from the production and decomposition of tree biomass, and (iii) greater uptake and utilization of nutrients from deeper layers of soils by deep-rooting trees (Isaac et al. 2014; Issah et al. 2014; Munroe and Isaac 2014). Biologically fixed nitrogen can be of great benefit if used on marginal (i.e., low nitrogen availability) lands. They can also reduce requirements for nitrogen fertilization for various agricultural crops, thereby reducing net fossil fuel needs and the C cost of manufacturing, transporting, and applying nitrogen fertilizers (Hutchinson et al. 2007). On the other hand, widespread use of “biological fertilizer” systems such as the inclusion of woody legumes in agroforestry can result

in substantial release of nitrogen into the atmosphere (Albrecht and Kandji 2003). Most nitrogen losses from these soils are thought to occur in the form of volatilization of ammonia and leaching of nitrate, which usually amounts to between 10 and 30% of the fixed nitrogen, respectively (Mosier et al. 1998; Galloway et al. 2003). Enhancing temporal synchrony between crop-nitrogen demand and soil-nitrogen input is a key strategy for reducing nitrogen losses via these processes (Galloway et al. 2003). These examples suggest that agroforestry systems can provide significant GHG mitigation options, but also require proper management to influence the amount of GHG emissions.

Given the high global warming potentials of CH<sub>4</sub> (25 times over a 100-year time horizon) and N<sub>2</sub>O (300 times) (Forster *et al.* 2007), reductions in the emissions of this gas from soils managed through agroforestry will also increase the ability of this land use to mitigate potential climate change. Studies on CH<sub>4</sub> emissions in Canadian agroforestry systems are small, but data collected to date suggests the impact of woody species on CH<sub>4</sub> emissions in agriculture may be significant. Working in the Prairies Ecozones of Saskatchewan, Amadi et al. (2016) reported greater CH<sub>4</sub> uptake (0.66 vs. 0.19 kg CH<sub>4</sub>-C ha<sup>-1</sup> y<sup>-1</sup>) and lower N<sub>2</sub>O emissions (0.65 vs. 2.5 kg N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup>) in shelterbelt areas than in adjacent herbland, respectively. Overall seasonal exchange of these gasses was reduced by 0.55 Mg CO<sub>2</sub>-eq ha<sup>-1</sup> y<sup>-1</sup> in shelterbelt than in herbland.

In fact, agroforestry could become a net source or sink of GHGs depending on the components (e.g., tree, crops, and livestock) and the type of management practices utilized. Silvopasture system, for example, can become a ‘hot spot’ for CH<sub>4</sub> emissions through indirect emissions from livestock (Beauchemin and McGinn 2006; Beauchemin et al. 2010). However, if woody species can modify the soil micro-environment and enhance CH<sub>4</sub> oxidation (Amadi et al. 2016), then the high CH<sub>4</sub> uptake will offset emissions from livestock activity, and the net effect

depends on the magnitude of uptake in the soil. Management strategies to mitigate GHG emissions in Canada's agriculture could also be directed towards replacing fossil fuel with energy produced from woody biomass. Biofuels use can offset C by preventing emissions from the fossil fuels which would otherwise have been used. Currently, woody biomass supply about 6% of Canada's energy consumption, an increase of about 3% in the 1970s (Natural Resource Canada 2014). Expanding the bioenergy sector could help Canada achieve an economy-wide emission target of 17% below 2005 level by 2020 (Liu et al. 2014). The question of how the bioenergy sector can be expanded without negatively impacting other land uses, such as food production remains an important source of discussion (Berndes et al. 2003). Agroforestry systems like alley cropping, shelterbelts, hedgerows, and riparian buffer can be ideal systems for bioenergy production because of their potential to grow woody crops alongside annual crops in the same land management unit. The planting of fast-growing trees at high densities on marginal agricultural land (i.e., biomass plantation) is an important aspect of agroforestry in Canada for bioenergy production, although this practice might be called afforestation in other areas (e.g., Pinno and Belanger 2008). Hybrid poplar is one of the most commonly used varieties for establishing biomass plantation in the Prairie Region of Canada (Yemshanov and McKenney 2008; Amichev et al. 2010). Some studies on the potential of willow as a bioenergy source have been conducted in eastern (Labrecque and Teodorescu 2005; Clinch et al. 2009; Cardinael et al. 2012; Nissim et al. 2013) and also in Prairie Region (Amichev et al. 2015). The use of marginal agricultural land for biomass production could be a wise policy since more areas can potentially be brought under biofuel production without adverse effects on other land uses such as food production.

## 5. Challenges, constraints and future research needs

Agroforestry systems across Canada could play an important role in sequestering C (Thevathasan et al. 2012) and reducing GHG emissions from agriculture (Amadi et al. 2016). These systems also hold great potential in providing a number of ecosystem services including reduction in nutrient leaching (Dougherty et al. 2009; Bergeron et al. 2011). However implementation of agroforestry practices faces significant challenges because of changing technical and socioeconomic circumstances (Alam et al. 2014). In this section, we discuss some of the challenges and constraints associated with practicing agroforestry in the Canadian prairies, likely with application in other regions of Canada.

Within the Prairie Region, shelterbelts, hedgerows and silvopasture systems often disappear as these systems naturally senesce, and in some situations, are being actively removed by farmers (Kulshreshtha 2010). Personal discussion with landowners suggests a contributing factor is that the practice does not provide direct benefits (i.e., benefits that can be measured through immediate market transactions) to landowners. This observation is corroborated by a study in southern Ontario in which the loss of income due to a reduction of the insurable crop acreage was cited as the reason for producers to not adopt agroforestry (Simpson 1999). Trees can house pests, particularly insects, which damage crops and increase production costs. Trees can also compete with neighboring crops or understory forage species in silvopastoral systems; research from the Parkland and Boreal regions in Alberta indicates that high densities of trembling aspen can reduce available sunlight and decrease soil and/or air temperatures, all of which can lead to reduced forage production (e.g., Powell and Bork 2006, 2007; LaRade and Bork 2011). However, these same studies found that moderate tree densities favor understory

growth, likely as a result of increased relative humidity, associated water conservation and increased moisture use efficiency, and reduced frost events in the understory (Powell and Bork 2007). The high capital cost for implementing agroforestry systems is another limitation for promoting agroforestry practices. Significant opportunity costs exist for foregoing crop production on land to be planted to trees, particularly in the face of fluctuating markets for crop commodities over time. Initial tree establishment costs combined with the loss of revenue due to removing cropland from production often deter Canadian farmers from adopting agroforestry (Valdivia et al. 2012). Overall, long-term adoption of agroforestry systems has been lower than expected in the region (Matthews et al. 1993; Kulshreshtha 2010). In fact, it is improbable for farmers to adopt new or maintain existing agroforestry systems unless it is proven to be more profitable.

Debt and budget deficits at the federal and provincial levels can also limit the role of government in providing incentives for agroforestry based farming systems. Farmers in the three Prairie Provinces of Alberta, Saskatchewan, and Manitoba no longer have access to support programs such as the Prairie Shelterbelt Program, which was terminated by the federal government as a cost-cutting measure. The Prairie Shelterbelt Program was established under the Prairie Farm Rehabilitation Act to deal with prolonged drought in the early 1930's in Canada (PFRA 2000), and enabled conservation measures and diversification of Prairie agriculture through the provision of free seedlings of trees and shrubs to Prairie farmers (Mackay et al. 1999). The absence of federal support and loss of the Prairie Shelterbelt Program in 2013 ended a long-term program that subsidized shelterbelt establishment in Canada. Furthermore, the structure of agriculture in Canada has changed over the last two decades toward larger farms to gain the economy of scale, many of which are incorporated and making use of rented lands

(Statistics Canada 2011). Corporate farms may be less interested in planting of trees that do not have immediate market value, whereas farming on rented land may also present obstacles to long-term conservation practices with trees (Thevathasan et al. 2012).

It is clear that a mix of innovative policies and associated market incentives would have a large impact on the adoption of agroforestry systems in Canada. The C-trading market is a mechanism for payment to landowners for C sequestration as an environmental service (Aldy and Stavins 2012; Crossman et al. 2012; Alam et al. 2014). However, society's willingness to pay for C sequestration under this market has yet to be clearly established. Alberta is the first province in Canada to actively implement a C market through a mandatory cap-and-trade system (AARD 2015). In general terms, this market requires high emitters to purchase C credits (initially valued at \$15 tonne<sup>-1</sup> CO<sub>2</sub>-eq) from external offset agencies such as landowners to achieve the projected GHG mitigation targets. Recent plans to increase this to \$30 tonne<sup>-1</sup> should further increase demand for C offsets. In Alberta, trading of C sequestered through tree retention or establishment in agricultural lands is still under revision (AARD 2015) and will rely on clear data highlighting the benefits of agroforestry systems in reducing atmospheric CO<sub>2</sub>. The Quebec's Cap-and-Trade System for Greenhouse Gas Emissions Allowances is another jurisdiction in Canada with a C pricing policies (Schott 2013). This C-trading market was established in 2013 but was linked with that of California in 2014 to create the largest C market in North America Winans et al. (2016) examined the C sequestration and C payments under the Quebec's Cap-and-Trade System for 10-year-old alley cropping systems with hybrid poplar. Results suggest landowners would be eligible for C payments up to \$2,758 CAD ha<sup>-1</sup> after 10 years of alley cropping practices.

Integrating agroforestry within the C market could be a wise policy; such mechanism would greatly benefit landowners in the form of revenue diversification and may be a reasonable incentive for agroforestry establishment (Freedman and Keith 1995; Kulshreshtha and Kort 2009). However, for C trading in agroforestry to be successful, a number of landscape-scale questions need to be addressed. These include understanding how much land is currently in agroforestry and how much additional land could be converted to agroforestry. Without economic incentives for practicing agroforestry, existing agroforestry systems, particularly hedgerows and silvopastures, may be lost in the future. Information is needed on the life spans of these systems and on the impact on GHG emissions and climate change if those systems are lost. Another interesting question is that the land area 'suitable' for trees may change with climate change itself. If existing forests disappear due to climate change, the question of who is responsible for the cost of losing them (including increases in CO<sub>2</sub> emissions) may also need to be addressed.

## **6. Conclusions**

Agroforestry systems are sustainable land use systems that maintain and often increase ecosystem C storage and contribute to GHG mitigation in agricultural landscapes. Areas occupied by woody species have the potential to store more C in above- and belowground components and in soils as compared to equivalent land areas without trees or shrubs. Further, well-managed agroforestry system can be net CO<sub>2</sub> sinks. However, the extent of the C stock and emissions via soil respiration can vary, depending on the type of agroforestry system employed, plant species composition, and system age and management intensity.

Much more broadly-based research would be needed to demonstrate that agroforestry systems have the capacity to serve as effective sinks for C and reduce GHG emissions. We emphasize that sampling designs for estimating C stocks in agroforestry should be standardized in order to improve data reliability and interpretation. We also recommend that future research should pay much attention to C stored in deeper soil layers (especially in forest component) in the estimation of SOC stocks in order to minimize potential bias brought on by incomplete soil sampling. More efforts are needed to quantify the fluxes of other trace gases such as CH<sub>4</sub> and N<sub>2</sub>O to determine net benefits of agroforestry on the atmosphere. Finally, integrating agroforestry with GHG emissions trading program would provide financial incentive to farmers and facilitate the adoption of these systems amidst agricultural producers in Canada.

**Table 2-1.** Vegetation C stocks (above- and belowground) of various agroforestry systems in different locations in Canada<sup>a</sup>.

Agroforestry system	Location	Age (year)	Cover type	Species type	Vegetation C (Mg ha <sup>-1</sup> )		References
					Aboveground	Belowground	
Alley cropping	Ontario	13	Forest	Hybrid Poplar	14.95	3.77	Peichl <i>et al.</i> (2006)
				Norway Spruce	7.15	2.08	
			Herbland	Barley monocrop	2.08	1.43	
	Ontario	25	Forest	Hybrid Poplar	16.3	10.8	Wotherspoon <i>et al.</i> (2014)
				Norway Spruce	9.0	4.0	
				Red oak	9.5	6.5	
				Black walnut	9.0	6.0	
Herbland			White cedar	11.3	4.8		
			Soybean monocrop	1.8	1.5		
Shelterbelt	Saskatchewan	33-54	Forest	Deciduous trees	4.6	2.6	Kort and Turnock (1999)
				Coniferous trees	2.7	1.1	
				Shrubs	7.8	7.4	
Silvopasture	Ontario	13	Forest	Hybrid Poplar forest	8.5	5.6	Gordon and Thevathasan (2005)
				Norway spruce forest	3.3	2.2	
			Herbland	Ryegrass	5.5	2.5	
Riparian buffer	Ontario	95	Forest	Riparian forest	63.7	20.0	Hazlett <i>et al.</i> (2005)
				Upslope forest	62.7	18.1	
	Quebec	9	Forest	Hybrid Poplar forest		9.1	Fortier <i>et al.</i> (2013)
				Woodland buffer		20.5	
Herbland			Herbaceous buffer		1.9		

<sup>a</sup> Whenever absent, root biomass C was calculated assuming it to be for hardwoods, conifers, and shrubs species to be 40%, 30%, and 50% of the aboveground C content, respectively (Freedman and Keith 1995).

**Table 2-2.** Mean soil C storage along soil depths in forest and herbland cover types of various agroforestry systems in Canada.<sup>a</sup>

Soil depth (cm)	Cover type	Soil C (Mg ha <sup>-1</sup> )	Reference(s)
0–5	Forest	11.1	Gordon and Thevathasan (2005)
	Herbland	9.8	Winans <i>et al.</i> (2014)
0–10	Forest	44.8	Baah-Acheamfour <i>et al.</i> (2014,2015)
	Herbland	37.8	
0–20	Forest	64.3	Oelbermann <i>et al.</i> (2006)
	Herbland	65.3	Peichl <i>et al.</i> (2006)
			Fortier <i>et al.</i> (2013) Winans <i>et al.</i> (2014)
0–30	Forest	88.90	Bamberick <i>et al.</i> (2010)
	Herbland	80.70	Baah-Acheamfour <i>et al.</i> (2014) Winans <i>et al.</i> (2014)
0–40	Forest	83.00	Oelbermann <i>et al.</i> (2006)
	Herbland	78.70	Cardinael <i>et al.</i> (2012) Wotherspoon <i>et al.</i> (2014)

<sup>a</sup> The analysis used published datasets from independent studies in agroforestry systems that sampled mineral soils from the 0-5, 0-10, 0-20, 0-30, or 0-40 cm depth in areas occupied by both woody species and herbaceous crops.

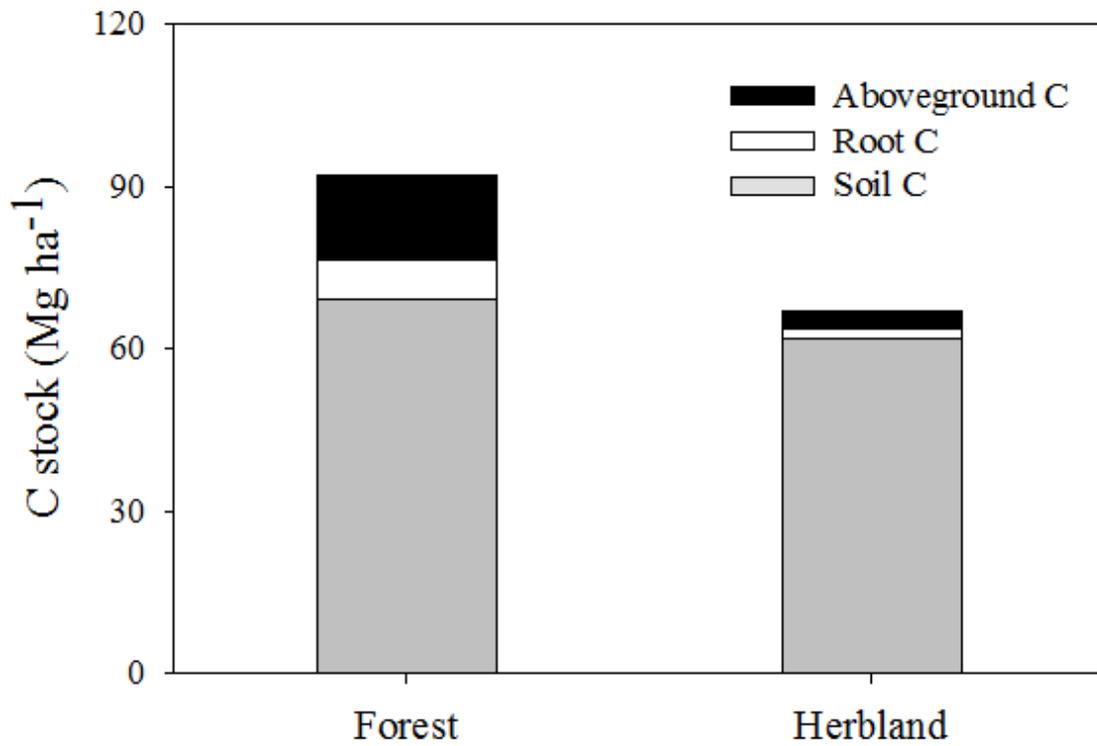


Fig. 2-1. Carbon storage in above- and belowground biomass and soil in areas dominated by woody species (forest) or non-woody vegetation (herbland) of agroforestry systems in Canada.

## References

- AARD. 2015. Agricultural carbon offsets: Information for Alberta's offset market [Online] Available from <http://worldwidewords.agriculture.alberta.ca/agcarbonoffsets> [accessed 10 November 2015].
- Alam, M., A. Oliver, A. Paquette, J. Dupras, J. Reveret and C. Messier. 2014. A general framework for the quantification and valuation of ecosystem services of tree-based intercropping systems. *Agrofor. Syst.* 88: 679–691.
- Albrecht, A. and S.T. Kandji. 2003. Carbon sequestration in tropical agroforestry systems. *Agric. Ecosyst. Environ.* 99: 15–27.
- Aldy, J.E. and R.N. Stavins. 2012. The promise and problems of pricing carbon: Theory and experience. *J. Environ. Dev.* 21: 152–180.
- Amadi, C.C., K.C. Van Rees and 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.
- Amichev, B.Y., M.J. Bentham, D. Cerkowniak, J. Kort, S. Kulshreshtha, C.P. Laroque, J.M. Piwowar and K.C. Van Rees. 2015. Mapping and quantification of planted tree and shrub shelterbelts in Saskatchewan, Canada. *Agrofor. Syst.* 89: 49–65.
- Amichev, B.Y., M. Johnston and K.C. Van Rees. 2010. Hybrid poplar growth in bioenergy production systems: Biomass prediction with a simple process-based model (3PG). *Biomass Bioenerg.* 34: 687–702.

- Baah-Acheamfour, M., C.N. Carlyle, E.W. Bork and S.X. Chang. 2014. Trees increase soil carbon and its stability in three agroforestry systems in central Alberta, Canada. *For. Ecol. Manage.* 328: 131–139.
- Baah-Acheamfour, M., S.X. Scott, C.N. Carlyle, and E W. Bork. 2015. Carbon pool size and stability are affected by trees and grassland cover types within agroforestry systems of western Canada. *Agric. Ecosyst. Environ.* 213: 105–113.
- Bambrick, A.D., J.K. Whalen, R.L. Bradley, A. Cogliastro, A.M. Gordon, A. Olivier and N.V. Thevathasan. 2010. Spatial heterogeneity of soil organic carbon in tree-based intercropping systems in Quebec and Ontario, Canada. *Agrofor. Syst.* 79: 343–353.
- Banerjee, S., M. Baah-Acheamfour, C. N. Carlyle, A. Bissett, A. E. Richardson, T. Siddique, E.W. Bork and S.X. Chang. 2015. Determinants of bacterial communities in Canadian agroforestry systems. *Environ. Microbiol.* Available from <http://dx.doi.org/10.1111/1462-2920.12986> [accessed 22 June 2015)
- Baron, V.S., E. Mapfumo, A.C. Dick, M.A. Naeth, E.K. Okine and D.S. Chanasyk. 2002. Grazing intensity impacts on pasture carbon and nitrogen flow. *J. Range Manage.* 55: 535–541.
- Beauchemin, K.A., H.H. Janzen, S.M. Little, T.A. McAllister and S.M. McGinn. 2010. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. *Agr. Syst.* 103: 371–379.

- Beauchemin, K.A. and S.M. McGinn. 2006. Enteric methane emissions from growing beef cattle as affected by diet and level of intake. *Can. J. Anim. Sci.* 86: 401–408.
- Bergeron, M., S. Lacombe, R.L. Bradley, J. Whalen, A. Cogliastro, M. Jutras and P. Arp. 2011. Reduced soil nutrient leaching following the establishment of tree-based intercropping systems in eastern Canada. *Agrofor. Syst.* 83: 321–330.
- Berndes, G., M. Hoogwijk and R. van den Broek. 2003. The contribution of biomass in the future global energy supply: A review of 17 studies. *Biomass Bioenerg.* 25: 1–28.
- Borden, K.A., M.E. Isaac, N.V. Thevathasan, A.M. Gordon and S.C. Thomas. 2014. Estimating coarse root biomass with ground penetrating radar in a tree-based intercropping system. *Agrofor. Syst.* 88: 657–669.
- Cardinael, R., N. Thevathasan, A. Gordon, R. Clinch, I. Mohammed and D. Sidders. 2012. Growing woody biomass for bioenergy in a tree-based intercropping system in southern Ontario, Canada. *Agrofor. Syst.* 86: 279–286.
- Clinch, R.L., N.V. Thevathasan, A.M. Gordon, T.A. Volk and D. Sidders. 2009. Biophysical interactions in a short rotation willow intercropping system in southern Ontario, Canada. *Agric. Ecosyst. Environ.* 131: 61–69.
- Crossman, N.D., B. Burkhard and S. Nedkov. 2012. Quantifying and mapping ecosystem services. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manage.* 8: 1–4.
- Dixon, R.K. 1995. Agroforestry systems: sources or sinks of greenhouse gases? *Agrofor. Syst.* 31: 99–116.

- Dougherty, M.C., T.N Thevathasan, A.M. Gordon, L. Lee and J. Kort. 2009. Nitrate and *Escherichia coli* NAR analysis in tile drain effluent from a mixed tree intercrop and monocrop system. *Agric. Ecosyst. Environ.* 131: 77–84.
- Evers, A.K., A. Bambrick, S. Lacombe, M.C. Dougherty, M. Peichl and Gordon, N.V Thevathasan, J. Whalen and R.L. Bradley. 2010. Potential greenhouse gas mitigation through temperate tree-based intercropping systems. *Open Agric. J.* 4: 49–57.
- Follett, R.F. and J.M. Kimble. 2000. The potential of US grazing lands to sequester carbon and mitigate the greenhouse effect. CRC Press.
- Fontaine, S., S. Barot, P. Barré, N. Bdioui, B. Mary and C. Rumpel. 2007. Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature* 450: 277–280.
- Forster, P.M.D., K.P. Shine and N. Stuber. 2007. It is premature to include non-CO<sub>2</sub> effects of aviation in emission trading schemes. *Atmos. Environ.* 41: 3941–3941.
- Fortier, J., D. Gagnon, B. Truax and F. Lambert. 2010. Biomass and volume yield after 6 years in multiclonal hybrid poplar riparian buffer strips. *Biomass Bioenergy* 34: 1028–1040.
- Fortier, J., B. Truax, D. Gagnon and F. Lambert. 2015. Biomass carbon, nitrogen and phosphorus stocks in hybrid poplar buffers, herbaceous buffers and natural woodlots in the riparian zone on agricultural land. *J. Environ. Manage.* 154: 333–345.
- Fortier, J., B. Truax, D. Gagnon and F. Lambert. 2013a. Mature hybrid poplar riparian buffers along farm streams produce high yields in response to soil fertility assessed using three methods. *Sustainability* 5: 1893–1916.

- Fortier, J., B. Truax, D. Gagnon and F. Lambert. 2013b. Root biomass and soil carbon distribution in hybrid poplar riparian buffers, herbaceous riparian buffers and natural riparian woodlots on farmland. *SpringerPlus* 2: 539–539.
- Freedman, B. and T. Keith. 1995. *Planting trees for Carbon Credits*. Publ by Dalhousie University, Halifax, Nova Scotia, 42 pp.
- Galloway, J. N., J. D. Aber, J. W. Erisman, S. P. Seitzinger, R. W. Howarth, E.B. Cowling and B. J. Cosby. 2003. The nitrogen cascade. *BioScience* 53: 341–56.
- Gordon, A.M. and T.N Thevathasan. 2005. How much carbon can be stored in Canadian agroecosystems using a silvopastoral approach? In: Mosquera –Losada MR., McAdams J., Rigueiro-Rodriquez A. (eds.) *Silvopasture and Sustainable Land Manage*, 1<sup>st</sup> edn. CABI Publishing, Wallingford, 210-217 pp.
- Haile, S.G., P.K.R. Nair and V.D. Nair. 2008. Carbon storage of different soil-size fractions in Florida silvopastoral systems. *J. Environ. Qual.* 37: 1789–1797.
- Hutchinson, J.J., C.A. Campbell and R.L. Desjardins. 2007. Some perspectives on carbon sequestration in agriculture. *Agric. For. Meteorol.* 142: 288–302.
- IPCC. 2000. *Land-use, land-use change, and forestry*. A special report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

- IPCC. 2013. Climate change: The physical science basis. Working group I contribution to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, United Kingdom.
- Isaac, M.E., G. Carlsson, C. Ghoulam, M. Makhani, N.V. Thevathasan and A.M. Gordon. 2014. Legume performance and nitrogen acquisition strategies in a tree-based agroecosystem. *Agroecol. Sustain. Food Syst.* 38: 686–703.
- Issah, G., A.A. Kimaro, J. Kort and J.D. Knight. 2014. Quantifying biological nitrogen fixation of agroforestry shrub species using N-15 dilution techniques under greenhouse conditions. *Agrofor. Syst.* 88: 607–617.
- Jose, S. 2009. Agroforestry for ecosystem services and environmental benefits: an overview. *Agrofor. Syst.* 76: 1–10.
- Kaiser, E., K. Kohrs, M. Kücke, E. Schnug, O. Heinemeyer and J. Munch. 1998. Nitrous oxide release from arable soil: Importance of N-fertilization, crops and temporal variation. *Soil Biol. Biochem.* 30: 1553–1563.
- Kebreab, E., K. Clark, C. Wagner-Riddle and J. France. 2006. Methane and nitrous oxide emissions from Canadian animal agriculture: A review. *Can. J. of Anim. Sci.* 86: 135–158.
- Kort, J. and R. Turnock. 1999. Carbon reservoir and biomass in Canadian prairie shelterbelts. *Agrofor. Syst.* 44: 175–186.

- Kort, J., L. White and E. Svendsen. 2008. Silvopasture principles and potential in Saskatchewan. In: Proceedings of the 2008 Soils and Crop Workshop. University of Saskatchewan, Saskatoon, 16 pp. 28–29 Feb. 2008.
- Kulshreshtha, S.N. 2010. Agroforestry development on the Canadian prairies. Nova Science Publishers.
- Labrecque, M. and T.I. Teodorescu. 2005. Field performance and biomass production of 12 willow and poplar clones in short-rotation coppice in southern Quebec (Canada). *Biomass Bioenerg.* 29: 1–9.
- Lal, R. 2001. World cropland soils as a source or sink for atmospheric carbon. *Adv. Agron.* 71: 145–191.
- LaRade, S.E. and E.W. Bork. 2011. Short Communication: Aspen forest overstory relations to understory production. *Can. J. Plant Sci.* 91: 847–851.
- Liu, T., B. McConkey, T. Huffman, S. Smith, B. MacGregor, D. Yemshanov and S. Kulshreshtha. 2014. Potential and impacts of renewable energy production from agricultural biomass in Canada. *Appl. Energy* 130: 222–229.
- Lutes, K., M. Oelbermann, N. Thevathasan and A. Gordon. 2016. Effect of nitrogen fertilizer on greenhouse gas emissions in two willow clones (*salix miyabeana* and *S. dasyclados*) in southern ontario, canada. *Agrofor. Syst.* 1: 1–12.

- Mackay, W.C., T.G. Miller, D.R. Moore and R. Woelcke. 1999. Performance characteristics of aeration devices. Prairie Farm Rehabilitation Administration. Prairie Farm Administration Paper, Peace River, AB, Canada.
- Mann, L.K. 1986. Changes in soil carbon storage after cultivation. *Soil Sci.* 142: 279–288.
- Matthews, S., S.M. Pease, A.M. Gordon and P.A. Williams. 1993. Landowner perceptions and the adoption of agroforestry practices in southern Ontario, Canada. *Agrofor. Syst.* 21: 159–168.
- Montagnini, F. and P.K.R. Nair. 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. *Agrofor. Syst.* 61-2: 281–295.
- Mosier, A., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger, and O Van Cleemput. 1998. Closing the global N<sub>2</sub>O budget: nitrous oxide emissions through the agricultural nitrogen cycle. *Nutr Cycl Agroecosys* 52:225–48.
- Munroe, J.W. and M.E. Isaac. 2014. N-2-fixing trees and the transfer of fixed-N for sustainable agroforestry: a review. *Agron. Sustain. Dev.* 34: 417–427.
- Naeth, M.A., A.W. Bailey, D.J. Pluth, D. Chanasyk and R.T. Hardin. 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland ecosystems of Alberta. *J. Range Manage.* 44: 7-12.
- Nair, P.K.R. 2011. Agroforestry systems and environmental quality: Introduction. *J. Environ. Qual.* 40: 784–790.

- Nair, P.K.R., B.M. Kumar and V.D. Nair. 2009. Agroforestry as a strategy for carbon sequestration. *J. Plant Nutr. Soil Sci.* 172:10–23.
- Natural Resources Canada. 2014. Energy sources and distribution. [online] <http://www.nrcan.gc.ca/energy/alternative-fuels/programs/12358> [accessed 20 April 2016].
- Nissim, W.G., F.E. Pitre, T.I. Teodorescu and M.Labrecque. 2013. Long-term biomass productivity of willow bioenergy plantations maintained in southern Quebec, Canada. *Biomass Bioenerg.* 56: 361–369.
- Oelbermann, M. and A.M. Gordon, 2000. Quantity and quality of autumnal litterfall into a rehabilitated agricultural stream. *J. Environ. Qual.* 29: 603–611.
- Oelbermann, M. and B.A. Raimbault. 2015. Riparian land-use and rehabilitation: Impact on organic matter input and soil respiration. *Environ. Manage.* 55: 496–507.
- Oelbermann, M. and R.P. Voroney. 2011. An evaluation of the century model to predict soil organic carbon: Examples from Costa Rica and Canada. *Agrofor. Syst.* 82: 37–50.
- Oelbermann, M., R.P. Voroney, N.V. Thevathasan, A.M. Gordon, D.C.L Kass and A.M. Schlönvoigt. 2006. Soil carbon dynamics and residue stabilization in a Costa Rican and southern Canadian alley cropping system. *Agrofor. Syst.* 68: 27–36.
- Paustian, K., C.V. Cole, D. Sauerbeck and N. Sampson. 1998. CO<sub>2</sub> mitigation by agriculture: An overview. *Clim. Change* 40: 135–162.

- Peichl, M., N. Thevathasan, A.M. Gordon, J. Huss and R.A. Abohassan. 2006. Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. *Agrofor. Syst.* 66: 243–257.
- PFRA. 2000. *Prairie agricultural landscapes: a land resource review*. ISBN 0–622–28574–3. Minister of Public Works and Government Services 2000, 179 pp
- Pinno, B.D. and N. Belanger. 2008. Ecosystem carbon gains from afforestation in the boreal transition ecozone of Saskatchewan (Canada) are coupled with the devolution of black chernozems. *Agric. Ecosyst. Environ.* 123: 56–62.
- Powell, G.W. and E.W. Bork. 2006. Aspen canopy removal and root trenching effects on understory vegetation. *For. Ecol. Manage.* 230: 79–90.
- Powell, G.W. and E.W. Bork. 2007. Effects of aspen canopy removal and root trenching on understory microenvironment and soil moisture. *Agrofor. Syst.* 70: 113–124.
- Post, W.M. and K.C. Kwon. 2000. Soil carbon sequestration and land use change: processes and potential. *Glob. Chang. Biol.* 6: 317–327.
- Schott, S. 2013. Carbon pricing options for Canada. *Canadian Public Policy* 39(Supplement 2), S109-S124.
- Schroeder, P. 1994. Carbon storage benefits of agroforestry systems. *Agrofor. Syst.* 27: 89–97.
- Sharrow, S.H. and S. Ismail. 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. *Agrofor. Syst.* 60: 123–130.

- Simpson, J.A. 1999. Effects of shade on maize and soybean productivity in a tree-based intercropping system. PhD Thesis, University of Guelph, Guelph.
- Statistics Canada. 2011. Demographic Changes in Canadian Agriculture. Statistics Canada Catalogue no. 96-325-X. Ottawa. Version updated November 2014. [Online] <http://www5.statcan.gc.ca/subject-sujet/result-resultat?pid=920&id=-920&lang=eng&type=OLC&pageNum=1&more=0> [accessed 13 May 2015].
- Thevathasan, N.V. and A.M. Gordon. 1997. Poplar leaf biomass distribution and nitrogen dynamics in a poplar-barley intercropped system in southern Ontario, Canada. *Agrofor. Syst.* 37: 79–90.
- Thevathasan, N.V., A.M. Gordon, R. Bradley, et al. 2012. Agroforestry research and development in Canada: The way forward. In: Nair PKR, Garrity DP (eds.), *The Future of Global Land Use*, Dordrecht, The Netherlands, 247–283 pp.
- Thevathasan, N.V., A.M. Gordon, J.A. Simpson, P.E. Reynolds, G. Price and P. Zhang. 2004. Biophysical and ecological interactions in a temperate tree-based intercropping system. *J. Crop Improv.* 12: 339–363.
- Udawatta, R.P. and S. Jose. 2011. Carbon sequestration potential of agroforestry practices in temperate North America. In: Kumar BM, Nair PKR (eds.) *Carbon Sequestration Potential of Agroforestry Systems: Opportunities and Challenges*. Springer, Dordrecht, 17-42 pp.

- Valdivia, C., C. Barbieri and M.A. Gold. 2012. Between forestry and farming: Policy and environmental implications of the barriers to agroforestry adoption. *Can. J. Agric. Econ.* 60: 155–175.
- Watson, C.A., D. Atkinson, P. Gosling, P.L. Jackson, F.W. Rayns. 2002. Managing soil fertility in organic farming systems. *Soil Use Manage.* 18: 239–247.
- Winans, K., J.K. Whalen, A. Cogliastro, D. Rivest, L. Ribaud. 2014. Soil carbon stocks in two hybrid poplar-hay crop systems in Southern Quebec, Canada. *Forests* 5: 1952–1966.
- Winans, K.S., J.K. Whalen, D. Rivest, A. Cogliastro and R.L. Bradley. 2016. Carbon sequestration and carbon markets for tree-based intercropping systems in southern Quebec, Canada. *Atmosphere* 7: 17.
- Wotherspoon, A., R. Voroney, N. Thevathasan and A. Gordon. 2015. Comparison of three methods for measurement of soil organic carbon. *Commun. Soil Sci. Plant Anal.* 46: 362–374.
- Wotherspoon, A., N.W. Thevathasan, A.M. Gordon, R.P. Voroney. 2014. Carbon sequestration potential of five tree species in a 25-year-old temperate tree-based intercropping system in southern Ontario, Canada. *Agrofor. Syst.* 88: 631–643.
- Yemshanov, D. and D. McKenney. 2008. Fast-growing poplar plantations as a bioenergy supply source for Canada. *Biomass Bioenerg.* 32: 185–197.

## **Chapter 3. Forest and grassland cover types increase soil carbon and its stability in agroforestry systems in western Canada\***

### **1. Introduction**

In response to the increased global demand for food and other agricultural products, more land has been brought under agricultural cultivation. Agricultural practices such as conventional tillage are some of the main contributors to the increased concentrations of carbon dioxide (CO<sub>2</sub>) and other greenhouse gases (GHGs), including methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O), in the atmosphere (Paustian et al. 2000). If GHG concentrations continue to increase, it is likely that global average temperature will rise further (IPCC 2013). Removing atmospheric carbon (C) and storing it within vegetation and soil pools in terrestrial ecosystems is one of the means to mitigate GHG emissions (IPCC 2013).

Agricultural lands could be used to remove large amounts of C from the atmosphere if trees are reintroduced to the system and managed together with crops and/or animals (Nair et al. 2010; Sainju et al. 2012). Agroforestry—an approach to farming where trees and/or shrubs are deliberately combined with crops and/or livestock as a way of increasing diversity and sustainability—is believed to be an effective and low-cost method of sequestering atmospheric C into vegetation and soil pools (Albrecht and Kandji 2003; Montagnini and Nair 2004). Consequently, the importance of agroforestry as a land use system is receiving wider attention

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not only in terms of agricultural sustainability but also as a tool to minimize climate change (Oelbermann et al. 2004; Takimoto et al. 2009). However, a knowledge gap exists on the rate of C inputs from forest and herbland (areas without trees) components and their contribution to stabilizing soil C (Oelbermann et al. 2004).

Historically, shelterbelts are a common agroforestry system practiced in western Canada; trees have been planted in shelterbelts for reducing soil erosion and protecting soils, crops, animals and farm yards from severe wind (Kort and Turnock 1998). Modern agricultural practices, such as conservation tillage, are reducing the need to plant trees and even leading to their removal in some areas. Approximately 30% (20.6 million hectares) of Alberta's total land area is used for crop and livestock production (Alberta Agriculture and Rural Development, 2010), representing a substantial opportunity to sequester C if agroforestry practices are adopted more widely. Conversely, removal of existing agroforestry systems from the agricultural landscape would represent a potential source of C. However, the net effects of land cultivation and forestry practices on soil C and nitrogen (N) pools, as well as the stability of these stores, have not been assessed for agroforestry systems on regional scales in Alberta, although such studies have been undertaken to determine the amount of C held in the prairie shelterbelts of Saskatchewan (Kort and Turnock 1998).

The complexities of land management practices applied to different land uses have a considerable impact on C storage in soils (Arevalo et al. 2009; Howlett et al. 2011). Forests can be significant sinks of atmospheric C compared to many row crop agricultural systems due to high C input associated with decomposing fine roots of trees and annual litterfall (Montagnini and Nair 2004; Ayres et al. 2009). For example, Haile et al. (2008) reported that areas under trees in silvopasture systems of Florida sequestered 33% more soil C than adjacent

open pasture. Most studies have also estimated more than a 35% increase in soil C 50 years after the establishment of agroforestry systems within temperate agricultural landscapes (e.g., Johnson 1992; Howlett et al. 2011). On the other hand, land use conversion from forest vegetation into cultivated agriculture has been reported to decrease total SOC (Takimoto et al. 2009). Cultivation reduces physical protection of soil organic matter from decomposition because of the destruction of soil structure, which enhances microbial decomposition of C (Costa and Foley 2000; Johnson and Curtis 2001). In addition, large amounts of biomass are physically removed from most cultivated fields, either through grain harvesting, straw removal, or both, further reducing the potential for C accumulation (Paustian et al. 2000).

Soil C is composed of fractions (pools) that have different rates of biochemical and microbial degradation, and land use activity can influence the distribution of organic C and N among these SOC pools (Cambardella and Elliott 1994; Teklay and Chang 2008).

Quantification of functional pool sizes may provide insight into the effects of land use on SOC stabilization (d'Annunzio et al. 2008). Physical, chemical, and biological techniques are often used to separate SOC into fractions that differ in their functional roles such as C stabilization (Christensen 2001; Six et al. 2002). Of these methods, physical fractionation of the whole soil into different components based on particle-size has routinely been used because the method is considered to be chemically less destructive and is better related to the function of soil C *in-situ* (Creamer et al. 2011; Arevalo et al. 2012). With particle-size fractionation, soils can be separated into fine- (< 53  $\mu\text{m}$ ), medium- (53–250  $\mu\text{m}$ ), and coarse (250–2000  $\mu\text{m}$ ) fractions. Particle-size fractionation allows us to consider the effect of different land use practices on the process of soil aggregation, including how much C is contained in each fraction and an estimate of the residence times of SOC under different land use systems (Six et al. 2000;

Christensen 2001; Gupta et al. 2009; Howlett et al. 2011). The C inside the coarse fraction (macroaggregates) is considered more labile than that associated with the fine fraction (microaggregates); C in the latter is better protected such that its decomposition rate is slower than those in the coarse fraction (Hassink 1997; Six et al. 2000; John et al. 2005).

In this study, I determined SOC and N in different agroforestry systems within the whole soil and particle-size fractions to: (i) assess the role of trees in facilitating long-term C storage in the surface soil (0–10 cm), and (ii) identify the influence of different agroforestry systems (hedgerow, shelterbelt, and silvopasture) and their component land cover types (forest vs. adjacent herbland) in facilitating SOC and N storage in the whole soil and different soil particle-size fractions in central Alberta, Canada. This study is expected to provide important baseline data on the amount of C and N in the whole soil and particle-size fractions in common agroforestry systems, and highlight opportunities to increase C storage and reduce GHG emissions across the region.

## **2. Materials and methods**

### **2.1. Site description**

This study was conducted across a 270 km long north-south soil/climate gradient (from 54° 43' N to 52° 28' N), spanning the prairie and parkland ecoregions of central Alberta, Canada (Fig. 3-1). Climate normal for the study area, based on data for the last 30 years (1971–2000) from 35 Environment Canada climate stations, show that the northern part of the study area experienced 115 to 125 frost-free days, while the southern sites experienced 125 to 145 frost-

free days. Mean annual air temperatures in the north and south were 1.9 and 3.3 °C respectively. Annual precipitation varies from 463 mm in the north to 497 mm in the south (Environment Canada, 2012). The area is characterized by three soil zones: Dark Gray Chernozemic and Gray Luvisolic soils (based on the Canadian system of soil classification) were predominant in the north, with the south dominated by Black Chernozemic soils (Soil Classification Working Group 1998).

This study used sites that represent three common agroforestry systems in the region: hedgerow, shelterbelt and silvopasture systems. Both the hedgerow and shelterbelt systems are boundary type agroforestry practices. In both systems, strips of permanent vegetation (3-5 m wide) consisting of trees, shrubs and grasses are planted or managed around edges of annual croplands. The hedgerow forests were 40- to 100-yr-old broad-leaved deciduous stands dominated by trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), saskatoon serviceberry (*Amelanchier alnifolia*), and chokecherry (*Prunus virginiana*), whereas shelterbelt forests were made up of 20- to 50-yr-old coniferous and deciduous trees dominated by white spruce (*Picea glauca*) with interspersed caragana (*Caragana arborescens*), willow (*Salix acutifolia*), and box elder (*Acer negundo*). Large areas were also devoted to extensive annual crop production in both systems. Most of the landowners practiced minimum tillage, applied fertilizers that include N (up to 120 kg ha<sup>-1</sup> annually), and grow barley (*Hordeum vulgare*), wheat (*Triticum aestivum*), rapeseed (*Brassica napus*) and pea (*Pisum sativum*) in rotation. The silvopasture was established by deliberately grazing existing understory vegetation in native aspen forests to provide alternative forage for livestock, particularly late in the growing season and during droughty periods. The trees in this system also provided shelter for the livestock grazing in this system. Both the grazed aspen forest and open pasture cover

types support livestock grazing in either rotational or season-long grazing systems. In this system, the size of the forests can vary depending on the amount of space available. Species composition and ages of the grazed aspen forests were similar to the hedgerow forest. Open pastures contain a mix of grasses and forbs, including N-fixing legumes.

## **2.2. Sampling design**

Because of the spatial arrangement of the components within the three agroforestry systems, the experiment was based on a split-plot design. The three agroforestry systems (hedgerow, shelterbelt, and silvopasture) were the main plots, with the two land cover types (forest and herbland) as the subplots. We used 35 agroforestry sites (12 hedgerows, 11 shelterbelts and 12 silvopastures) along the soil-climate gradient. As much as possible, sites were randomly selected to cover the entire study area. One transect was established within each pair of forest and herbland subplots. Depending on the site condition, the length of the transect varied from 30-50 m long. The transect in the area under trees was established in the center of the forested sub-plot, while that in the herbland sub-plot was located at least one tree height (~ 30 m) from the nearest tree to reduce the immediate influence of trees on adjacent herbland fields. As much as possible, we established the paired forest and herbland transects on the same ecosite with similar landform, elevation, drainage and slope. Soil samples were collected between September and October in 2012 after crop harvest and after the ranchers had removed their livestock to alternative pastures. Ten soil cores (3.2 cm diameter, 10 cm deep) were collected at three- to four-meter intervals along the transects. Cores from each transect were combined to

form a composite sample, placed in plastic bags, and kept cool ( $\sim 4\text{ }^{\circ}\text{C}$ ) until they were processed in the laboratory.

### **2.3. Soil physical and chemical characterization**

Soil samples were air-dried at room temperature ( $\sim 20\text{--}25\text{ }^{\circ}\text{C}$ ) and passed through a 2 mm sieve (#10 U.S. Standard Testing Sieve). Soil pH was measured in a 1:2 (*w:v*) mix of soil to 0.01 M  $\text{CaCl}_2$  solution (Kalra and Maynard 1991) with a digital pH meter (Model PHH-200, Omega Eng. Inc., Stamford CT). Ammonium ( $\text{NH}_4^+$ ) and nitrate ( $\text{NO}_3^-$ ) concentrations were determined by an autoanalyzer after extraction using a 2 M KCl solution at a ratio of 1:5 (soil:KCl; *w:v*). To estimate soil water holding capacity (WHC), a known mass of oven-dried soil ( $105\text{ }^{\circ}\text{C}$  for 48 h) was placed in a porous funnel and soaked with water to saturation. The sample was then placed in a humid enclosure and allowed to drain freely under gravity for 24 h, and reweighed.

### **2.4. SOC fractionation**

In the laboratory, a subsample of each sample was oven-dried at  $50\text{ }^{\circ}\text{C}$  for 2 days and passed through a 2 mm sieve. Next, 100 g of the sieved soil was weighed into 500 mL containers to which 150 mL of distilled water was added. The soil sample was slaked on a flatbed shaker for 30 min before ultrasonically dispersing with a Fisher Sonic Dismembrator (Model 300, Fisher Scientific, Pittsburgh PA) at 360W for two minutes. The sample was then poured into a 250  $\mu\text{m}$  sieve. Samples were submerged in water and sieved manually by moving the sieve up and

down about five cm for 50 times in two minutes (Haile et al. 2008). The fraction remaining on top of the 250  $\mu\text{m}$  sieve was collected into a preweighed aluminum pan. The remaining solution was poured through a 53  $\mu\text{m}$  sieve, and the sieving procedure described above was repeated. This procedure yielded three soil fractions (< 53  $\mu\text{m}$ , 53–250  $\mu\text{m}$ , 250–2000  $\mu\text{m}$ ) that were dried in a forced-air oven at 60 °C overnight and weighed (Howlett et al. 2011). Whole and fractionated soils were ground to a fine powder using a ball mill (Mixer Mill MM200, Thomas Scientific, Swedesboro NJ) for 30 seconds, then analyzed for total organic C and total organic N by dry combustion using an elemental analyser (NA-1500 series, Carlo Erba, Milan, Italy).

## 2.5. Statistical analysis

Data on soil physical and chemical characteristics, as well as organic C and N in both the whole soil and particle-size fractions were analyzed using the Statistical Analysis Software (SAS v. 9.3, SAS Institute Inc. 2013). The ANOVA assumption of normality was assessed with a Shapiro-Wilk test using Proc UNIVARIATE (data not shown). All data conformed to a normal distribution except for the soil particle-size distribution (Shapiro-Wilk test  $P < 0.05$ ). A log-transformation was applied for soil particle-size distribution to meet the assumption of normality. Data for the measured parameters were then analyzed according to the following linear model using the Proc MIXED procedure in SAS:

$$Y_{ijk} = \mu + A_i + (A\gamma)_{ik} + L_j + (AL)_{ij} + \varepsilon_{ijk} \ ;$$

where  $Y_{ijk}$  is a dependent variable (SOC, N or particle-size distribution),  $\mu$  is the overall mean,  $A_i$  and  $L_j$  are the effects of the  $i$ th and  $j$ th AF system and land-use, respectively,  $(A\gamma)_{ik}$  and  $\varepsilon_{ijk}$  are the random variable error within the experiment. Mean separation was conducted using the Fisher's protected least significant difference (LSD) test. Given that the study was conducted across a 270 km long stretch of land that had substantial variation in soil properties and vegetation composition, the risk of a type II error in the analysis was considered to be high, even though the sample size was relatively large. Consequently, a  $P$  value of 0.10 was used to assess significance to reduce the type II error.

### 3. Results

#### 3.1. Soil properties

Interaction effects of agroforestry system by land cover type were not significant for pH,  $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , and WHC (data not shown). Soil pH and  $\text{NH}_4^+$  concentration were significantly affected by agroforestry system and land cover type (Table 1). Soil pH in the hedgerow was significantly lower than that in the shelterbelt system, while there was no significant difference between the silvopasture and the other agroforestry systems. Herbland soils were more acidic compared with forest soils. The concentration of  $\text{NH}_4^+$  in the shelterbelt did not differ significantly from that in the hedgerow, but was significantly lower than that in silvopasture system. Between land cover types, concentrations of  $\text{NH}_4^+$  were 57% greater in the soil of forests than in the adjacent herbland lands. One feature of soil mineral N in all areas was the dominance of  $\text{NO}_3^-$ ;  $\text{NO}_3^-$  levels were always 2 to 5 times greater than that of  $\text{NH}_4^+$  (Table 3-

1). However, there was no significant difference in  $\text{NO}_3^-$  concentrations for any of the land-use treatments studied. Soil WHC only significantly differed between land cover types; WHC was greater in forests than in neighboring herblands.

### **3.2. SOC and N in whole soil**

Interaction effects of agroforestry system by land cover type were not significant for SOC, organic N, and C/N ratios, while SOC, N and C/N ratios in the whole soil varied significantly among agroforestry systems and land cover type (Table 3-2). Mean SOC concentration in the silvopasture was similar to that in the hedgerow, but was greater than that in the shelterbelt system ( $P = 0.01$ ) (Table 3-3). Between land cover types, SOC concentration was 31% greater ( $P = 0.02$ ) in forest soils compared with their paired herbland counterpart. Organic N in whole soils did not vary significantly among agroforestry systems. However, organic N concentration in the forest soils was significantly greater than that in the neighboring herbland soils. Soil C/N ratios were within a narrow range (12–15) and were significantly different among agroforestry systems (Table 3-2), except between the hedgerow and shelterbelt system (data not shown).

### **3.3. SOC and N in fractionated soil**

Mean weight of particles in the fine and coarse fractions only significantly differed among the agroforestry systems (Table 3-2). There was no significant difference in weight of soil particles in the medium fraction for any of the land-use treatments studied. In all areas, the fine fraction was most abundant among all particle-size fractions (Table 3-4). Across the agroforestry

systems, the fine, medium, and coarse soil fractions accounted for an average of 53, 27 and 20%, respectively, of total soil weight. In contrast, the coarse soil fraction was at least 62% greater in the silvopasture than in the hedgerow or shelterbelt system (Table 3-4).

The distribution of C across soil fractions within the hedgerow and shelterbelt systems revealed that most of the SOC was usually held in the fine fraction, and to a lesser extent, in the coarse fraction (Table 3-3). In contrast, SOC was almost equally distributed across soil fractions within the silvopasture system. The C in the fine fraction was similar between the hedgerow and silvopasture, but was significantly greater in the hedgerow and silvopasture than in the shelterbelt system. In the coarse fraction, SOC in the silvopasture was significantly greater than that in the hedgerow and shelterbelt systems. Overall, SOC in the fine fraction was significantly greater under forest soils than under their paired herbland (Table 3-3). No significant difference in SOC within the coarse fraction was noted between the two land cover types.

The distribution pattern of organic N among soil fractions in all systems was similar to that of C, with the majority found in the fine fraction ( $3 \text{ g N kg}^{-1}$  soil), followed by the medium fraction ( $1.4 \text{ g N kg}^{-1}$  soil), and then the coarse fraction ( $1.2 \text{ g N kg}^{-1}$  soil) (Table 3-3). Organic N concentrations in the three soil fractions were significantly different among the agroforestry systems, except for the fine fraction. Nitrogen in the medium fraction was significantly greater in the forested land cover than in the herbland, otherwise land cover type did not affect N distribution in particle-size fractions (Table 3-3).

Interaction effects of agroforestry system by land cover type were significant for SOC and organic N only in the medium fraction (Table 3-2). Within the medium fraction, SOC concentration was similar among the forest in the hedgerow system and grazed aspen forest

and open pasture, both in the silvopasture system (24 g SOC kg<sup>-1</sup> soil, on average).

Accordingly, SOC in the medium fraction for these three land uses was significantly greater than that of both annual croplands tested (i.e., those paired with hedgerows and shelterbelts), as well as the shelterbelts themselves (Fig. 3-2). Interaction effects of agroforestry system by land cover type were significant for soil C/N ratios in all three particle-size fractions (Table 3-2).

Soil C/N ratios decreased with decreasing size of the particle-size fraction in all areas; greatest in the coarse, intermediate in the medium, and lowest in the fine fraction (Fig. 3-3).

#### **4. Discussion**

Lower soil pH in the hedgerow than in the shelterbelt system suggests that the tree species in the former may have acidified the soil to a greater extent by producing more organic acids during litter decomposition (Binkley and Richter 1987; Amonette et al. 2004). In fact, species that characterized the hedgerow systems were mostly deciduous, whereas the shelterbelts were mostly made up of conifers. Common expectations included greater acidification of soils under conifers than under deciduous (Mead and Comforth 1995). However, the young age of the studied shelterbelts, coupled with the time lags required for conifer tree litter to break down, could also have slowed acidification. The lower soil pH in the herblands than the forests is inconsistent with previous results in the study area (Arevalo et al. 2009). Application of ammonium based fertilizers to annual crops is common and may contribute to soil acidity when the existing soil minerals and organic matter fail to provide sufficient buffering that keeps the released N in the NH<sub>4</sub><sup>+</sup> form (Nzila et al. 2002).

The greater NH<sub>4</sub><sup>+</sup> concentrations in the silvopasture than in the other agroforestry

systems could be related to the greater abundance of N-fixing plant species common in open pastures in this region (Government of Alberta 2013). For example, white clover (*Trifolium repens*) is a common species in open pastures in the study area and is capable of fixing up to 400 kg ha<sup>-1</sup> of N annually (Frame 2005). Although inputs from excreta (urine and feces) of grazing animals can contribute substantial NH<sub>4</sub><sup>+</sup> to the soil, this response may be short lived since N usually cycles very quickly. Moreover, the high NH<sub>4</sub><sup>+</sup> concentration found in silvopasture was unlikely to be associated solely from animal excreta because few pastures had animals in them at the time of soil sampling.

Organic N input from trees and understory vegetation, as well as greater N loss from open agricultural land areas in the form of NO<sub>3</sub><sup>-</sup> leaching and/or denitrification, could have led to the lower soil NH<sub>4</sub><sup>+</sup> concentration in herblands than in forested lands (Nzila et al. 2002). This result could also have been caused by the removal of crop and forage during harvest and grazing, respectively. The high NO<sub>3</sub><sup>-</sup> concentrations observed are attributed to high immobilization and nitrification rates at the end of the growing season, during which soil sampling occurred (Nason et al. 1988).

The combination of the two silvopasture land uses—grazed aspen forest and neighboring open pasture—may have helped to improve C inputs to the whole soil and led to greater C storage relative to the shelterbelt system (Sharrow and Ismail 2004). The grazed aspen forests had tree and understory vegetation that was more diverse than the shelterbelt forest, which in contrast, represented a relatively simple forest with limited structural and compositional diversity (Kort and Turnock 1998). Open pastures occupied by perennial grass species can also contribute substantially to overall SOC in a silvopasture system because of their high organic matter production capacity, especially in the top 15 cm depth, and their

structural stability (Elliott 1986; Haile et al. 2008). Grazing by livestock has also been reported to produce positive effects on soil C as a result of changes in plant productivity (Schnabel et al. 2001), enhanced physical breakdown and soil incorporation of plant litter (Reeder and Schuman 2002), changes in soil physical properties, and increased allocation of assimilated C to roots (Heitschmidt and Stuth 1991; Gulluscio et al. 2009). Although grazing has been viewed as having a positive impact on the amount of SOC, these results are far from consistent, and many studies have found C losses due to intensive grazing (Potter et al. 2001).

As for the shelterbelt forest, pronounced shading from the overstory canopy may have inhibited the germination and growth of understory vegetation, and in the process markedly reduced C input (Post and Kwon 2000; Paul et al. 2002). In addition, shelterbelt litter was comprised mainly of lignified spruce needles. Thus, litter and fine root turnover in shelterbelt systems may occur much more slowly in comparison to other, less-lignified plant biomass that dominates the hedgerow and silvopasture systems (Binkley and Richter 1987). This in turn may contribute substantially to increasing below- and aboveground C stores in shelterbelts (Thevathasan and Gordon 1997; Nair et al. 1998). However, the lack of mixing of surface litter material with mineral soil due to a slowly decomposing duff layer within shelterbelts may also slow rates of organic C input to the soil (Binkley and Richter 1987). As this organic layer was not included in the calculation of SOC, but would contribute to total ecosystem C, the SOC profiles for each system should be interpreted in this context. Soils in both the hedgerow and silvopasture systems did not differ significantly in terms of total organic C and N concentrations. The study sites for these two systems were mature broad-leaved deciduous forests with abundant aboveground litter input. The moderate to high plant diversity found in both hedgerow and pasture, particularly in the forested components, is also likely to lead to

high inputs of fast decomposing fine roots and leaf litter into the soil (Howlett et al., 2011).

SOC and N concentrations in the whole soil were markedly greater in forests than in neighboring herblands. Carbon inputs from both above- and belowground litterfall from trees, shrubs and herbaceous understory species, may have contributed to greater C accumulation in forest soils (Campton and Boone 2000). While most of these C sources can be lost through decomposition, some residual C may become incorporated into the soil, humified, and finally ended up in long-term soil C pools (Howlett et al. 2011). Over time, the C pool within forest soils can become quite large. There is similar evidence of greater SOC in areas where trees have been integrated into agricultural lands compared to adjacent fields without trees (Jobbagy and Jackson 2000; Haile et al. 2008; Takimoto et al. 2009). The overall lower amounts of organic C and N in herbland soils could also have been caused by more intensive management practices and high decomposition rates. Land management practices such as the removal of plant biomass and tillage can decrease C input from litterfall and root exudates (Li et al. 2014). Tillage reduces the physical protection of soil organic matter from decomposition because of the destruction of soil structure, which enhances microbial decomposition of labile C (Costa and Foley 2000; Johnson and Curtis 2001). In addition, a large amount of biomass is physically removed annually from most agricultural fields, either through grain harvest, straw removal, or both (Paustian et al. 2000), further reducing the potential for organic matter accumulation.

The finding that the SOC and organic N concentrations in the fine particle-size fraction were larger than those in the medium and coarse fractions across study sites is significant, as the C in the fine fraction is usually more stable (Haile et al. 2008) with turnover times ranging from decades to centuries (Yamashita et al. 2006). Greater stability ensures longer-term

sequestration within the ecosystem, and provides enhanced offset potential of atmospheric CO<sub>2</sub> (Haile et al. 2008). Different factors may contribute to the high amount of C and N in the fine fraction in all land uses; such factors include low decomposition rates of silt-and-clay associated organic matter (Hassink 1997), the transfer of stabilized decomposition products from other fractions to the fine fraction, and the accumulation of more stable organic matter (Christensen 2001). This could also be due to the high soil silt plus clay content (48.4%) and the effect of greater organic matter input to the topsoil. Arevalo et al. (2009) also observed high SOC concentrations in the fine particle-size fraction (> 53 µm) on sites unrelated to the present study, within a chronosequence of five different land-uses (agriculture, 2-year-old hybrid poplar, 9-year-old hybrid poplar, grassland, and native forest vegetation) in north central Alberta, Canada.

Both the hedgerow and silvopasture systems contained more C in the fine fraction per kg of the whole soil than the shelterbelt system; that was expected because of differences in age, plant species composition and the resulting diversity (e.g., deciduous broad-lived forests vs. the evergreen coniferous forests). We suspect that we did not observe high C in the fine fraction of the shelterbelt system because 30 years (on average) of shelterbelt establishment might be too short a period for significant transfer of lignified plant biomass to the fine fraction.

While both the forest and neighboring herbland contained substantial amounts of C in the fine fraction, the forest areas had greater C stores in this fraction. In all areas, almost 46% of the C in the whole soil of the forested land cover was found in the fine fraction. Numerous studies (e.g., Nzila et al. 2002; Bronick and Lal 2005; Arevalo et al. 2009) have indicated that forests contained more soil C in the fine fraction than their herbland counterparts. Although a

slower SOC incorporation rate in the fine particle-size fraction is expected within the forest where the ecosystem is less disturbed (absence of tillage) and where soil aggregation is expected to be greater compared to the other land uses (Arevalo et al. 2009; Gupta et al. 2009), a slow macroaggregate turnover in the forest soil will allow for the formation of fine occluded organic matter that gradually becomes encrusted within the silt, clay and microbial products, eventually forming more stable microaggregates within macroaggregates (Six et al. 2000). On the other hand, physical damage to aggregates due to tillage could lead to rapid transfer of organic materials from the medium and coarse to the fine soil fraction in agricultural lands (Arevalo et al. 2009; Gupta et al. 2009). However, the net effects of tillage on the C pool in the fine fraction may vary depending on the intensity, frequency, and timing of disturbance. With continued tillage, organic material accumulated in the fine fraction may slowly decompose causing a net loss of C from the soil over time (Tiessen and Stewart 1983; Bronick and Lal 2005).

SOC and N in the coarse fraction encompass an active C pool that responds quickly in the short-term to changes in soil management, land use, and vegetation regimes (Haile et al. 2008). The SOC and N concentrations in the coarse fraction were quantitatively greater in the silvopasture than in the other agroforestry systems. I suspect this result primarily reflects the more frequent occurrence of silvopasture on lower quality unproductive lands, which often includes a coarse soil texture considered less suited for cropping. C/N ratios decreased with decreasing particle-size fractions, consistent with other studies indicating high C/N ratios occur in the coarse fraction (Christensen 2001; Arevalo et al. 2009), as the coarse fraction contains more organic materials that are undecomposed or partially decomposed (Sollins et al. 1996; Amonette and Russell 2004). Overall, C/N ratios were greater in the forest soils than in their

adjacent agricultural fields irrespective of the soil fraction, which is not surprising given that litter from trees and shrubs, tends to contain more lignin and other recalcitrant compounds low in N (Sollins et al. 1996).

## **5. Conclusions**

Integrating trees and crops into agroforestry systems within the agricultural landscape have the potential to store more C than monocultural agricultural land-use. Most SOC and N were found in the fine fraction, with the least found in the coarse fraction, regardless of the agroforestry system studied. As the C in the fine fraction is more stable than that in the coarse fraction, and the amount of C in the fine fraction per kg of whole soil was markedly greater in both hedgerow and silvopasture than in the shelterbelt system, retaining and promoting hedgerows and silvopastures will better promote long-term storage of C in the soil as compared with the shelterbelt system. On the other hand, the forested component of the agroforestry systems is particularly important for C storage given that they had greater SOC and N concentrations than the adjacent herblands, including harboring larger pools of SOC in the more stable fine fraction. The fact that such high amounts of C can be stored by integrating trees and crops through agroforestry systems is important in the context of developing management strategies aimed at long-term mitigation of atmospheric CO<sub>2</sub>.

**Table 3-1.** Chemical and physical properties (means with standard errors in parentheses) in the 0–10 cm soil layer as affected by agroforestry systems and land-uses (forest and agriculture) in central Alberta, Canada. The interaction term between agroforestry system and land-use was not significant for any of the parameters listed in this table, therefore, only the main treatment effects are listed.

Cover type	pH	NH <sub>4</sub> <sup>+</sup> (mg kg <sup>-1</sup> ) <sup>†</sup>	NO <sub>3</sub> <sup>-</sup> (mg kg <sup>-1</sup> )	WHC (%)
<b>Agroforestry system</b>				
Hedgerow	5.41 (0.12) <sup>b</sup>	2.39 (0.46) <sup>ab</sup>	4.82 (1.03) <sup>a</sup>	84.16 (8.86) <sup>a</sup>
Shelterbelt	5.90 (0.13) <sup>a</sup>	1.50 (0.48) <sup>b</sup>	5.80 (1.09) <sup>a</sup>	72.69 (10.07) <sup>a</sup>
Silvopasture	5.75 (0.12) <sup>ab</sup>	3.03 (0.48) <sup>a</sup>	5.14 (1.09) <sup>a</sup>	96.49 (9.26) <sup>a</sup>
LSD <sub>0.10</sub>	4.06	2.51	0.22 <sup>†</sup>	1.52
Prob > F	0.03	0.05	0.80	0.24
<b>Land-use</b>				
Forest	5.81 (0.12) <sup>a</sup>	3.22 (0.38) <sup>a</sup>	4.60 (0.86) <sup>a</sup>	93.79 (7.60) <sup>a</sup>
Agriculture	5.56 (0.10) <sup>b</sup>	1.40 (0.39) <sup>b</sup>	5.90 (0.88) <sup>a</sup>	75.11 (7.81) <sup>b</sup>
LSD <sub>0.10</sub>	3.64	10.77	1.11	2.92
Prob > F	0.04	0.00	0.30	0.09

<sup>†</sup>NH<sub>4</sub><sup>+</sup>, ammonium; NO<sub>3</sub><sup>-</sup>, nitrate; WHC, water holding capacity.

Within a column and cover type, means with the same superscript letter(s) are not significantly different at  $p < 0.10$  according to the Fisher's protected multiple comparison test (LSD).

**Table 3-2.** Analysis of variance (*F* and *P* values) of the effect of agroforestry system, land-use (forest and agriculture) within agroforest, and their interactions, on soil particle-size fractions, organic C and N concentrations, and C/N ratios of the 0–10 cm soil layer in central Alberta, Canada.

Description	Agroforestry System			Land-use			Agroforestry system x Land-use		
	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>
<b>Soil particle-size fraction<sup>§</sup></b>									
< 53 µm	2	<b>8.97</b>	<b>&lt;0.00</b>	1	0.18	0.67	2	1.89	0.16
53–250 µm	2	1.79	0.18	1	0.04	0.84	2	1.24	0.30
250–2000 µm	2	<b>36.24</b>	<b>&lt;0.00</b>	1	0.52	0.47	2	1.32	0.28
<b>SOC (g kg<sup>-1</sup>)</b>									
< 53 µm	2	<b>1.96</b>	<b>0.03</b>	1	<b>4.62</b>	<b>0.01</b>	2	0.17	0.68
53–250 µm	2	<b>4.47</b>	<b>0.01</b>	1	<b>10.47</b>	<b>0.00</b>	2	<b>4.40</b>	<b>0.02</b>
250–2000 µm	2	<b>3.66</b>	<b>0.03</b>	1	<b>3.97</b>	<b>0.05</b>	2	0.37	0.69
Whole soil	2	<b>3.85</b>	<b>0.03</b>	1	<b>5.68</b>	<b>0.02</b>	2	0.90	0.41
<b>N (g kg<sup>-1</sup>)</b>									
< 53 µm	2	0.46	0.63	1	0.27	0.60	2	1.18	0.32
53–250 µm	2	<b>3.10</b>	<b>0.05</b>	1	<b>4.51</b>	<b>0.04</b>	2	<b>6.80</b>	<b>0.00</b>
250–2000 µm	2	<b>3.88</b>	<b>0.03</b>	1	2.16	0.15	2	1.57	0.22
Whole soil	2	2.31	0.11	1	<b>3.57</b>	<b>0.06</b>	2	0.90	0.41
<b>C/N</b>									
< 53 µm	2	1.45	0.25	1	0.23	0.63	2	<b>2.83</b>	<b>0.07</b>
53–250 µm	2	<b>3.87</b>	<b>0.03</b>	1	<b>10.9</b>	<b>0.00</b>	2	<b>4.43</b>	<b>0.02</b>
250–2000 µm	2	<b>3.54</b>	<b>0.04</b>	1	<b>11.95</b>	<b>0.00</b>	2	<b>2.89</b>	<b>0.07</b>
Whole soil	2	<b>3.39</b>	<b>0.04</b>	1	0.20	0.66	2	0.42	0.66

<sup>§</sup>Soil particle-size fraction: fine fraction (< 53 µm), medium fraction (53–250 µm), coarse fraction

(250–2000 µm); *p* values with less than 10% significance and corresponding *F* values are in bold.

**Table 3-3.** Soil organic C and N concentrations (means with standard errors in parentheses) of whole soil and various particle-size fractions in the 0–10 cm soil layer as affected by agroforestry system and land-uses (forest and agriculture) in central Alberta, Canada

Size fraction	Cover type	SOC (g kg <sup>-1</sup> )	N (g kg <sup>-1</sup> )
<b>Fine fraction (&lt; 53 µm)</b>	<b>Agroforestry system</b>		
	Hedgerow	34.3 (2.7) <sup>a</sup>	2.9 (0.2) <sup>a</sup>
	Shelterbelt	28.8 (3.0) <sup>b</sup>	2.7 (0.2) <sup>a</sup>
	Silvopasture	29.3 (2.8) <sup>a</sup>	3.0 (0.2) <sup>a</sup>
	<b>Land-use</b>		
	Forest	35.1 (2.2) <sup>a</sup>	2.9 (0.1) <sup>a</sup>
	Agriculture	30.1 (1.2) <sup>b</sup>	2.7 (0.1) <sup>a</sup>
<b>Medium fraction (53–250 µm)</b>	<b>Agroforestry system</b>		
	Hedgerow	18.3 (2.3) <sup>ab</sup>	1.3 (0.2) <sup>ab</sup>
	Shelterbelt	13.2 (2.4) <sup>b</sup>	1.1 (0.1) <sup>b</sup>
	Silvopasture	23.3 (2.3) <sup>a</sup>	1.7 (0.1) <sup>a</sup>
	<b>Land-use</b>		
	Forest	22.7 (1.9) <sup>a</sup>	1.6 (0.1) <sup>a</sup>
	Agriculture	13.8 (1.9) <sup>b</sup>	1.1 (0.1) <sup>b</sup>
<b>Coarse fraction (250–2000 µm)</b>	<b>Agroforestry system</b>		
	Hedgerow	13.9 (5.8) <sup>b</sup>	1.0 (0.4) <sup>b</sup>
	Shelterbelt	8.0 (6.1) <sup>b</sup>	0.6 (0.3) <sup>b</sup>
	Silvopasture	29.8 (5.8) <sup>a</sup>	1.8 (0.3) <sup>a</sup>
	<b>Land-use</b>		
	Forest	24.0 (4.9) <sup>a</sup>	1.4 (0.3) <sup>a</sup>
	Agriculture	10.4 (4.8) <sup>b</sup>	0.8 (0.2) <sup>a</sup>
<b>Whole soil</b>	<b>Agroforestry system</b>		
	Hedgerow	65.2 (8.4) <sup>ab</sup>	5.1 (0.6) <sup>a</sup>
	Shelterbelt	47.7 (8.7) <sup>b</sup>	4.0 (0.7) <sup>a</sup>
	Silvopasture	81.3 (8.4) <sup>a</sup>	6.0 (0.6) <sup>a</sup>
	<b>Land-use</b>		
	Forest	76.4 (6.9) <sup>a</sup>	6.0 (0.5) <sup>a</sup>
	Agriculture	53.1 (6.9) <sup>b</sup>	4.3 (0.5) <sup>b</sup>

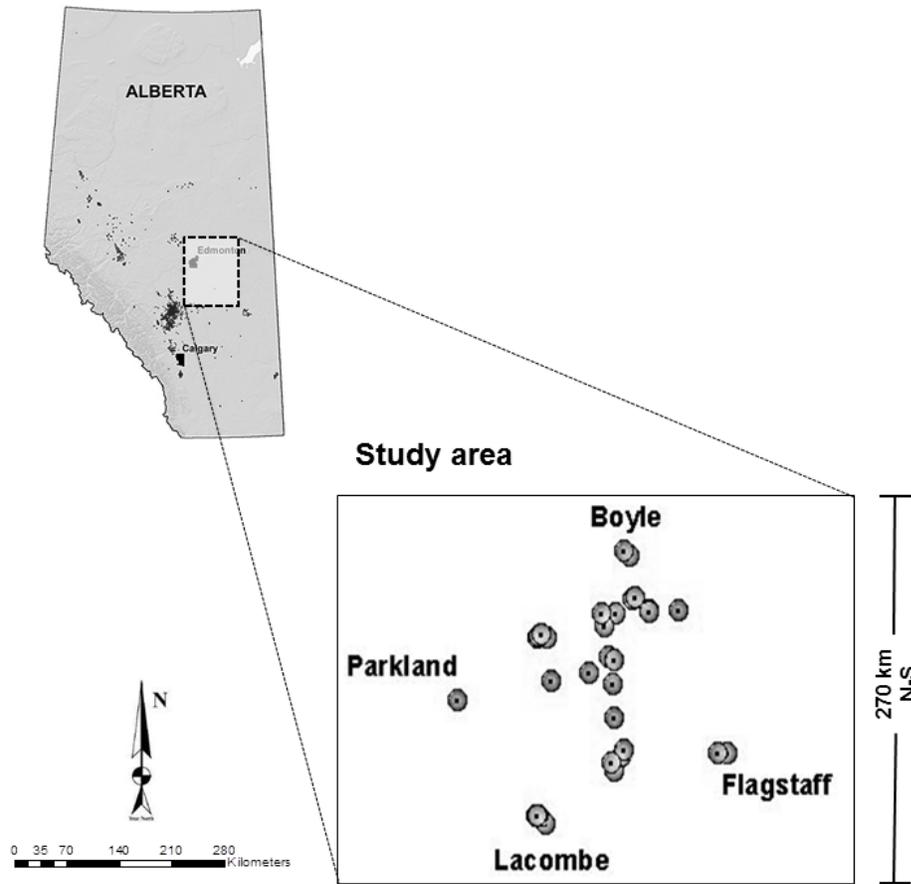
Within a column and cover type, means with the same superscript letter(s) are not significantly different

at  $p < 0.10$  according to the Fisher's protected multiple comparison test (LSD).

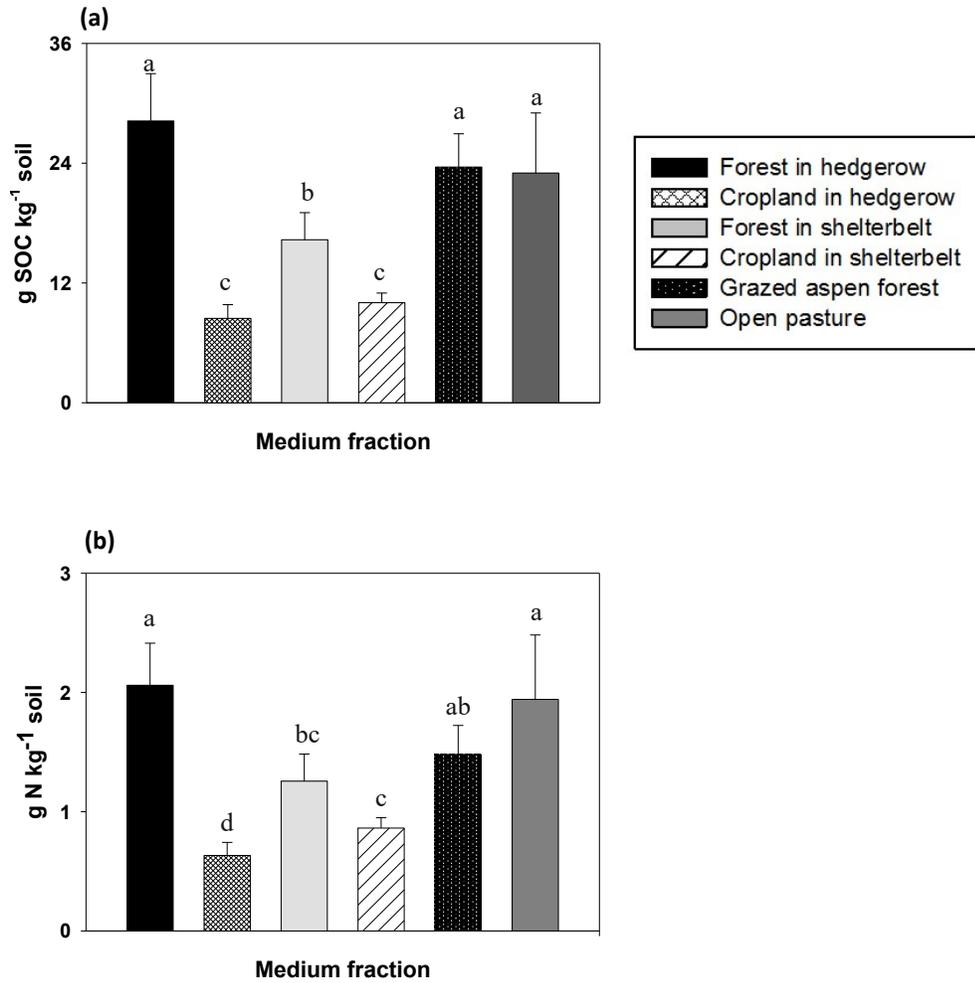
**Table 3-4.** Soil particle-size distribution (means with standard errors in parentheses) in the 0–10 cm soil layer as affected by agroforestry system and contrasting land-uses (forest and agriculture) in central Alberta, Canada.

Cover type	Particle-size distribution (%) <sup>§</sup>		
	< 53 $\mu\text{m}$	53–250 $\mu\text{m}$	250–200 $\mu\text{m}$
<b>Agroforestry system</b>			
Hedgerow	53.5 (2.0) <sup>a</sup>	27.2 (1.6) <sup>ab</sup>	18.9 (1.2) <sup>b</sup>
Shelterbelt	57.4 (2.0) <sup>a</sup>	28.6 (1.6) <sup>a</sup>	14.9 (1.3) <sup>c</sup>
Silvopasture	45.6 (2.1) <sup>b</sup>	24.4 (1.5) <sup>b</sup>	29.9 (1.2) <sup>a</sup>
<b>Land-use</b>			
Forest	51.6 (1.6) <sup>a</sup>	26.9 (1.3) <sup>a</sup>	21.8 (1.1) <sup>a</sup>
Agriculture	52.6 (1.6) <sup>a</sup>	26.5 (1.3) <sup>a</sup>	20.7 (1.1) <sup>a</sup>

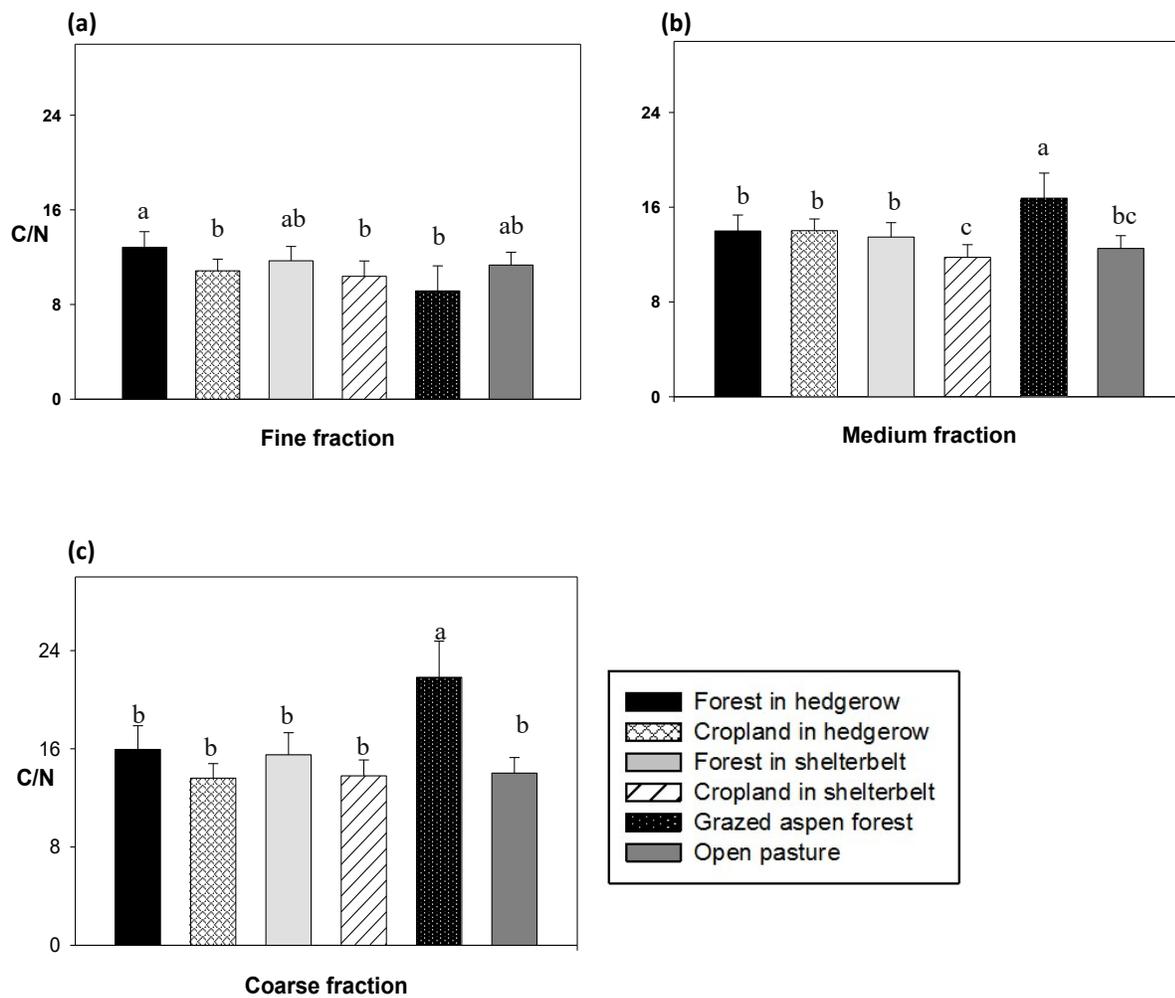
<sup>§</sup>Soil particle-size distribution is reported as percent of total whole sample weight; Within a column and cover type, means with the same superscript letter(s) are not significantly different at  $p < 0.10$  according to the Fisher's protected multiple comparison test (LSD).



**Fig.3-1.** Maps of the Province of Alberta showing the location of the study area; the gray dots indicate the distribution of the study sites in central Alberta. The study sites span a 270 km long north-south soil/climate gradient.



**Fig. 3-2.** Soil organic carbon (a) and nitrogen (b) concentrations in the medium (53–250  $\mu\text{m}$ ) fraction in the 0–10 cm soil as affected by land-use type in three agroforestry systems in central Alberta, Canada. Mean ( $\pm$  SE) values with different lowercase letter(s) were significantly different among the land-uses at  $p < 0.10$ .



**Fig. 3-3.** Soil C/N ratios a) in the fine (<math><53 \mu\text{m}</math>), b) medium (<math>53\text{--}250 \mu\text{m}</math>), and c) coarse (<math>250\text{--}2000 \mu\text{m}</math>) fractions in the 0–10 cm soil layer, as affected by land-use type in three agroforestry systems in central Alberta, Canada. Mean ( $\pm$  SE) with different lowercase letter(s) were significantly different among the land-uses at  $p < 0.10$ .

## References

- Adams, B.W., Ehlert, G., Stone, C., Alexander, M., Lawrence, D., Willoughby, D., Hincz, C., Burkinshaw, A., Carlson, J., France, K., 2009. Rangeland Health Assessment for Grassland, Forest and Tame Pasture. Alberta Sustainable Resources Department. Public Lands Division. Edmonton. AB. Pub. No. T/044.
- Alberta Agriculture and Rural Development, 2010. Alberta Fertilizer Guide [online]. [http://www1.agric.gov.ab.ca/\\$department/deptdocs.nsf/all/agdex3894/\\$file/541-1.pdf?OpenElement](http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex3894/$file/541-1.pdf?OpenElement) [site last visited in August 2013]
- Albrecht, A., Kandji, S.T., 2003. Carbon sequestration in tropical agroforestry systems. *Agric. Ecosyst. Environ.* 99, 15–27.
- Amonette, J., Kim, J., Russell, C., 2004. Enhancement of soil carbon sequestration: A catalytic approach. *J. Am. Chem. Soc.* 227, U1094–U1094.
- Arevalo, C.B.M., Bhatti, J.S., Chang, S.X., Sidders, D., 2009. Ecosystem carbon stocks and distribution under different land-uses in north central Alberta, Canada. *For. Ecol. Manage.* 257, 1776–1785.
- Arevalo, C.B.M., Chang, S.X., Bhatti, J.S., Sidders, D., 2012. Mineralization potential and temperature sensitivity of soil organic carbon under different land uses in the Parkland Region of Alberta, Canada. *Soil Sci. Soc. Am. J.* 76, 241–251.

- Ayres, E., Steltzer, H., Berg, S., Wallenstein, M.D., Simmons, B.L., Wall, D.H., 2009. Tree species traits influence soil physical, chemical, and biological properties in high elevation forests. *Plos One* 4. e5964. DOI: 10.1371/journal.pone.0005964
- Binkley, D., Richter, D., 1987. Nutrient cycles and H<sup>+</sup> budgets of forest ecosystems. *Adv. Ecol. Res.* 16, 1–51.
- Bronick, C.J., Lal, R., 2005. Manuring and rotation effects on soil organic carbon concentration for different aggregate size fractions on two soils in northeastern Ohio, USA. *Soil Till. Res.* 81, 239-252.
- Cambardella, C.A., Elliott, E.T., 1994. Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. *Soil Sci. Soc. Am. J.* 58, 123–130.
- Christensen, B.T., 2001. Physical fractionation of soil and structural and functional complexity in organic matter turnover. *Eur. J. Soil Sci.* 52, 345–353.
- Compton, J.E., Boone, R.D., 2000. Long-term impacts of agriculture on soil carbon and nitrogen in New England forests. *Ecology* 81, 2314–2330.
- Costa, M.H., Foley, J.A., 2000. Combined effects of deforestation and doubled atmospheric CO<sub>2</sub> concentrations on the climate of Amazonia. *J. Clim.* 13, 18–34.
- Creamer, C.A., Filley, T.R., Boutton, T.W., Oleynik, S., Kantola, I.B., 2011. Controls on soil carbon accumulation during woody plant encroachment: Evidence from physical fractionation, soil respiration, and delta C-13 of respired CO<sub>2</sub>. *Soil Biol. Biochem.* 43,1678–1687.

d'Annunzio, R., Conche, S., Landais, D., Saint-Andre, L., Joffre, R., Barthes, B.G., 2008.

Pairwise comparison of soil organic particle-size distributions in native savannas and Eucalyptus plantations in Congo. *For. Ecol. Manage.* 255, 1050–1056.

Elliott, E.T., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* 50, 627–633.

Environment Canada. 2012. Alberta Weather Condition [online]

[http://weather.gc.ca/forecast/canada/index\\_e.html?id=AB](http://weather.gc.ca/forecast/canada/index_e.html?id=AB), [site last visited in May 2013]

Frame, J., 2005. Forage Legumes for Temperate Grasslands. Science Publishers Inc., Enfield, NH, USA. 309 pp.

Golluscio, R.A., Austin, A.T., Garcia Martinez, G.C., Gonzalez-Polo, M., Sala, O.E., Jackson, R.B., 2009. Sheep grazing decreases organic carbon and nitrogen pools in the Patagonian Steppe: combination of direct and indirect effects. *Ecosystems* 12, 686–697.

Government of Alberta. 2013. Central parkland range plant community guide. Range plant communities and range health assessment guidelines for the central parkland subregion of Alberta. Second approximation, Pub. No. T/125. Prepared by Kupsch, T., France, K., Loonen, H., Burkinshaw, A., Willoughby, M.G., and Ron L. McNeil Landwise Inc., Range Management Branch, Alberta Environment and Sustainable Resource Development, Edmonton, AB.

Gupta, N., Kukal, S.S., Bawa, S.S., Dhaliwal, G.S., 2009. Soil organic carbon and aggregation under poplar based agroforestry system in relation to tree age and soil type. *Agrofor. Syst.* 76, 27–35.

- Haile, S.G., Nair, P.K.R., Nair, V.D., 2008. Carbon storage of different soil-size fractions in Florida silvopastoral systems. *J. Environ. Qual.* 37, 1789–1797.
- Hassink, J., 1997. The capacity of soils to preserve organic C and N by their association with clay and silt particles. *Plant Soil* 191, 77–87.
- Heitschmidt, R.K., Stuth, J., 1991. *Grazing management, an ecological perspective*. Timber Press, Inc. Portland, Oregon. 18 pp.
- Howlett, D.S., Mosquera-Losada, M.R., Nair, P.K.R., Nair, V.D., Rigueiro-Rodriguez, A., 2011. Soil carbon storage in silvopastoral systems and a treeless pasture in Northwestern Spain. *J. Environ. Qual.* 40, 825–832.
- IPCC, 2013. *Climate Change 2013: The Physical Science Basis. Summary for Policy Makers*. Stockholm. Sweden.
- Jobbagy, E., Jackson, R., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecol. Appl.* 10, 423–436.
- John, B., Yamashita, T., Ludwig, B., Flessa, H., 2005. Storage of organic carbon in aggregate and density fractions of silty soils under different types of land use. *Geoderma* 128, 63–79.
- Johnson, D.W., 1992. Effects of forest management on soil carbon storage. *Water Air Soil Pollut.* 64, 83–120.
- Johnson, D.W., Curtis, P.S., 2001. Effects of forest management on soil C and N storage: meta-analysis. *For. Ecol. Manage.* 140, 227–238.

- Kalra, Y.P., Maynard, D.C., 1991. *Methods Manual for Forest Soil and Plant Analysis*.  
Information Report - Northwest Region, Forestry Canada viii + 116 pp.
- Kort, J., Turnock, R., 1998. Carbon reservoir and biomass in Canadian prairie shelterbelts.  
*Agrofor. Syst.* 44, 175–186.
- Li, Y.F., Zhang, J.J., Chang, S.X., Jiang, P.K., Zhou, G.M., Shen, Z.M., Wu, J.S., Lin, L., Wang, S.S., Shen, M.C., 2014. Converting native shrub forests to Chinese chestnut plantations and subsequent intensive management affected soil C and N pools. *For. Ecol. Manage.* 312, 161–169.
- Mead, D.J., Comforth, I.S., Eds., 1995. *Proceedings of the Tree and Soil Workshop, Lincoln University 28 February–2 March 1994*. Agronomy Society of New Zealand Special Publication No. 10. Lincoln University Press, Canterbury.
- Montagnini, F., Nair, P.K.R., 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. *Agrofor. Syst.* 61, 281–295.
- Nair, P.K.R., Buresh, R.J., Mugendi, D.N., Latt, C.R., 1998. Nutrient cycling in tropical agroforestry systems: Myths and Science, in: Buck, L.E., Lassoie, J.P., Fernandes, E.C.M. (eds), *Agroforestry in Sustainable Agricultural Systems*. CRC Press
- Nair, P.K.R., Nair, V.D., Kumar, B.M., Showalter, J.M., 2010. Carbon sequestration in agroforestry systems. *Adv. Agron.* 108, 237–307.
- Nason, G.E., Pluth, D.J., McGill, W.B., 1988. Volatilization and foliar recapture of ammonia following spring and fall application of nitrogen-15 urea to a Douglas-fir ecosystem. *Soil Sci. Soc. Am. J.* 52, 821–828.

- Nzila, J.D., Bouillet, J.P., Laclau, J.P., Ranger, J., 2002. The effects of slash management on nutrient cycling and tree growth in Eucalyptus plantations in the Congo. *For. Ecol. Manage.* 171, 209–221.
- Oelbermann, M., Voroney, R.P., Gordon, A.M., 2004. Carbon sequestration in tropical and temperate agroforestry systems: a review with examples from Costa Rica and southern Canada. *Agric. Ecosyst. Environ.* 104, 359–377.
- Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K., 2002. Change in soil carbon following afforestation. *For. Ecol. Manage.* 168, 241–257.
- Paustian, K., Six, J., Elliott, E.T., Hunt, H.W., 2000. Management options for reducing CO<sub>2</sub> emissions from agricultural soils. *Biogeochemistry* 48, 147–163.
- Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: processes and potential. *Glob. Change Biol.* 6, 317–327.
- Potter, K.N., Daniel, J.A., Altorn, W., Torbert, H.A., 2001. Stocking rate effect on soil carbon and nitrogen in degraded soils. *J. Soil Water Conserv.* 56, 233–236.
- Prescott, C.E., Blevins, L.L., Staley, C., 2004. Litter decomposition in British Columbia forests: Controlling factors and influences of forestry activities. *J. Ecosyst. Manage.* 5, 44–57.
- Reeder, J.D., Schuman, G.E., 2002. Influence of livestock grazing on C sequestration in a semi-arid mixed-grass and short-grass rangelands. *Environ. Pollut.* 116, 457–463.
- Sainju, U.M., Stevens, W.B., Caesar-TonThat, T., Liebbig, M.A., 2012. Soil greenhouse gas emissions affected by irrigation, tillage, crop rotation, and nitrogen fertilization. *J. Environ. Qual.* 41, 1774–1786.

- SAS Institute Inc., 2013. SAS User's Guide: Statistics SAS/C Online Doc, Release 9.30. SAS Inc., Cary, NC
- Schnabel, R.R., Franzluebbers, A.J., Stout, W.L., Sanderson, M.A., Stuedemann, J.A., 2001. The effects of pasture management practices. In: Follet, R.F., Kimble, J.M., and Lal, R. (eds.): The potential of U.S. grazing lands to sequester carbon and mitigate the greenhouse effect. 291–322 pp. Lewis Publishers, Boca Raton, FL.
- Sharrow, S.H., Ismail, S., 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. *Agrofor. Syst.* 60, 123–130.
- Six, J., Conant, R., Paul, E., Paustian, K., 2002. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant Soil* 241, 155–176.
- Six, J., Elliott, E.T., Paustian, K., 2000. Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biol. Biochem.* 32, 2099–2103.
- Sollins, P., Homann, P., Caldwell, B., 1996. Stabilization and destabilization of soil organic matter: Mechanisms and controls. *Geoderma* 74, 65–105.
- Soil Classification Working Group. 1998. The Canadian System of Soil Classification. NRC Research Press, Ottawa, Canada, 187 pp.
- Takimoto, A., Nair, V.D., Nair, P.K.R., 2009. Contribution of trees to soil carbon sequestration under agroforestry systems in the West African Sahel. *Agrofor. Syst.* 76, 11–25.
- Teklay, T., Chang, S.X., 2008. Temporal changes in soil carbon and nitrogen storage in a hybrid poplar chronosequence in northern Alberta. *Geoderma* 144, 613–619.

- Thevathasan, N.V., Gordon, A.M., 1997. Poplar leaf biomass distribution and nitrogen dynamics in a poplar-barley intercropped system in southern Ontario, Canada. *Agrofor. Syst.* 37, 79–90.
- Tiessen, H., Stewart, J.W.B., 1983. Particle-size fractions and their use in studies of soil organic matter. 2. Cultivation effects on organic matter composition in size fractions. *Soil Sci. Soc. Am. J.* 47, 509–514.
- Yamashita, T., Flessa, H., John, B., Helfrich, M., Ludwig, B., 2006. Organic matter in density fractions of water-stable aggregates in silty soils: Effect of land use. *Soil Biol. Biochem.* 38, 3222–3234.

## **Chapter 4. Forest and grassland cover types increase carbon pool size and stability in agroforestry systems: evidence from density fractionation\***

### **1. Introduction**

Soil organic matter (SOM) is a complex mix of plant and animal residues in various degrees of decomposition that affects soil quality, tilth and productivity, and regulates global carbon (C) cycling (Alvarez and Alvarez 2000; Lal 2002). Most studies over the past several decades demonstrate that conversion of lands from forest to herbland can significantly decrease SOM levels (Burke et al. 1989; Bonde et al. 1992; Paustian et al. 2000; Sanju et al. 2012). Cultivation alters the amount, timing, and quality of organic residue input into soils (Sollins et al. 1996; Campbell et al. 1999), which in turn, has an effect on the size, rate of recycling and distribution of C among SOM pools (Kang 1997; Christensen 2000). In contrast, forested land cover can increase SOM because of the continuous deposition of plant litter (Oelbermann et al. 2004; Paul et al. 2002) and limited removal of biomass because of infrequent harvesting (Six et al. 1998; Ayres et al. 2009; Nascente et al. 2013).

Agroforestry is a unique land use system that intentionally blends perennial vegetation and herbaceous land cover types to enhance crop productivity, profitability, and overall soil quality in agroecosystems. In essence, agroforestry combines trees with either annual crops and/or perennial pastures to increase sustainability of agricultural lands (Montagnini and Nair

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2004; Nair et al. 2009). Further, these systems contribute other ecosystem services such as providing wildlife habitat (Jose 2009), maintaining biodiversity (Altieri 1999), reducing erosion (Lenka et al. 2012), and enhancing microbial communities in soil (Banerjee et al. 2015). Trees in agroforestry systems constitute a significant avenue of organic matter (and nutrient) addition to the soil ecosystem (Haile et al. 2008; Takimoto et al. 2008; Isaac et al. 2011). Integrating trees into the agricultural landscape can increase the above- and belowground total productivity of agroecosystems, modify rooting depth and root distribution, and enhance organic matter input to the soil from litterfall (Kass et al. 1997; Paul et al. 2002; Partey 2011; Albrecht and Kandji 2003). However, increasing SOM accumulation over time through the incorporation of trees is just one step to promoting SOM build-up, as the stability of resulting SOM pools is what ultimately regulates the cycling of C and soil quality (Crow et al. 2007; Dorodnikov et al. 2011; Creamer et al. 2013).

Soil organic C (SOC) is composed of fractions (pools) that differ in stability (Sollins et al. 1996; von Lützow et al. 2006; Strosser 2010). Organic C may be available in the soil as either: (i) relatively fresh (labile) SOM not protected in soil aggregates, (ii) SOM physically protected through occlusion in aggregates, or (iii) SOM chemically stabilized through association with mineral surfaces (Swanston and Myrold 1997; Amonette and Russel 2004; von Lützow et al. 2007). Land use and management practices can influence SOC and distribution among various pools (Tiessen and Stewart 1983; Teklay and Chang 2008; Mujuru et al. 2013). The labile SOC fraction has a significant influence on soil quality and productivity (Chen et al. 2012), and tends to respond the most to management activities (Crow et al. 2007; Duval et al. 2013). As such, labile SOC is a more sensitive indicator of management and land use change compared to other fractions. Therefore, isolation of these functional pools will help to elucidate

the net impact of land management systems on C storage, its stability and overall soil quality (Jastrow 1996; Oades 1984; Chen et al. 2012).

The study in Chapter 1 relied on soil particle size separation to isolate functional SOM pools in the three agroforestry systems (hedgerows, shelterbelts, and silvopastures). Soil size fractions obtained by wet-sieving after shaking for 30 min in water are assumed to offer different degrees of SOM stabilization (Hassink 1997; Moni et al. 2012). However, fractionation by particle size does not allow differentiation of many of the light fraction materials (e.g., incompletely decomposed organic residues – labile organic matter) from the more decomposed and protected mineral-bound organic matter (Moni et al. 2012), thus providing only a rough separation between active, intermediate, and passive SOM pools (von Lützow et al. 2007). A better understanding of SOM storage and dynamics within ecosystems requires more comprehensive separation of the various functional C pools, particularly the labile light fraction of SOM.

The present study expands on the previous work in Chapter 1 by examining the effects of three agroforestry systems (hedgerow, shelterbelt, and silvopasture) and their inherent land cover types (forested vs herbland) on (i) mineral soil organic C and nitrogen (N) in the bulk soil and (ii) the organic C and N distribution among the light, occluded, and heavy fractions in the 0-10 and 10-30 cm soil layers. The distribution of C and N among the different SOM fractions could provide important insight into how agroforestry systems affect the quality and long-term stability of organic matter in soils.

## **2. Materials and Methods**

## 2.1. Site description

This study was conducted in central Alberta, Canada, at 36 study sites distributed across a large geographic range. Study sites were located between 54° 43' and 52° 28' N latitude, and between 113° 44' and 113° 17' W longitude. Elevations of the study sites ranged from 533 to 850 m above mean sea level. Average air temperature based on 30 years of data (1981-2010) collected from 26 Environment Canada climate stations, was 1.9 °C and 3.3 °C, and mean annual precipitation was 463 mm and 497 mm, in the north and south portions of the study area, respectively (Environment Canada 2012). Landforms in the study area vary markedly from relatively level plains in the south to moderately or strongly rolling hills in the northwest. Historical vegetation in the region includes the Dry Boreal Mixedwood and Aspen Parkland Natural Subregions (Adams et al. 2009). Dominant soil types vary from Luvisols (Soil Classification Working Group 1998) in the north, to Dark Gray Chernozems in the central portion, and Black Chernozems in the south of the sampling area.

This study examined three dominant agroforestry systems: hedgerow, shelterbelt, and silvopasture systems. Each of these systems consisted of two land cover types: forested (areas with trees) and herbland (areas without trees). Hedgerow systems were made up of naturally regenerating perennial vegetation that included woody species as hedgerows at field edges and adjacent annual cropland. Shelterbelt systems were also comprised of trees and shrubs planted in 1–2 rows as shelterbelts, and were adjacent to annual cropland. Silvopasture systems contained a mosaic of grazed aspen forest and grassland land cover types. Hedgerows are usually 40- to 100-year-old broad-leaved deciduous stands intermixed with trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), saskatoon serviceberry (*Amelanchier alnifolia*), and

chokecherry (*Prunus virginiana*). Shelterbelts generally consist of one or two rows of trees (3-5 m wide) comprising 20- to 50-year-old coniferous and/or deciduous trees dominated by white spruce (*Picea glauca*) with interspersed caragana (*Caragana arborescens*), willow (*Salix acutifolia*), and box elder (*Acer negundo*). Croplands in both the hedgerow and shelterbelt system are typically planted to cereal, oilseed or pulse crops such as *Hordeum vulgare* L., *Triticum aestivum* L., *Brassica napus* L., and *Pisum sativum* L., with minimum tillage practices and ample fertilizer applied annually that includes N up to 120 kg ha<sup>-1</sup> year<sup>-1</sup>. Silvopastures encompass mostly *Populus tremuloides* forested patches intermixed with grasslands comprised of a variety of species, often including *Bromus inermis* Leyss and *Poa pratensis* L. Tree species composition and ages of the *Populus tremuloides* dominant forest vegetation were similar to the hedgerows; however the hedgerows contained more understory herbs and shrubs species compared to the forest vegetation found in the silvopasture system. Fewer understory herb and shrub species occurred in silvopastures because cattle (*Bos taurus*) grazing tended to simplify the understory by thinning out tall forbs and shrubs.

## 2.2. Sampling design

The experiment used a split-plot in a completely randomized design (CRD). Whole plots consisted of the three agroforestry systems (i.e., hedgerow, shelterbelt, and silvopasture), each of which was divided into forested and adjacent hermland as sub-plots. Across the study area, 36 sites (12 hedgerows, 12 shelterbelts and 12 silvopastures) were selected along the soil/climate gradient. Within each site (i.e. agroforestry plot), one 30 m long transect was established inside each of the forested and paired hermland sub-plots. Transects in the forested sub-plots were established in the center of the treed zone,

whereas those in the herbland land cover type were located at least one tree height (~ 30 m) from the edge of the treed zone.

Soil samples were collected in June 2013. Samples were collected from two depths in the mineral soil, 0-10 and 10-30 cm, using a 3.2 cm diameter core, thereby excluding the fresh litter, fibric and humified surficial organic matter (i.e. mulch or LFH) layer. Along each transect, 10 cores were systematically collected, separated into the two depth-classes, combined within a class, and then mixed in the field to form a composite sample from each transect. Additional soil samples were collected at each depth by inserting three metal rings of 106 cm<sup>3</sup> volume into the soil for bulk density measurements. Soil samples were placed in plastic bags and kept cool (~4 °C) until processed. In the laboratory, fresh soil samples were sieved to pass a 2 mm screen (# 10 U.S. Standard Sieve) and separated into two subsamples. One subsample was stored in a -20 °C freezer for chemical analyzes, while the other subsample was air-dried at room temperature (~ 20-25 °C) in preparation for organic C and N analysis.

### **2.3. Soil physical and chemical analyses**

Soil organic C and N concentrations of the whole soils were analyzed by dry combustion using a LECO Tru-Spec CN analyzer (Leco Corp., St. Joseph, MI, USA). For samples with pH>6.4, organic C concentrations were determined after acid fumigation to remove inorganic C following Harris et al. (2001). To determine bulk density, samples taken with the metal rings were oven-dried at 105 °C to constant mass and weighed. Bulk density of the soil was then calculated by dividing dry mass by metal ring volume. Soil texture (clay < 2 µm, silt 2–63 µm, sand 63–2000 µm) was determined by the hydrometer method (Carter and Gregorich 2006) after dispersing 40

g of soil (100 g if loamy sand or sandy soil) in 400 mL of Calgon<sup>®</sup> [(NaPO<sub>3</sub>)<sub>6</sub>] solution (50 g L<sup>-1</sup>). Ammonium (NH<sub>4</sub>-N) and nitrate (NO<sub>3</sub>-N) were determined by auto-analyzer (SmartChem Discrete Wet Chemistry Analyzer, Westco Scientific Limited, Brookfield, CT) after extraction using 2 M KCl solution at a ratio of 1:5 (soil: KCl; *w:v*). To estimate cation exchange capacity (CEC), 30 mL of 0.1 M BaCl<sub>2</sub> solution was added to 3 g air-dried soil in a 50 mL centrifuge tube. The soil sample was slowly shaken on a flatbed shaker for 2 h (15 rpm). Cation exchange capacity was then analyzed by atomic absorption spectrophotometry (AAS) after centrifuging the soil solution (23 min) and filtering the supernatant with Whatman No. 41 filter paper (20 μm pore size).

#### **2.4. Soil density fractionation**

Soil samples were separated by physical fractionation to obtain three organic matter density fractions (light, occluded, and heavy fraction). These fractions were recovered by flotation in a 1.6 g cm<sup>-3</sup> density solution of sodium polytungstate (SPT, 3Na<sub>2</sub>WO<sub>4</sub>.9WO<sub>3</sub>.XH<sub>2</sub>O; SOMETU, Berlin). A hydrometer was used to verify solution density. The general procedure involved adding 741 g of SPT to 859 mL of water to produce the 1.6 g cm<sup>-3</sup> of SPT solution. Next, 10 g of air-dried soil (< 2 mm) was weighed into a conical centrifuge tube, to which 40 mL of the prepared SPT solution was added. The tube containing the soil and SPT was gently mixed by inverting 5 to 10 times and allowing it to stand for 1 h. The suspension was then centrifuged at 2250 g for 23 min.

The resulting supernatant was aspirated through a Whatman GF/F filter (0.7 μm pore size) and drawn through a Buchner's funnel using vacuum pressure. The resulting material

contained the light fraction, which was then rinsed five times with deionized water to remove the SPT, and dried at 50 °C for 48 h. Next, the SPT solution ( $1.6 \text{ g cm}^{-3}$ ) was added to the soil material after the light fraction materials were removed. The sample was dispersed ultrasonically with a Fisher Dismembrator (Model 300, Fisher Scientific, Pittsburgh, PA) at 360 W for 5 min (Haile et al., 2008). The sample was allowed to stand for 23 min, then centrifuged, filtered, rinsed, and oven-dried as described for the light fraction, and this yielded the occluded fraction. The remaining precipitate that contained the heavy fraction was also thoroughly rinsed at least five times for clayey soils and three times for sandy soils in order to remove all SPT. The heavy fraction was oven-dried at 50 °C. Finally, the light, occluded, and heavy fractions were each ground to a fine powder using a ball mill (Mixer Mill MM200, Thomas Scientific, Swedesboro NJ) for 30 seconds, then analyzed for organic C and N by combustion with the LECO Tru-Spec CN analyzer. Soil bulk density measured for each sampling depth was used to convert C and N concentrations in bulk soils and fractions to mass stocks ( $\text{Mg ha}^{-1}$ ).

## **2.5. Statistics**

Effects of agroforestry systems (hedgerow, shelterbelt, and silvopasture) and land cover types (forested, adjacent herbland) within the agroforestry systems on soil physical and chemical properties, as well as on SOC and N stocks, and percent mass distributions of the various SOM fractions, were each analyzed using Statistical Analysis Software (SAS v. 9.3, SAS Institute Inc. 2013). All data were tested for normality and homogeneity of variance using Shapiro-Wilk's and Levene's tests, respectively. A log transformation was applied to the percent mass distribution of light, occluded, and heavy fractions to meet the assumption of normality. However,

untransformed data are presented in this paper for ease of interpretation. Analysis of variance was then performed using the Proc MIXED procedure in SAS and according to Eqn. 1:

$$Y_{ijk} = \mu + A_i + (A\gamma)_{ik} + L_j + (AL)_{ij} + \varepsilon_{ijk} \quad ; \quad (1)$$

where  $Y_{ijk}$  is a dependent variable,  $\mu$  is the overall mean,  $A_i$  and  $L_j$  are the effects of the  $i$ th and  $j$ th agroforestry system and land cover type, respectively,  $(A\gamma)_{ik}$  and  $\varepsilon_{ijk}$  are the random variable error within the experiment. Fisher's protected least significant difference (LSD) test was used to identify and separate significant main and interaction effects at  $P < 0.10$ . Following the previous study Chapter 1, a  $P$  value of 0.10 was used to evaluate significance and minimize risk of a type II error. Even though the sample size was large ( $n = 12$ ), the risk of a type II error was considered high given that the study was conducted across a wide geographic range (270 km long soil/climate gradient) that had substantial variation in soil properties and vegetation composition.

### **3. Results**

#### **3.1. Soil properties**

There was no interaction between agroforestry system and land cover type on any soil parameter (all  $P > 0.10$ ) in the 0-10 cm layer (Table 4-1). Although soil bulk density was not different among agroforestry systems, there was a significant difference between land cover types such that soils in herblands had 19% greater bulk densities than areas occupied by trees. Neither

agroforestry system nor land cover type affected  $\text{NH}_4\text{-N}$  content or CEC, but  $\text{NO}_3\text{-N}$  content was significantly altered by both agroforestry system and land cover type (Table 4-1). Nitrate content in the silvopasture system was only 22 to 25% that of the other two systems. The amount of  $\text{NO}_3\text{-N}$  was also 4 times greater in herblands compared to areas occupied by trees across the three agroforestry systems. Clay content was significantly lower and sand content greater in the silvopasture system compared to the other two agroforestry systems. However, clay, silt, and sand contents were statistically similar ( $P \geq 0.10$ ) between the two land cover types.

### **3.2. Percent mass distributions of soil density fractions**

Average recovery of soil samples during density fractionation was 97% in all land use treatments (Fig. 4-1). Incomplete recovery of total soil mass during the fractionation procedure could be attributed in part to the loss of mostly clay particles in the discarded filtrate and a portion of the SOM being solubilized by the SPT solution. Mean masses of the light, occluded, and heavy fractions within the 0-10 and 10-30 cm layers were not affected by the interaction of agroforestry system and land cover type (Table 4-2). On average, the masses of light, occluded, and heavy fractions accounted for 13, 14, and 70% of the dry bulk soil, respectively, for the 0-10 cm layer, and 7, 12, and 79% of dry bulk soil weight for the 10-30 cm layer (Fig. 4-1). Within the 0-10 cm layer, the mass of the heavy fraction in the shelterbelt system was 18% and 27% greater than that found in the hedgerow and silvopasture, respectively. In terms of land cover type, the amount of light fraction SOM was significantly greater in forest than in neighbouring herblands. In contrast however, the mass of heavy fraction SOM in herblands was significantly greater than that found in forested areas. Soils under herblands contained significant amounts of occluded SOM in the

10-30 cm layer compared to the areas with trees in all agroforestry systems. However, areas with trees had a greater proportion of heavy SOM in bulk soils of the 10-30 cm layer compared to adjacent herblands.

### **3.3. SOC and N stocks in soil density fractions and bulk soil**

The interaction between agroforestry system and land cover type was not significant for any of the SOC and N stocks, as well as C/N ratios, regardless of soil depth or fractional component (Table 4-3). Soil C effects were particularly pronounced in the 0-10 cm layer in relation to agroforestry system and land cover type, and in the C/N ratio in response to agroforestry system (Table 4-3). Levels of C in the top 10 cm of mineral soil were significantly greater in the silvopasture than the hedgerow and shelterbelt systems, although differences varied among fractions. The silvopasture had more light fraction C and less heavy fraction C than the shelterbelt system (Table 4-4). Soil organic C did not differ among agroforestry systems within the 10-30 cm layer. On average, the silvopasture system had 30 and 46 Mg ha<sup>-1</sup> more C than the hedgerow and shelterbelt systems within the 0-10 and 10-30 cm layers, respectively. Between land cover types, forested areas had significantly more C than neighboring herblands for both soil layers, although this increase was attributed to C in the light fraction at both soil depths, as well as occluded C in the shallow (0-10 cm) depth (Table 4-4).

The absence of significant effects of agroforestry system on bulk soil N content was consistent for the 0-10 and 10-30 cm soil layers (Table 4-4). Despite this however, N stocks varied among agroforestry systems within individual soil fractions. The content of N in the heavy fraction was greater in shelterbelts, followed by hedgerows and then silvopastures, a trend

that occurred for both soil layers. A similar pattern was detected for N in the occluded fraction at the 10-30 cm soil layer. Between land cover types, forested areas had greater N stocks (2.7 Mg N ha<sup>-1</sup>, on average) than in the neighboring herblands for both soil layers, a response particularly evident in the light fraction of soils within the top 10 cm of soil.

Substantial variation existed in C/N ratio among agroforestry systems within SOM fractions. The largest C/N ratios in both the 0-10 cm and 10-30 cm layers were mostly in the silvopasture (Table 4-4). The C/N ratios of soils in forested areas were greater than in adjacent herbland soils under herbaceous vegetation in both soil layers, but only within the light fraction.

#### **4. Discussion**

Across all land use treatments, soil bulk density in areas occupied by trees was lower than that in the adjacent herblands and indicates a greater degree of soil aggregation and potentially reduced compaction from vehicles and/or livestock in the former (Huntington et al. 1989). Frequent traffic passes induced by a combination of large grazing animals in grasslands (Chanasyk and Naeth 1995; Donkor et al. 2002) and conventional tillage in annual croplands (Grant et al. 1993; Lafond et al. 2011) could disrupt soil structure leading to corresponding increases in bulk density in herblands exposed to intensive agricultural land use. The finding that clay and sand contents were significantly different between the silvopasture and the other two agroforestry systems is important, because it indicates that silvopastures are generally established on (or relegated to) lower quality ecosites characterized by less silt/clay contents. Soil texture is a major driver of composition and productivity across grassland ecosystems in North America (Epstein et al. 1997). Generally, fine-textured soils will support greater ecosystem production than coarse-

textured soils in areas of high mean annual precipitation (near 800 mm); the opposite pattern results if mean annual precipitation is closer to 370 mm (Epstein et al. 1997). Most of the silvopastures studied are located in areas receiving between 448 mm and 463 mm of mean annual precipitation (Environment Canada 2012) and this might negatively impact resulting C levels in silvopastures compared to the other systems.

In a related study where soil samples were taken at exactly the same sites between September and October 2012, no differences in NO<sub>3</sub>-N levels were found for any of the agroforestry systems studied (Baah-Acheamfour et al. 2014). However, significant amounts of NO<sub>3</sub>-N in both the hedgerow and shelterbelt systems relative to the silvopasture in the present study could have come from the application of N-based fertilizers to their inherent croplands at the start of the growing season. Furthermore, cattle (*Bos taurus*) grazing in silvopastures during summer are likely to deposit substantial N as urine and feces. However, this response may be short-lived since the bulk of N nitrifies to NO<sub>3</sub>-N during the early season after snow melt and is subject to leaching (Potter et al. 2001). In addition, perennial grasses and other herbs in the silvopasture system are able to take up any early season pulse of mineral N, which could contribute to the lower NO<sub>3</sub>-N levels observed in silvopastures at the time of soil sampling in June compared to that at the end of the growing season (Baah-Acheamfour et al. 2014; Campbell et al. 1999; Gebauer and Ehleringer 2000).

When SOC and N stocks in bulk soils were compared within any given layer, particularly at the 0-30 cm depth, the silvopasture system stored more C than both the hedgerow and shelterbelt agroforestry systems. Variations in SOM accumulation among agroforestry systems can reflect the type of tree species present and the intensity of past and present land management practices. For instance, Howlett et al. (2011) quantified the amount of C stored in the 0-25 cm

soil layer in silvopasture plots of Silver birch (*Betula pendula*) and Monterey pine (*Pinus radiata*), and found the former (i.e. deciduous tree system) contained more C than the latter. Paul et al. (2002) also indicated that deciduous species were better at sequestering C in soils than conifers due to their relatively high N levels in foliage (thus lowering C/N ratios). Soil microbes can assimilate organic matter better in deciduous systems that form C inputs to the soil. At the time of sampling for the present study in June 2013, around 95% of the forested areas of the silvopasture system were dominated by deciduous trees and understory shrubs such as trembling aspen, balsam poplar, willow, box elder and chokecherry. On the other hand, around 70% of shelterbelt systems were dominated by white spruce, with the remaining 30% comprised of deciduous trees and shrubs. Apart from plant species composition, surface soils underlying perennial grasslands are often filled with fine roots of grasses and other herbaceous vegetation. Don et al. (2009) found 59 to 86% of total root biomass in grassland was in the 0-5 cm layer. This could account for why silvopasture systems contained more C in the 0-30 cm soil layer, presumably because of high inputs of SOM associated with decomposing fine roots of grasses and other plants in this extensively vegetated hermland.

In terms of management practices, grazing by livestock within silvopastures could also produce a positive effect on SOM by physically incorporating litter (via trampling) and C (via variably broken down excreta) into the surface soil (Reeder and Schuman 2002). Even though the shelterbelt system contained small amounts of C in mineral soil, the areas occupied by trees contained a sizeable amount of LFH layer that must be assessed in determining overall soil C budgets. For instance, we examined the amount of C stored in the LFH layer for the three agroforestry systems in the summer of 2014; C stocks in the LFH layer for the hedgerow, silvopasture, and shelterbelt systems were 0.02, 0.03, and 0.04 Mg ha<sup>-1</sup>, respectively

(unpublished data). However, this C is susceptible to loss with fire and other surface disturbances and cannot be considered well protected for long-term mitigation of atmospheric CO<sub>2</sub>. Lower C levels in agroforestry systems containing annual cropping such as the hedgerow and shelterbelt systems were also found in the previous investigation (Baah-Acheamfour et al. 2014), and are attributed to the effect of physical changes to soils and associated aggregates in the areas subject to conventional cropping practices (Dalal and Mayer 1986; Kleber et al. 2004; John et al. 2005; Lafond et al. 2011), increases in microbial activity (Malhi et al. 2011), and the ongoing removal of crop biomass through harvested products (Burke et al. 1989; Bonde et al. 1992; Paustian et al., 2000; Sanju et al. 2012). Biomass removal from croplands can also influence soil conditions by removing insulating litter and warming the soil surface, thereby enhancing organic matter decomposition and loss of SOC (Witt et al. 2000).

Even though this study focused only on the upper 0-30 cm of mineral soil, the results clearly demonstrate that integrating trees into agricultural landscapes is important in the context of increasing soil C accumulation. Inclusion of trees within the agricultural landscape could also affect properties of deeper soil, not only because their roots extend to deeper depth but also as a result of the role of deeper soils in regulating long-term nutrient dynamics and C stabilization (Lal 2009). As a result, soil C in deeper layers should be taken into account when forested and herbaceous land cover types are being compared.

Land use systems may affect the distribution of SOM in various soil particle and density-size fractions within ecosystems, and these fractions are strongly involved in the storage and stabilization of soil C (Six et al. 2000). In this study, a large amount of C in the light fraction of the 0-10 cm layer was found in the silvopasture system and is consistent with findings in a previous study where the majority of C in soils of silvopastures was held in the more labile

coarse size fraction (Baah-Acheamfour et al. 2014). Maia et al. (2007) also analyzed labile SOM pools in traditional agroforestry, intensive cropping, native forest and silvopasture systems, and found soils underlying the silvopasture to preserve, and in some cases increase, the size of the labile organic C pool. I postulate that because the silvopasture systems I studied lack previous soil tillage and have reduced biomass removal they may have greater capacity to retain more labile SOM than the other agroforestry systems. The retention of more labile forms of organic matter in soils of the silvopasture system is likely to increase SOC accumulation (Montagnini and Nair 2004; Oelbermann et al. 2004); however, the bulk of that C could respond relatively rapidly to future changes in land cover (i.e. conversion to cropland) or management practices, highlighting the importance of maintaining existing land cover types to protect this C.

The generally greater C and N stocks in the heavy fraction of both the hedgerow and shelterbelt system soils compared to the silvopastures at the 0-10 and 10-30 cm layers indicate that existing SOC was relatively more stable in the first two systems than in the silvopasture. The accumulation of almost 45% (on average) of bulk soil C in the heavy fraction of both the hedgerow and shelterbelt system soils could ensure longer-term C sequestration (e.g. under environmental change), and provide more stable offset potential of atmospheric C (Cambardella and Elliott 1994; Paul et al. 1997; Tonnejck et al. 2006). Nevertheless, larger total C accumulation in silvopasture system soils suggests that this system directly reduces CO<sub>2</sub> levels in the atmosphere more than the other two systems despite being relegated to low quality sites (i.e., coarse-textured soils).

Larger amounts of C and N in the light and occluded fractions were also detected in soils under forest compared to adjacent herblands. A combination of factors such as favorable moisture and temperature conditions, increased microbial activities, reduced quantity and quality

of litter inputs, and greater management disturbances during the cropping season, may all promote greater mineralization of light and occluded SOM within herblands compared to those land cover types occupied by trees (e.g. hedgerows, shelterbelts and grazed aspen forests). Contrary to expectations, I did not observe particularly low amounts of C in the heavy fraction of herbland soils, possibly because a portion of the C in the light or occluded fraction, which is generally assumed to be promptly lost upon cultivation, could actually be present in the heavy soil fraction of this land cover type (Tiessen and Stewart 1983; Teklay and Chang 2008). Overall, C/N ratios decreased with increasing particle-density fractions (i.e. representing a shift to more C in the light fraction) for all land use treatments. High C/N ratios in the light fraction were most likely due to a large concentration of organic C in freshly decomposing plant residues, which in the silvopasture system, would be augmented by recycling of nutrients by animals in both the grazed aspen forest and adjacent perennial grassland. Low C/N ratios in the heavy fraction indicate a high degree of decomposition (Baisden et al. 2002). The heavy fraction also contains minerals such as phyllosilicates that provide a large surface area and numerous reactive sites where N-containing compounds (amine, amide, pyrrole) can directly be sorbed onto mineral surfaces (von Lützow 2007).

## **5. Conclusions**

The results demonstrate the potential for trees to increase C storage and stabilization within the top 30 cm of mineral soil in agroforestry systems distributed across the agricultural landscapes of central Alberta, Canada. On average, the silvopasture system stored 18.2% more C in the bulk soil than the other two agroforestry systems in the 0-30 cm soil layer. Between land cover types, areas occupied by trees had greater organic C and N stocks in the bulk mineral soil than the

adjacent herblands without trees, irrespective of the soil layer sampled. Mean SOC stocks in the light, occluded, and heavy soil fractions were 29, 26, and 42%, respectively, of organic C in the bulk soil within the 0-10 cm layer. Within the 10-30 cm layer, SOC stocks in the light, occluded, and heavy soil fractions comprised 30, 34, and 29%, respectively, of organic C in the bulk soil. Results also showed that soil C in the more labile light fraction of the 0-10 cm layer was significantly greater in silvopasture systems than that of the shelterbelt system. Furthermore, C in the more stable heavy fraction was greater in shelterbelts than in the silvopasture system at the 0-10 cm layer. Within each agroforestry system and across soil depths, treed areas consistently stored more C and N in all soil density fractions than the adjacent herbland without trees. Overall, our results highlight the benefits of using *Populus* based silvopasture systems for increasing organic C accumulation in surface mineral soils, particularly in the labile light fraction of SOM, and the ability of *Picea* based shelterbelt systems to enhance the size of stable SOM pools in agricultural lands of central Alberta, Canada.

**Table 4-1.** Bulk soil chemical and physical properties (means with standard errors in parentheses) at the 0–10 cm depth as affected by agroforestry system and land cover type (forested and herbland) in central Alberta, Canada. The interaction effects of agroforestry system by land cover type were not significant for any of the parameters listed in this table, therefore, only the main treatment effects are listed ( $n = 12$ ).

Cover type/ Depth	BD <sup>a</sup> (Mg m <sup>-3</sup> )	NH <sub>4</sub> -N (g m <sup>-2</sup> )	NO <sub>3</sub> -N (g m <sup>-2</sup> )	CEC (cmol kg <sup>-1</sup> )	Clay (%)	Silt (%)	Sand (%)
0-10 cm							
<b>Agroforestry</b>							
Hedgerow	1.19 (0.05)a	2.64 (0.71)a	3.55 (0.57)a	42.30 (2.3)a	26.15 (1.78)a	43.07 (1.98)a	30.78 (2.12)b
Shelterbelt	1.28 (0.04)a	1.87 (0.74)a	4.18 (0.6)a	38.95 (2.3)a	25.87 (1.86)a	45.06 (1.87)a	29.06 (2.12)b
Silvopasture	1.21 (0.04)a	1.21 (0.74)a	0.9 (0.62)b	40.05 (2.42)a	21.64 (1.78)b	41.02 (1.81)a	37.34 (2.19)a
LSD <sub>0.10</sub>	0.92	0.94	7.94	0.54	2.63	2.18	4.31
Prob > F	0.41	0.40	<0.01	0.58	0.08	0.13	0.02
<b>Land cover type</b>							
Forested	1.12 (0.03)b	1.41 (0.61)a	1.1 (0.5)b	45.76 (1.89)a	25.05 (1.47)a	43.43 (1.50)a	31.52 (1.73)a
Herbland	1.33 (0.03)a	2.3 (0.58)a	4.65 (0.47)a	35.11 (1.95)a	24.08 (1.47)a	42.75 (1.59)a	33.17 (1.73)a
LSD <sub>0.10</sub>	13.95	1.34	26.22	15.27	0.41	0.23	0.45
Prob > F	<0.01	0.25	<0.01	<0.01	0.52	0.64	0.50

<sup>a</sup> BD, bulk density; NH<sub>4</sub>-N, ammonium; NO<sub>3</sub>-N, nitrate; CEC, cation exchange capacity; Within a column and cover type, means with the same

letter(s) are not significantly different at  $p < 0.10$  according to the Fisher's protected multiple comparison test (LSD)

**Table 4-2.** Analysis of variance (*F* and *P* values) of the effect of agroforestry system, land cover type (forested and herbland) within agroforest, and their interactions, on SOM density fractions of the 0–10 and 10–30 cm soil layers in central Alberta, Canada<sup>a</sup>.

Soil density fraction/ Depth	Agroforestry system			Land cover type			Agroforestry system x Land cover type		
	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>
<b>0-10 cm</b>									
Light fraction	2	2.12	0.13	1	<b>5.90</b>	<b>0.02</b>	2	0.47	0.62
Occluded fraction	2	0.49	0.61	1	1.64	0.21	2	0.51	0.60
Heavy fraction	2	<b>5.38</b>	<b>0.01</b>	1	<b>13.2</b>	<b>&lt;0.01</b>	2	0.48	0.63
<b>10-30 cm</b>									
Light fraction	2	1.21	0.31	1	2.62	0.11	2	2.01	0.15
Occluded fraction	2	1.12	0.33	1	<b>7.75</b>	<b>&lt;0.01</b>	2	0.33	0.72
Heavy fraction	2	1.06	0.36	1	<b>7.80</b>	<b>&lt;0.01</b>	2	2.18	0.13

<sup>a</sup> A separate analysis was done for each soil layer (0-10 and 10-30 cm); *p* values with less than 10% significance and corresponding *F* values are in bold.

**Table 4-3.** Analysis of variance (*F* and *P* values) of the effect of agroforestry system, land cover type (forested and herbland) within agroforest, and their interaction, on C and N stocks in SOM density fractions, and C/N ratios of the 0–10 and 10–30 cm soil layers in central Alberta, Canada<sup>a</sup>.

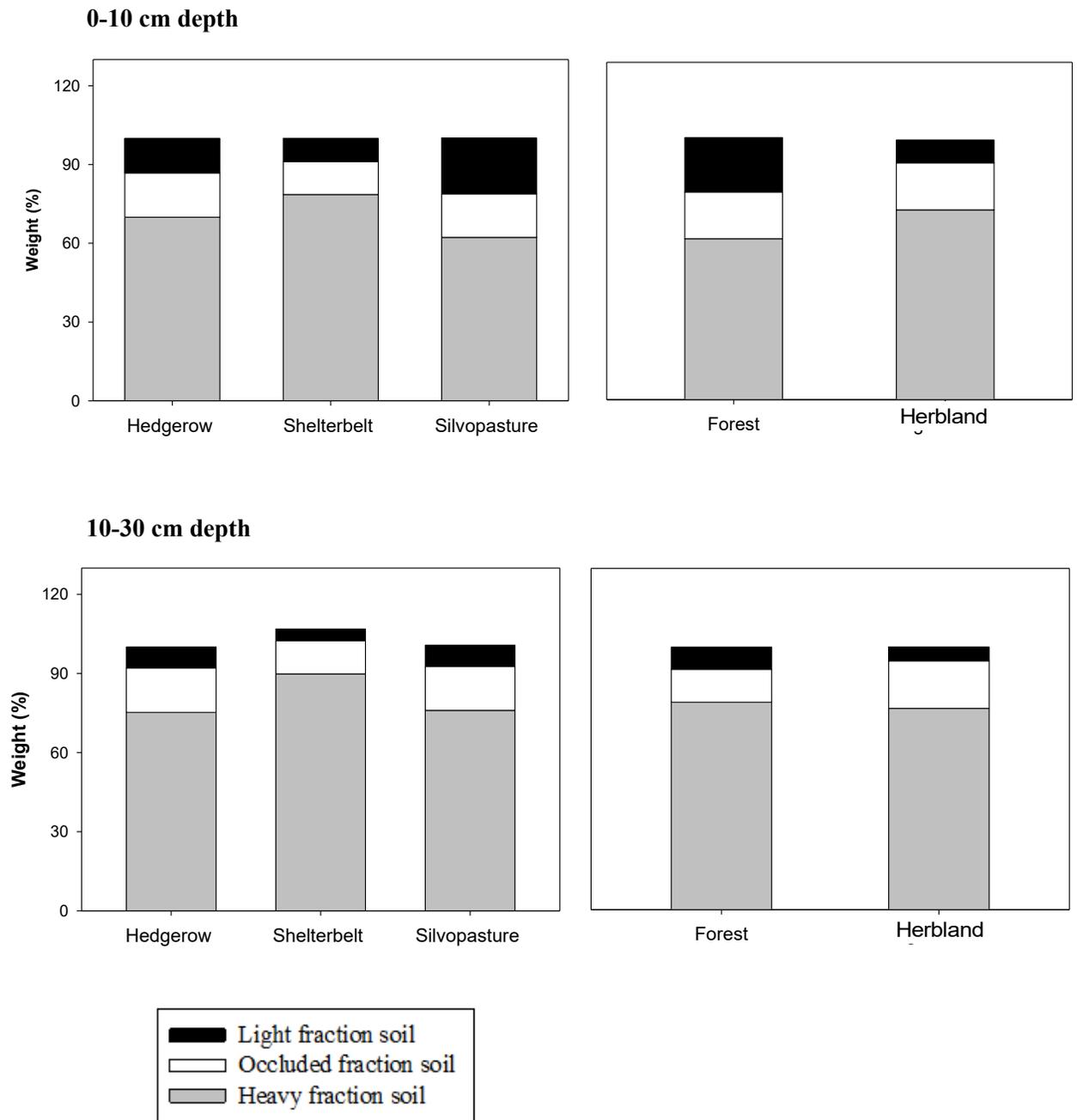
Description/ Depth	Agroforestry system			Land cover type			Agroforestry system x Land cover type		
	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>
0-10 cm									
<b>Soil C (Mg ha<sup>-1</sup>)</b>									
Light fraction	2	<b>2.99</b>	<b>0.08</b>	1	<b>6.80</b>	<b>0.01</b>	2	0.42	0.66
Occluded fraction	2	0.06	0.94	1	<b>3.24</b>	<b>0.08</b>	2	0.11	0.89
Heavy fraction	2	<b>2.77</b>	<b>0.08</b>	1	0.05	0.98	2	1.39	0.26
Bulk soil	2	<b>2.71</b>	<b>0.08</b>	1	<b>4.21</b>	<b>0.05</b>	2	0.38	0.69
<b>Soil N (Mg ha<sup>-1</sup>)</b>									
Light fraction	2	1.29	0.29	1	<b>5.12</b>	<b>0.03</b>	2	0.35	0.71
Occluded fraction	2	0.48	0.62	1	1.97	0.18	2	0.27	0.76
Heavy fraction	2	<b>8.12</b>	<b>&lt;0.01</b>	1	0.10	0.75	2	0.38	0.68
Bulk soil	2	1.61	0.21	1	<b>7.13</b>	<b>0.01</b>	2	0.32	0.73
<b>C/N</b>									
Light fraction	2	0.46	0.63	1	<b>11.3</b>	<b>&lt;0.01</b>	2	0.21	0.81
Occluded fraction	2	1.37	0.28	1	0.05	0.82	2	1.40	0.26
Heavy fraction	2	<b>3.51</b>	<b>0.04</b>	1	0.26	0.61	2	0.40	0.67
Bulk soil	2	<b>17.54</b>	<b>&lt;0.01</b>	1	0.80	0.26	2	0.36	0.71
10-30 cm									
<b>Soil C (Mg ha<sup>-1</sup>)</b>									
Light fraction	2	1.4	0.29	1	<b>2.95</b>	<b>0.09</b>	2	0.68	0.51
Occluded fraction	2	1.9	0.17	1	0.60	0.44	2	1.22	0.31
Heavy fraction	2	<b>5.56</b>	<b>&lt;0.01</b>	1	2.79	0.10	2	0.16	0.83
Bulk soil	2	0.23	0.79	1	<b>6.47</b>	<b>0.01</b>	2	0.58	0.57
<b>Soil N (Mg ha<sup>-1</sup>)</b>									
Light fraction	2	1.79	0.18	1	1.83	0.18	2	0.42	0.42
Occluded fraction	2	<b>3.36</b>	<b>0.05</b>	1	0.33	0.56	2	0.30	0.74
Heavy fraction	2	<b>10.16</b>	<b>&lt;0.01</b>	1	2.19	0.14	2	2.24	0.23
Bulk soil	2	0.91	0.41	1	<b>6.54</b>	<b>0.02</b>	2	0.50	0.61
<b>C/N</b>									
Light fraction	2	<b>7.6</b>	<b>&lt;0.01</b>	1	<b>6.20</b>	<b>0.02</b>	2	1.64	0.21
Occluded fraction	2	<b>9.30</b>	<b>0.01</b>	1	0.09	0.76	2	0.83	0.44
Heavy fraction	2	<b>23.46</b>	<b>&lt;0.01</b>	1	0.05	0.83	2	2.29	0.12
Bulk soil	2	<b>8.12</b>	<b>&lt;0.01</b>	1	2.29	0.14	2	1.94	0.61

<sup>a</sup> A separate analysis was done for each soil layer (0-10 and 10-30 cm); *p* values with less than 10% indicate significance and corresponding *F* values are in bold.

**Table 4-4.** Carbon and N stocks (means with standard errors in parentheses) of various SOM density fractions in the 0–10 and 10–30 cm soil layers as affected by agroforestry system and land cover type (forested and herbland) in central Alberta, Canada ( $n = 12$ )<sup>a</sup>. The interaction effects of agroforestry system by land cover type were not significant for any of the parameters listed in this table, therefore, only the main treatment effects are listed ( $n = 12$ ).

Density fraction	Cover type <sup>b</sup>	C (Mg ha <sup>-1</sup> )		N (Mg ha <sup>-1</sup> )		C/N ratios	
		0-10 cm	10-30 cm	0-10 cm	10-30 cm	0-10 cm	10-30 cm
Light fraction	<b>Agroforestry</b>						
	Hedgerow	31.7 (12)ab	42.1 (10.4)a	2.2 (0.7)a	2.7 (0.7)a	14.4 (0.6)a	15.6 (0.7)b
	Shelterbelt	16.6 (15)b	17.5 (10.4)a	1.1 (0.9)a	1.1 (0.7)a	15.1 (0.7)a	15.9 (0.7)b
	Silvopasture	49.5 (12)a	28.2 (10.1)a	2.9 (0.4)a	1.6 (0.6)a	17.1 (0.6)a	17.6 (0.6)a
	<b>Land cover type</b>						
	Forested	52.6 (10.1)a	39.6 (7.7)a	3.1 (0.6)a	2.3 (0.5)a	17.0 (0.6)a	17.2 (0.5)a
	Herbland	34.8 (11.5)b	18.9 (9.2)b	2.9 (0.9)b	1.3 (0.6)a	12.0 (0.5)b	14.5 (0.6)b
Occluded fraction	<b>Agroforestry</b>						
	Hedgerow	22.4 (2.2)a	33.1 (2.8)a	1.3 (0.1)a	1.8 (0.2)b	17.2 (2.0)a	18.4 (0.5)b
	Shelterbelt	20.7 (2.6)a	37.7 (2.6)a	1.3 (0.2)a	2.2 (0.2)a	16.0 (2.1)a	17.1 (0.6)c
	Silvopasture	20.8 (2.6)a	29.8 (3.2)a	1.2 (0.2)a	1.6 (0.2)b	17.3 (2.2)a	18.6 (0.5)a
	<b>Land cover type</b>						
	Forested	24.1 (1.9)a	34.8 (2.3)a	1.4 (0.1)a	1.9 (0.1)a	17.2 (1.9)a	18.3 (0.4)a
	Herbland	18.7 (2.2)b	32.2 (2.5)a	1.2 (0.1)a	1.8 (0.2)a	15.6 (1.7)a	17.9 (0.4)a
Heavy fraction	<b>Agroforestry</b>						
	Hedgerow	28.4 (1.9)ab	31.1 (3.2)a	2.7 (0.1)b	2.5 (0.2)b	10.5 (1.1)b	12.4 (0.4)a
	Shelterbelt	33.1 (2.1)a	35.1 (3.2)a	4.1 (0.2)a	3.6 (0.2)a	8.1 (1.1)b	9.8 (0.4)b
	Silvopasture	25.9 (2.4)b	20.4 (3.3)b	2.1 (0.2)c	2.2 (0.2)b	12.3 (1.2)a	9.3 (0.4)b
	<b>Land cover type</b>						
	Forested	44.1 (1.7)a	25.8 (2.6)a	3.1 (0.1)a	3.1 (0.2)a	14.2 (0.9)a	8.3 (0.4)a
	Herbland	38.4 (1.6)a	32.2 (2.7)a	2.9 (0.1)a	2.6 (0.2)a	13.2 (0.9)a	12.4 (0.7)a
Bulk soil	<b>Agroforestry</b>						
	Hedgerow	77.2 (7.2)b	101 (15.4)a	6.6 (0.5)a	6.5 (0.9)a	11.7 (0.2)b	15.5 (3.3)a
	Shelterbelt	69.8 (7.2)b	92.7 (16.1)a	6.9 (0.6)a	7.2 (0.9)a	10.1 (0.2)b	12.9 (3.4)b
	Silvopasture	101 (8.6)a	100 (15.4)a	7.9 (0.7)a	7.1 (0.8)a	12.8 (0.3)a	14.1 (3.3)b
	<b>Land cover type</b>						
	Forested	89.6 (6.8)a	119 (12.8)a	7.4 (0.5)a	8.0 (0.7)a	12.1 (0.2)a	14.9 (2.7)a
	Herbland	75.6 (6.9)b	77.4 (12.8)b	6.8 (0.5)b	6.1 (0.7)b	11.1 (0.3)a	12.7 (2.3)a

<sup>a</sup> Each sampling layer (0-10 and 10-30 cm) is reported separately; <sup>b</sup> Within a column and cover type, means with the same letter(s) are not significantly different at  $p < 0.10$  according to the Fisher's protected multiple comparison test (LSD).



**Fig. 4-1.** The percent distribution of light, occluded, and heavy fraction soil separated from bulk soils in the 0-10 and 10-30 cm layers inside three agroforestry systems (hedgerow, shelterbelt, and silvopasture systems) and their component land cover types (forested and herbland) in central Alberta, Canada.

## References

- Adams, B.W., Ehlert, G., Stone, C., Alexander, M., Lawrence, D., Willoughby, D., Hincz, C., Burkinshaw, A., Carlson, J., France, K., 2009. Rangeland Health Assessment for Grassland, Forest and Tame Pasture. Alberta Sustainable Resources Department. Public Lands Division. Edmonton. AB. Pub. No. T/044.
- Albrecht, A., Kandji, S.T., 2003. Carbon sequestration in tropical agroforestry systems. *Agric. Ecosyst. Environ.* 99, 15–27.
- Altieri, M.A., 1999. The ecological role of biodiversity in agroecosystems. *Agric. Ecosyst. Environ.* 74, 19–31.
- Alvarez, R., Alvarez, C.R., 2000. Soil organic matter pools and their associations with carbon mineralization kinetics. *Soil Sci. Soc. Am. J.* 64, 184–189.
- Amonette, J., Kim, J., Russell, C., 2004. Enhancement of soil carbon sequestration: A catalytic approach. *J. Am. Chem. Soc.* 227, U1094–U1094.
- Ayres, E., Steltzer, H., Berg, S., Wallenstein, M.D., Simmons, B.L., Wall, D.H., 2009. Tree species traits influence soil physical, chemical, and biological properties in high elevation forests. *Plos One* 4. e5964. DOI: 10.1371/journal.pone.0005964.
- Baah-Acheamfour, M., Carlyle, C. N., Bork, E. W., Chang, S. X., 2014. Trees increase soil carbon and its stability in three agroforestry systems in central Alberta, Canada. *For. Ecol. Manage.* 328, 131–139.

- Baisden, W. T., Amundson, R., Cook, A. C., Brenner, D. L., 2002. Turnover and storage of C and N in five density fractions from California annual grassland surface soils. *Global Biogeo. Cycles* 16, 64–1.
- Banerjee, S., Baah-Acheamfour, M., Carlyle, C.N., Bissett, A., Richardson, A.E., Siddique, T., Bork, E., Chang, S. (2015). Determinants of bacterial communities in Canadian agroforestry systems. *Environ. Microbiol.* doi: 10.1111/1462-2920.12986.
- Bonde, T.A., Christensen, B.T., Cerri, C.C., 1992. Dynamics of soil organic matter as reflected by natural C-13 abundance in particle-size fractions of forested and cultivated Oxisols. *Soil Biol. Biochem.* 24, 275–277.
- Burke, I.C., Yonker, C.M., Parton, W.J., Cole, C.V., Flach, K., Schimel, D.S., 1989. Texture, climate, and cultivation effects on soil organic matter content in US grassland soils. *Soil Sci. Soc. Am. J.* 53, 800-805.
- Cambardella, C.A., Elliott, E.T., 1994. Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. *Soil Sci. Soc. Am. J.* 58, 123–130.
- Campbell, C. A., Biederbeck, V. O., Wen, G., Zentner, R. P., Schoenau, J., Hahn, D., 1999. Seasonal trends in selected soil biochemical attributes: effects of crop rotation in the semiarid prairie. *Can. J. Soil Sci.* 79, 73–84.
- Carter, M.R., Gregorich, E.G., 2006. *Soil Sampling and Methods of Analysis*, second ed. Francis & Taylor Group, LLC, CRC Press. Boca Raton, FL 334887–2742.

- Chanasyk, D. S., Naeth, M. A., 1995. Grazing impacts on bulk density and soil strength in the foothills fescue grasslands of Alberta, Canada. *Can. J. Soil Sci.* 75, 551–557.
- Chen, Y., Li, Y., Awada, T., Han, J., Luo, Y., 2012. Carbon sequestration in the total and light fraction soil organic matter along a chronosequence in grazing enclosures in a semiarid degraded sandy site in China. *J. Arid Land* 4, 411–419.
- Christensen, B.T., 2000. Levels of structural and functional complexity in soil OM turnover as revealed by physical fractionations. *DIAS Report, Plant Production*, 133–149.
- Creamer, C.A., Filley, T.R., Boutton, T.W., 2013. Long-term incubations of size and density separated soil fractions to inform soil organic carbon decay dynamics. *Soil Biol. Biochem.* 57, 496–503.
- Crow, S.E., Swanston, C.W., Lajtha, K., Brooks, J.R., Keirstead, H., 2007. Density fractionation of forest soils: methodological questions and interpretation of incubation results and turnover time in an ecosystem context. *Biogeochemistry* 85, 69–90.
- Dalal, R.C., Mayer, R.J., 1986. Long-term trends in fertility of soils under continuous cultivation and cereal cropping in southern Queensland .IV. Loss of organic-carbon from different density functions. *Aust. J. Soil Res.* 24, 301–309.
- Don, A., Scholten, T., Schulze, E. D., 2009. Conversion of cropland into grassland: Implications for soil organic carbon stocks in two soils with different texture. *J. Plant Nutr. Soil Sci.* 172, 53–62.

- Donkor, N. T., Gedir, J. V., Hudson, R. J., Bork, E. W., Chanasyk, D. S., Naeth, M. A., 2002. Impacts of grazing systems on soil compaction and pasture production in Alberta. *Can. J. Soil Sci.* 82, 1–8.
- Dorodnikov, M., Kuzyakov, Y., Fangmeier, A., Wiesenberg, G.L.B., 2011. C and N in soil organic matter density fractions under elevated atmospheric CO<sub>2</sub>: Turnover vs. stabilization. *Soil Biol. Biochem.* 43, 579–589.
- Duval, M.E., Galantini, J.A., Iglesias, J.O., Canelo, S., Martinez, J.M., Wall, L., 2013. Analysis of organic fractions as indicators of soil quality under natural and cultivated systems. *Soil Till. Res.* 131, 11–19.
- Environment Canada., 2012. Alberta weather condition [online]  
[http://weather.gc.ca/forecast/canada/index\\_e.html?id=AB](http://weather.gc.ca/forecast/canada/index_e.html?id=AB), [site last visited in June 2014]
- Epstein, H.E., Lauenroth, W.K., Burke, I.C., 1997. Effects of temperature and soil texture on ANPP in the US Great Plains. *Ecology* 78, 2628–2631.
- Gebauer, R. L., Ehleringer, J. R., 2000. Water and nitrogen uptake patterns following moisture pulses in a cold desert community. *Ecology*, 81, 1415–1424
- Grant, C. A., Lafond, G. P., 1993. The effects of tillage systems and crop sequences on soil bulk density and penetration resistance on a clay soil in southern Saskatchewan. *Can. J. Soil Sci.* 73, 223–232.
- Haile, S.G., Nair, P.K.R., Nair, V.D., 2008. Carbon storage of different soil-size fractions in Florida silvopastoral systems. *J. Environ. Qual.* 37, 1789–1797.

- Harris, D., Horwáth, W. R., van Kessel, C., 2001. Acid fumigation of soils to remove carbonates prior to total organic carbon or carbon-13 isotopic analysis. *Soil Sci. Soc. Am. J.* 65, 1853–1856.
- Hassink, J., 1997. The capacity of soils to preserve organic C and N by their association with clay and silt particles. *Plant Soil* 191, 77–87.
- Howlett, D.S., Mosquera-Losada, M.R., Nair, P.K.R., Nair, V.D., Rigueiro-Rodriguez, A., 2011. Soil carbon storage in silvopastoral systems and a treeless pasture in Northwestern Spain. *J. Environ. Qual.* 40, 825–832.
- Huntington, T. G., Johnson, C. E., Johnson, A. H., Siccama, T. G., Ryan, D. F., 1989. Carbon, organic matter, and bulk density relationships in a forested Spodosol. *Soil Sci.* 148, 380–386.
- Isaac, M.E., Harmand, J.M., Lesueur, D. Lelon, J. 2011. Tree age and soil phosphorus conditions influence N<sub>2</sub>-fixation rates and soil N dynamics in natural populations of *Acacia senegal*. *For. Ecol. Manage.* 261: 582–588.
- Jastrow, J. D., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. *Soil Biol. Biochem.* 28, 665–676.
- John, B., Yamashita, T., Ludwig, B., Flessa, H., 2005. Storage of organic carbon in aggregate and density fractions of silty soils under different types of land use. *Geoderma* 128, 63-79.

- Jose, S., 2009. Agroforestry for ecosystem services and environmental benefits: an overview. *Agrofor. Syst.* 76, 1–10.
- Kang, B. T., 1997. Alley cropping—soil productivity and nutrient recycling. *For. Ecol. Manage.* 91, 75–82.
- Kass, D.C.L., Sylvester-Bradley, R., Nygren, P., 1997. The role of nitrogen fixation and nutrient supply in some agroforestry systems of the Americas. *Soil Biol. Biochem.* 29, 775-785.
- Kleber, M., Mertz, C., Zikeli, S., Knicker, H., Jahn, R., 2004. Changes in surface reactivity and organic matter composition of clay subfractions with duration of fertilizer deprivation. *Eur. J. Soil Sci.* 55, 381–391.
- Lafond, G. P., Walley, F., May, W. E., & Holzapfel, C. B., 2011. Long term impact of no-till on soil properties and crop productivity on the Canadian prairies. *Soil Till. Res.* 117, 110–123.
- Lal, R., 2002. Soil carbon dynamics in cropland and rangeland. *Environ. Pollut.* 116, 353-362.
- Lal, R., 2009. Challenges and opportunities in soil organic matter research. *Eur. J. Soil Sci.* 60, 158–169.
- Lenka, N.K., Dass, A., Sudhishri, S., Patnaik, U. S., 2012. Soil carbon sequestration and erosion control potential of hedgerows and grass filter strips in sloping agricultural lands of eastern India. *Agric. Ecosyst. Environ.* 158, 31–40.

- Maia, S.M.F., Xavier, F. A. S., Oliveira, T. S., Mendonça, E. S., Araújo Filho, J. A., 2007. Organic carbon pools in a Luvisol under agroforestry and conventional farming systems in the semi-arid region of Ceará, Brazil. *Agrofor. Syst.* 71, 127–138.
- Malhi, S.S., Nyborg, M., Solberg, E.D., McConkey, B., Dyck, M., Puurveen, D., 2011. Long-term straw management and N fertilizer rate effects on quantity and quality of organic C and N and some chemical properties in two contrasting soils in Western Canada. *Biol. Fert. Soils* 47, 785–800.
- Moni, C., Derrien, D., Hatton, P. J., Zeller, B., Kleber, M., 2012. Density fractions versus size separates: does physical fractionation isolate functional soil compartments? *Biogeosciences* 9, 5181–5197.
- Montagnini, F., Nair, P.K.R., 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. *Agrofor. Syst.* 61, 281–295.
- Mujuru, L., Mureva, A., Velthorst, E.J., Hoosbeek, M.R., 2013. Land use and management effects on soil organic matter fractions in Rhodic Ferralsols and Haplic Arenosols in Bindura and Shamva districts of Zimbabwe. *Geoderma*, 209, 262–272.
- Nair, P.K.R., Nair, V.D., Kumar, B.M., Haile, S.G., 2009. Soil carbon sequestration in tropical agroforestry systems: a feasibility appraisal. *Environ. Sci. Policy* 12, 1099–1111.
- Nascente, A.S., Li, Y.C., Costa Crusciol, C.A., 2013. Cover crops and no-till effects on physical fractions of soil organic matter. *Soil Till. Res.* 130, 52–57.

- Oades, J.M., 1984. Soil organic matter and structural stability: mechanisms and implications for management. *Plant Soil* 76, 319-337.
- Oelbermann, M., Voroney, R.P., Gordon, A.M., 2004. Carbon sequestration in tropical and temperate agroforestry systems: a review with examples from Costa Rica and southern Canada. *Agric. Ecosyst. Environ.* 104, 359–377.
- Partey, S. T., 2011. Effect of pruning frequency and pruning height on the biomass production of *Tithonia diversifolia* (Hemsl) A. Gray. *Agrofor. Syst.* 83, 181–187.
- Paul, E.A., Follett, R.F., Leavitt, S.W., Halvorson, A., Peterson, G.A., Lyon, D.J., 1997. Radiocarbon dating for determination of soil organic matter pool sizes and dynamics. *Soil Sci. Soc. Am. J.* 61, 1058–1067.
- Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K., 2002. Change in soil carbon following afforestation. *For. Ecol. Manage.* 168, 241–257.
- Paustian, K., Six, J., Elliott, E.T., Hunt, H.W., 2000. Management options for reducing CO<sub>2</sub> emissions from agricultural soils. *Biogeochemistry* 48, 147–163.
- Potter, K.N., Daniel, J.A., Altom, W., Torbert, H.A., 2001. Stocking rate effect on soil carbon and nitrogen in degraded soils. *J. Soil Water Conserv.* 56, 233–236.
- Reeder, J.D., Schuman, G.E., 2002. Influence of livestock grazing on C sequestration in a semi-arid mixed-grass and short-grass rangelands. *Environ. Pollut.* 116, 457–463.

- Sainju, U.M., Stevens, W.B., Caesar-TonThat, T., Liebig, M.A., 2012. Soil greenhouse gas emissions affected by irrigation, tillage, crop rotation, and nitrogen fertilization. *J. Environ. Qual.* 41, 1774–1786.
- SAS Institute Inc., 2013. SAS User's Guide: Statistics SAS/C Online Doc, Release 9.30. SAS Inc., Cary, NC
- Six, J., Elliott, E.T., Paustian, K., 2000. Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biol. Biochem.* 32, 2099–2103.
- Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Sci. Soc. Am. J.* 62, 1367–1377.
- Soil Classification Working Group., 1998. The Canadian System of Soil Classification. NRC Research Press, Ottawa, Canada, 187 pp.
- Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma*, 74, 65–105.
- Strosser, E., 2010. Methods for determination of labile soil organic matter: An overview. *J. Agrobiol.* 27, 49–60.
- Swanston, C. W., Myrold, D. D., 1997. Incorporation of nitrogen from decomposing red alder leaves into plants and soil of a recent clear-cut in Oregon. *Can. J. For. Res.* 27, 1496–1502.

- Takimoto, A., Nair, P.K.R., Nair, V.D., 2008. Carbon stock and sequestration potential of traditional and improved agroforestry systems in the West African Sahel. *Agric. Ecosyst. Environ.* 125, 159–166.
- Teklay, T., Chang, S.X., 2008. Temporal changes in soil carbon and nitrogen storage in a hybrid poplar chronosequence in northern Alberta. *Geoderma*, 144, 613–619.
- Tiessen, H., Stewart, J.W.B., 1983. Particle-size fractions and their use in studies of soil organic matter. 2. Cultivation effects on organic matter composition in size fractions. *Soil Sci. Soc. Am. J.* 47, 509–514.
- Tonneijck, F.H., van der Plicht, J., Jansen, B., Verstraten, J.M., Hooghiemstra, H., 2006. Radiocarbon dating of soil organic matter fractions in Andosols in northern Ecuador. *Radiocarbon*, 48, 337.
- von Lützow, M., Kögel-Knabner, I., Ekschmitt, K., Flessa, H., Guggenberger, G., Matzner, E., Marschner, B., 2007. SOM fractionation methods: Relevance to functional pools and to stabilization mechanisms. *Soil Biol. Biochem.* 39, 2183–2207.
- von Lützow, M., Koegel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner, B., Flessa, H., 2006. Stabilization of organic matter in temperate soils: Mechanisms and their relevance under different soil conditions - a review. *Eur. J. Soil Sci.* 57, 426–445.
- Witt, C., Cassman, K.G., Olk, D.C., Biker, U., Liboon, S.P., Samson, M.I., Ottow, J.C.G., 2000. Crop rotation and residue management effects on carbon sequestration, nitrogen cycling and productivity of irrigated rice systems. *Plant and Soil* 225, 263–278.

## Chapter 5: Forest and grassland cover types reduce net greenhouse gas emissions from agricultural soils

### 1. Introduction

Agriculture is a major source of greenhouse gases (GHG) and is key contributor to global climate change. The most recent estimate of GHG emissions from agriculture in 2012 in Canada ranged from 56 (Environment Canada, 2013) to 72 Mt CO<sub>2</sub>-eq (carbon dioxide equivalent) per annum (FAO, 2014). Carbon dioxide accounts for 18.5% of total GHG emissions from Canada's agriculture sector, followed by methane (CH<sub>4</sub>) at 21% and nitrous oxide (N<sub>2</sub>O) at 51.5% (FAO, 2014). However, agriculture can serve as both a source and sink of GHGs, which has led to an increased interest in quantifying the contribution of different land use systems and management practices on GHG emissions. Although opportunities exist to use best management practices (BMPs) to increase carbon sequestration and reduce net GHG emissions from agriculture (Paustian *et al.*, 2000; Lal, 2004; Smith *et al.*, 2008), the knowledge gap on the effects of BMPs on GHG emissions has hampered efforts to mitigate GHG emissions from agriculture.

Emission of GHGs from soils is a result of biochemical processes that are strongly affected by land management practices (Desjardins *et al.*, 2005; Sainju *et al.*, 2012, 2014). Crop and grazing management systems can influence the emission of CO<sub>2</sub> and other GHGs by altering the quantity and quality of plant residue input to the soil (Jensen *et al.*, 2005), varying the input

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of organic and inorganic fertilizers, and changing soil physical properties such as soil aggregation, bulk density and water content (Paustian *et al.*, 2000; Reeder & Shuman, 2002; Lal, 2004). Management practices can also affect emissions by directly changing the soil microenvironment such as temperature and near surface wind speed (Nelson, 2002). Tillage of soil increases CO<sub>2</sub> emission rates because incorporation of crop residues increases their decomposition rate, and soil structure is altered, which increases CO<sub>2</sub> diffusivity and microbial respiration (Gacengo *et al.*, 2009). Tillage has also been found to decrease CH<sub>4</sub> uptake by soils because of the alteration of ecological niches and associated reduction in the population size of CH<sub>4</sub>-oxidizing bacteria (Willison *et al.*, 1995; Sainju *et al.*, 2008). Field measurements have revealed inhibitory effects of nitrogen fertilization on soil CH<sub>4</sub> uptake (Liu *et al.*, 2006) because ammonium is toxic to CH<sub>4</sub>-oxidizing bacteria (Müller *et al.*, 2006). The principal source of N<sub>2</sub>O emissions from agricultural soils is through nitrification and denitrification (Mosier *et al.*, 2006; Butterbach-Bahl *et al.*, 2013), both of which are stimulated by application of nitrogen fertilizers (Ruser *et al.*, 2006; Syakila & Kroeze, 2011; Martins *et al.*, 2015), irrigation (Sainju *et al.*, 2012), tillage (Ball *et al.*, 1999), and the practice of fallowing where no crop is grown (Teepe *et al.*, 2000).

A variety of management options exist for the mitigation of soil GHG emissions from agricultural land use systems (Smith *et al.*, 2012). The most common options include adopting less intensive cropping and grazing systems, increasing cover cropping, reducing summer fallow and tillage, and enhancing fertilizer-use efficiency (Paustian *et al.*, 2000; Cao *et al.*, 2004; Lal, 2004). Land use changes such as the conversion of cropland to perennial pasture and the establishment or retention of woody species within the agricultural landscape in the form of

agroforestry have previously been promoted as management practices capable of offsetting GHG emissions because of their potential to sequester atmospheric carbon in the soil (Mutuo *et al.*, 2005; Breuer *et al.*, 2006; Clary, 2012; Schoeneberger *et al.*, 2012).

Incorporating woody species within the agricultural landscape can directly influence CO<sub>2</sub> emissions by altering the respiration of plant roots and their associated mycorrhizal fungi (Boone *et al.*, 1998). Woody species can indirectly affect GHG emissions by controlling mineralizable carbon and nitrogen contents in soils, and altering soil temperature and water content (Meinzer *et al.*, 2001; Curiel *et al.*, 2007). A decrease in soil temperature slows microbial processes responsible for the mineralization of soil organic matter, and hence the release of GHGs (Evers *et al.*, 2010). Trees lower surface evaporation from the soil, and thus increase soil wetness because of shading and reduced air movement (Green *et al.*, 2003). An increase in soil wetness may enhance soil N<sub>2</sub>O emissions via denitrification (Vargas *et al.*, 2010; Butterbach-Bahl *et al.*, 2013) and favor CH<sub>4</sub> emissions through anaerobic decomposition of soil organic matter (Abdalla *et al.*, 2009). Further, the widespread use of “biological fertilizer” systems such as the inclusion of legumes can enhance the availability of mineralizable nitrogen resulting from the production and decomposition of leguminous biomass (e.g., Albrecht & Kandji, 2003). However, over supply of mineralizable nitrogen in soils could result in the substantial loss of nitrogen in the form of volatilization of ammonia and leaching of nitrate (Mosier *et al.*, 1991). Meanwhile, biological immobilization of mineral nitrogen may occur when C/N ratios in the organic matter undergoing decomposition exceed 30 (Brady & Weil, 2008), indicating that more lignified woody debris could increase microbial nitrogen immobilization and reduce N<sub>2</sub>O emissions by decreasing nitrate availability (Dougherty *et al.*, 2009; Evers *et al.*, 2010; Bergeron *et al.*, 2011). Immobilization can also reduce the inhibitory effect of soil mineral nitrogen on the activity of

CH<sub>4</sub>-oxidising microorganisms, which could lead to an increase in CH<sub>4</sub> uptake in agricultural soils (Mosier *et al.*, 1991).

Several studies have analyzed the benefit of mitigating GHG by establishing or retaining woody species in cropped fields (e.g., Peichl *et al.*, 2006; Wotherspoon *et al.*, 2014), but these efforts have focused on net carbon sequestration. Given the potential of woody species to impact emissions of all three major GHGs, true mitigation can be realized only if the overall effect reduces the net GHG balance from soil relative to adjacent cropped fields without trees. Also, soil CO<sub>2</sub> emission represents the sum of respiration by microbial organisms (heterotrophic respiration) and that from roots (autotrophic respiration). Autotrophic respiration is linked to the consumption of synthesized organic compounds by primary producers themselves, is independent of total carbon levels in the soil, but reflect current vegetation productivity (Horwath *et al.*, 1994; Ohashi *et al.*, 2000). Excluding the amount of CO<sub>2</sub> emitted through autotrophic respiration from the amount of GHG emitted from the soil is important for understanding the potential climate change impact of GHG emissions from the studied soil.

Our main objective was to conduct a large-scale field study to examine how three contrasting agroforestry systems (hedgerow, shelterbelt, and silvopasture) and their component land cover types (forest and herbland areas without trees) affect net GHG balance from soil by evaluating soil CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions. In addition, global warming potential of microbially mediated net GHG emission (GWP<sub>m</sub>) from soils under the various agroforestry systems and their component land cover was derived by excluding root contribution to total CO<sub>2</sub> emission.

## **2. Materials and Methods**

## 2.1. Site description

Soil CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions were measured in a field study established in central Alberta, Canada, at 36 study sites distributed across a 270 km long north-south soil/climate gradient of increasing moisture availability (Appendix 5-1). Sites were located between 54° 43' and 52° 28' N latitude, and between 113° 44' and 113° 17' W longitude, at an average elevation of 692 m asl. Data acquired from 26 Environment Canada weather stations across the study sites indicate that the mean (30-yr) annual temperature was 1.9 and 2.4 °C, and mean annual precipitation was 463 mm and 448 mm, in the north and south of the study area, respectively (Environment Canada, 2015). Precipitation during the 2014 sampling season (April 1 to September 30) was greater than normal (366 mm) with a total of 407 mm, compared with 306 mm in 2013 (Appendix 5-2). Air temperature during the 2013 sampling period was generally higher than that during 2014. Mean air temperature was 13.8 °C in 2013 and 13.2°C in 2014. The study sites represent three soil zones: Gray Luvisols and Dark Gray Chernozems in the north, Black Chernozems and Dark Gray Chernozems in the central zone, and Black Chernozems and Gray Luvisols in the south part of the study area (Soil Classification Working Group, 1998). Physical and chemical properties of soils sampled at the 0-10 cm layer prior to gas measurements at the sites are in Table 5-1.

The study design included three agroforestry systems: hedgerow, shelterbelt, and silvopasture systems, with each system containing two paired land cover types: forest and herbland. Hedgerow systems were made up of naturally regenerating perennial vegetation that included woody species along the edge of annual cropland. Common plant species in the hedgerow included trembling aspen (*Populus tremuloides*), balsam poplar (*Populus*

*balsamifera*), saskatoon berry (*Amelanchier alnifolia*), and chokecherry (*Prunus virginiana*). Shelterbelt systems consisted of trees and shrubs intentionally planted in 1 or 2 parallel rows along the edge of annual cropland. Trees planted in the shelterbelt were typically white spruce (*Picea glauca*) with interspersed caragana (*Caragana arborescens*), willow (*Salix acutifolia*), and box elder (*Acer negundo*). Herblands in the hedgerow and shelterbelt systems were monoculture annual crops and were typically rotated among barley (*Hordeum vulgare*), wheat (*Triticum aestivum*), canola (*Brassica napus*) or pea (*Pisum sativum*) crops. Most of the landowners practice minimum tillage, and apply fertilizers that include nitrogen up to 120 kg ha<sup>-1</sup> annually during spring seeding in early to mid May. The silvopastoral system was established by deliberately grazing existing understory vegetation in native aspen forests to provide alternative forage for livestock, particularly in the late growing season and during low rainfall periods. Trees in this system also provided shelter for livestock. Both the grazed aspen forest and herbaceous agricultural land cover (i.e., open pasture) components in the silvopasture system support livestock grazing in either rotational or season-long grazing systems (Liebig *et al.*, 2005). In this system, the size of forests can vary depending on the amount of space available but can occupy up to 80% of the total land area. Vegetation in grazed aspen forests contained a mixture of trembling aspen, balsam poplar, willow, white birch (*Betula papyrifera*), with the occasional white spruce. Open pastures contain a mix of grasses and forbs, including nitrogen-fixing legumes. We did not observe any fertilization activity in either the grazed aspen forest or open pasture during the experimental period.

## **2.2. Sampling design**

The field study used a split-plot design, with agroforestry system as the whole plot and land cover type (herbland or forest) as the split-plot (i.e., subplot). A total of thirty-six paired sites that included four replicates of each agroforestry system within each of the three soil zones, were sampled in this study. Sites were selected to cover the entire study area given our anticipation that there would be variation in soil and climatic conditions across the 270 km long north-south soil/climate gradient. Two permanent GHG sampling points were established in each subplot. Sampling points in the forested subplots were established in the center of the treed area to minimize edge effects, whereas those in the herbland were located at least one tree height (~ 30 m) from the edge of the forest to avoid the influence of trees. Wire cages, secured to the ground with metal stakes, were used to protect the GHG measurement equipment in the silvopasture system from damage by cattle (*Bos taurus*).

### **2.3. Measurements of total CO<sub>2</sub>,CH<sub>4</sub>, and N<sub>2</sub>O emissions**

Soil GHG emissions were measured monthly at each sampling point from mid-May through September, during each of 2013 and 2014. Gas measurements were made between 10:00 and 15:00 h of the day using static (Plexiglas) chambers (65 cm long × 15 cm wide × 15 cm high); each chamber was driven 6 cm into the soil. Chambers had a beveled lower edge to facilitate insertion and minimize soil compaction. The lid of the chamber was detachable, lined with reflective aluminum foil to maintain ambient air temperatures in the chamber headspace during measurement, had an electric fan (5 mm diameter) to circulate air and ensure thorough gas mixing inside the chamber during sampling, and was fitted with two butyl rubber septum ports for gas sampling. Chambers in the annual cropland component of both the hedgerow and

shelterbelt systems were removed during tillage, planting/seeding, and fertilization operations, and reinstalled near the initial location. Gas samples were collected at 0, 10, 20, and 30 min after closing the chamber using a 20-mL syringe through the sampling port installed on the lid and stored in 12 mL pre-evacuated glass vials (exetainers) fitted with a butyl rubber septa (Labco Ltd., Lampeter, Wales, UK). Concentrations of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O in the gas samples were determined with a gas chromatograph (GC, Model CP-3800, Varian Co., CA). The GC was equipped with a thermal conductivity detector (TCD), a flame ionization detector (FID) and an electron capture detector (ECD) for analyzing CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O concentrations, respectively, on one injection. The temperature of the GC's column oven was set at 50 °C, and the TCD, FID and ECD was controlled at 120, 200 and 320 °C, respectively.

#### **2.4. Determining heterotrophic CO<sub>2</sub> emission rate**

A trenching method was used to exclude root respiration from total soil CO<sub>2</sub> emission (Kelting *et al.*, 1998); this method assumes that root severance around the circumference of plastic soil collars (about 35 cm long) reduces root respiration to negligible levels within three months after trenching. To achieve this, PVC (Schedule SDR35) soil collars 21.2 cm in diameter and 35 cm long were inserted into the soil at each of two permanent sampling points set up within the forest and herbland subplots of each agroforestry system in September, 2012. Soil collars were maintained free of vegetation by spraying with Roundup<sup>®</sup> (Glyphosate, 540 g L<sup>-1</sup>, diluted 10:1 with water) at a monthly application rate of 5 L ha<sup>-1</sup>. In addition, aboveground live vegetation was gently removed by hand from inside the collars at regular intervals but mulch or LFH material was retained. Heterotrophic respiration was measured monthly from mid-May through

September, during each of 2013 and 2014 with a Li-Cor 8100A soil respiration system (LI-8100A, LI-COR Biosciences, Lincoln, Nebraska, USA), equipped with an infrared gas analyzer and a 20 cm inside diameter flow-through chamber. Measurements were made on the same day, and in most cases, within 30 min of static chamber measurements of total CO<sub>2</sub> emissions.

## **2.5. Soil temperature and water content measurements**

Soil temperature and volumetric water content (hereafter referred to as water content) were measured within the 0-10 cm soil layer. Soil temperature and water content were measured inside the static chamber and plastic soil collar with Omega T-handled Type E thermocouples (Model 8100-201, Omega Engineering, Montreal, QC) and time domain reflectometry (TDR) probes (Model Theta Probe ML2X, Delta-T Devices, Cambridge, England), respectively, at the time of each gas sampling. Soil temperature and water content were also monitored continuously in the same depth with 12-Bit Temperature Smart Sensors (Model S-THB-M002, Onset Computer Corporation, Bourne, MA) and 10HS Soil Moisture Smart Sensors (Model S-SMD-MOO5, Onset Computer Corporation, Bourne, MA), respectively, at one of the two sampling points in each subplot. Readings were taken every two hours and recorded on a HOBO Micro Station datalogger (Model H21-002, Onset Computer Corporation, Bourne, MA). One temperature and moisture probe was installed in each of the forest and herbland cover types at each agroforestry site. Soil temperature and water content did not differ inside and outside the chamber ( $r^2 = 0.96$ ; data not shown).

## **2.6. Calculations**

Soil CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O emissions were calculated from the slope of the changes in gas concentrations in the chamber headspace over time. Slope was estimated by fitting two different regression models (linear and quadratic) to the concentration measurements by ordinary least-square regression (Silva *et al.*, 2015). Initial selection of best-fit model(s) for sample sets was done using the  $r^2 > 0.85$  threshold (Lambert & Fréchet, 2005); however, the final model was selected by comparing their respective adjusted  $r^2$  values (Stolk *et al.*, 2009). The amount of CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emitted from each unit of land area per time was calculated following Eqn. [1] (Zhang *et al.*, 2010):

$$Flux = \frac{\Delta C}{\Delta t} \times \frac{mPV}{ART} = H \times \frac{\Delta C}{\Delta t} \times \frac{mP}{RT} \quad [1]$$

where,  $\frac{\Delta C}{\Delta t}$  is the slope from the regression analysis, the value  $m$  is the molecular weight of the trace gases,  $P$  is atmospheric pressure (101.3 kPa),  $R$  is a gas constant (8.314 J mol<sup>-1</sup> K<sup>-1</sup>), and  $T$  is the air temperature in the chamber (°C), while  $V$ ,  $H$ , and  $A$  are the volume, height, and area of ground covered by the static chamber, respectively. The GWP<sub>m</sub> of soil in each subplot was calculated as the sum of CH<sub>4</sub>, and N<sub>2</sub>O fluxes and CO<sub>2</sub> released through heterotrophic respiration by converting each gas concentration to CO<sub>2</sub> equivalent over a 100-yr time scale, using a conversion factor of 1 for CO<sub>2</sub> from heterotrophic respiration, 25 for CH<sub>4</sub>, and 298 for N<sub>2</sub>O (Myhre *et al.*, 2013).

The soil temperature and water content data periodically collected during gas sampling was used to model relationships between GHG emission rates and soil temperature or water content. Weekly averages of the continuously measured environmental data were used to

investigate the effect of the studied land use systems on seasonal variations of soil temperature and water content. Further, we examined the sensitivity of soil CO<sub>2</sub> emissions to temperature changes by fitting a first-order exponential model (Eqn. 2) to mean CO<sub>2</sub> emission for each sampling date to mean values of soil temperatures periodically measured within the 0-10 cm soil layer.

$$Flux = \beta_o \exp^{\beta_1 T} \quad [2]$$

where  $\beta_o$  and  $\beta_1$  are fitted constants and  $T$  is temperature. The temperature sensitivity coefficient  $Q_{10}$  (which denotes the rate of change in CO<sub>2</sub> emissions for each 10 °C change in soil temperature) was calculated using Eqn. [3]:

$$Q_{10} = \exp^{10\beta_1} \quad [3]$$

## 2.7. Statistical analysis

Data on soil CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O emissions, as well as on soil-based GWP, together with soil temperature and water content, were analyzed with repeated measures analysis using PROC MIXED in SAS (SAS Institute Inc., 2014). Agroforestry system and land cover type were treated as fixed effects, sampling date was treated as the repeated measures variable and replication was treated as a random variable. Residual plots and influential plots in the mixed model procedure were evaluated to make sure that the model assumptions of normality and equal variance were

satisfied. Data on CH<sub>4</sub> emissions collected over time were highly skewed to the right; therefore these data were log-transformed before further analysis; however, untransformed data are presented for ease of interpretation. Data on all other parameters did not require transformation. We used LS-means in ANOVA, and subsequently carried out a Fisher's protected test to identify differences between treatment means and treatment-by-sampling date effects when ANOVA indicated significant treatment effects.

In a study of this nature where gas samples were collected across a large geographic area (along a 270 km long soil/climate gradient) that had substantial variation in soil properties and vegetation composition, the risk of a Type II error could be high even if sample size was large. Given our concern with committing a Type II error, all tests with  $p < 0.10$  were considered significant.

### **3. Results**

#### **3.1. Soil temperature and water content**

Soil temperature and water content in the 0-10 cm layer varied throughout the growing season in 2013 and 2014 and were markedly affected by the various land uses. In both 2013 and 2014, soil temperature varied significantly among agroforestry systems, land cover types, sampling dates, as well as most interactions (Table 5-2). Within each agroforestry system, herbland areas, particularly the croplands, had greater soil temperature as compared to areas under trees during 2013 (Fig. 5-1a). These differences were larger from mid-May through July but diminished thereafter. In summer 2014, soil temperature in the silvopasture was on average 2.9 °C greater

than in the hedgerow and shelterbelt systems (Fig. 5-1b), and was lower in forest than in herbland soils before mid-August (Fig. 5-1c).

In both 2013 and 2014, soil water content was affected by most treatments and their interactions (Table 5-2). Overall, the herbland and forest cover types within the shelterbelt system showed higher soil water content in 2013 than the other treatment combinations examined during the early and later part of the sampling period, but the opposite occurred in mid-summer (Fig. 5-1d). In 2014, the shelterbelt system again showed greater soil water content than hedgerow and silvopasture systems, particularly from July through September (Fig. 5-1e), and forest land covers consistently had lower soil water content than their herbland counterparts across all agroforestry systems (Fig. 5-1f). Overall, there were greater amplitudes of soil temperature and water content in annual cropland than in open pasture and forest land covers.

### **3.2. Soil CO<sub>2</sub> emissions**

In 2013, most soil CO<sub>2</sub> emissions occurred from June to August (81%), with the peak in July at 62.9 kg C ha<sup>-1</sup> d<sup>-1</sup> (data not shown), coincident with a period of high soil temperature and management practices like tillage and grazing. Total soil CO<sub>2</sub> emissions varied significantly among agroforestry systems, sampling dates, and the interaction between agroforestry system and land cover type (Table 5-3). The open pasture areas in the silvopasture system had CO<sub>2</sub> emissions similar to that of forests in the hedgerow, but were generally greater than emissions in the other land uses (Fig. 5-2a). In 2014, soil CO<sub>2</sub> emissions varied with agroforestry system and land cover type, and both of which also interacted with sampling date (Table 5-3). Soil CO<sub>2</sub> emissions were different among agroforestry systems only in June, July, and August (Fig. 5-2b).

In June, hedgerows had 38 and 25% greater CO<sub>2</sub> emissions than shelterbelt and silvopasture systems, respectively. In July through August, emissions of soil CO<sub>2</sub> were more than 30% greater in the silvopasture than in the other two systems. Comparative emissions of CO<sub>2</sub> between land cover types were different only in June and July (Fig. 5-2c), which was about 30% lower in forest than in herbland in June, with the opposite pattern in July.

### **3.3. Soil CH<sub>4</sub> uptake**

Soil CH<sub>4</sub> production was affected by both agroforestry system and land cover type in 2013 and 2014, with both main effects interacting with sampling date in 2014 (Table 5-3). Negative values in most sampling dates suggested that there was an overall net CH<sub>4</sub> uptake by soils. The CH<sub>4</sub> uptake increased from 2.1 g C ha<sup>-1</sup> d<sup>-1</sup> in May to 9.2 g C ha<sup>-1</sup> d<sup>-1</sup> in September during 2013 within all land use systems (data not shown). Methane was consumed more rapidly by soils in the silvopasture than in the other two systems (Fig. 5-3a), particularly in August and September of 2014 (Fig. 5-3c), while forest land cover across all agroforestry system had greater CH<sub>4</sub> uptake compared to herblands (Figs. 5-3b). Soil CH<sub>4</sub> uptake in 2014 had a similar pattern to that in 2013, with the exception that CH<sub>4</sub> uptake across all land use systems declined from 9.1 g C ha<sup>-1</sup> d<sup>-1</sup> in August to 6.8 in September immediately after a heavy precipitation (> 15 mm) event. CH<sub>4</sub> uptake was different among agroforestry systems from August through September (Fig. 5-3c), which was 48% greater in silvopasture than the other two systems. Mean CH<sub>4</sub> uptake was 59% lower in herbland than in forested areas in 2014 (Fig.5-3d).

### **3.4. Soil N<sub>2</sub>O emissions**

Most land use systems exhibited strong seasonal patterns in soil N<sub>2</sub>O emission, with the largest occurring during early to midsummer. In both years, soil N<sub>2</sub>O emission was affected by agroforestry system, land cover type, sampling date, and interactions between sampling date and agroforestry system or land cover type (Table 5-3). Soil N<sub>2</sub>O emission was different among the three agroforestry systems from May to July in 2013; emissions in the silvopasture were almost the same as the shelterbelt, but was 84% lower than that in the hedgerow system (Fig. 5-4a). The effect of land cover type on N<sub>2</sub>O emission in 2013 was evident from June through July, with 94% greater N<sub>2</sub>O emission in herbland compared to forest (Fig. 5-4b). Soil N<sub>2</sub>O emissions were minimal from August to September and were not affected by agroforestry system or land cover type. In 2014, 77% of soil N<sub>2</sub>O emissions occurred from June through July, with 38% greater N<sub>2</sub>O emissions in the shelterbelt than in the other two systems (Figs. 5-4c), and 120% greater N<sub>2</sub>O emissions in herbland than in forested areas (Fig. 5-4d).

### **3.5. Global warming potential of microbe-mediated soil GHG emission (GWP<sub>m</sub>)**

Over the two seasons and across the sites, the CO<sub>2</sub> emitted through heterotrophic respiration from soils was 5% (on average) greater in hedgerow and shelterbelt than in the silvopasture system (Fig. 5-5a), and was also different between the forest and herbland cover types, with emissions 14% lower in the forest than in the herbland (Fig. 5-5b). Combining the CO<sub>2</sub> equivalents of soil CH<sub>4</sub> and N<sub>2</sub>O fluxes with the CO<sub>2</sub> emitted via heterotrophic respiration, the silvopasture system had 14% and 9% lower GWP<sub>m</sub> than the hedgerow and shelterbelt systems,

respectively (Fig. 5-5a). A similar effect of land cover type was observed in both years, with a lower  $GWP_m$  in forest (68) compared to adjacent herbland (89 kg CO<sub>2</sub> ha<sup>-1</sup>) (Fig. 5-5b).

### **3.6. Environmental controls of GHG emissions**

Emissions of CO<sub>2</sub> were more affected by soil temperature than water content (Figs. 5-6a, d). Soil CO<sub>2</sub> emissions increased with soil temperature, which explained at least 52% of the variation in both years. While the relationship between soil-based CO<sub>2</sub> emission and water content was significant only in 2014 ( $p = 0.07$ ), this relationship ( $r^2 = 0.13$ ) remained much weaker than that of soil temperature ( $r^2 = 0.54$ ). Sensitivity of soil CO<sub>2</sub> emission to temperature change was affected by land cover type and its interaction with agroforestry system only in 2013 (Table 5-4). The  $Q_{10}$  values were greater (indicating higher temperature sensitivity) in annual croplands than neighboring hedgerow (5.2) and shelterbelt forests (5.1), with no difference in the silvopasture system (Fig. 5-7). Across all land use systems, CH<sub>4</sub> uptake was negatively related ( $r^2 = 0.38$ ;  $p < 0.01$ ) to soil water content in both years (Fig. 5-6b, e), with no significant relationship with soil temperature, while N<sub>2</sub>O emissions were positively related to both soil temperature ( $r^2 = 0.28$ ;  $p < 0.05$ ) and water content ( $r^2 = 0.43$ ;  $p < 0.01$ ) (Fig. 5-6c, f).

## **4. Discussion**

### **4.1. Total CO<sub>2</sub> emission from soils**

The silvopasture system contributed more towards total microbial CO<sub>2</sub> emission (excluding autotrophic respiration) than the other two agroforestry systems because grazed aspen forest and

open pasture, both permanent cover types in silvopasture, were found to emit more CO<sub>2</sub> than annual croplands. The continuous presence of live vegetation during the active growth period increased CO<sub>2</sub> emission in both the grazed aspen forest and open pasture. The greater concentrations of organic carbon and dissolve organic carbon in the near surface (0-10 cm) soils of the silvopasture (Table 5-1) suggest this system has more carbon in the soil despite having a greater CO<sub>2</sub> emission. Peichl *et al.* (2006) and Wotherspoon *et al.* (2014) did find a similar result that showed greater soil CO<sub>2</sub> emission in areas under trees than under annual crop in an alley cropping system in southern Ontario, Canada. They found that net CO<sub>2</sub> emission rate was always the greatest closest to the tree row at 0 m (28.2 to 20.5 g CO<sub>2</sub> m<sup>-2</sup> day<sup>-1</sup>), and the lowest at 6 m (12.9 to 16.9 g CO<sub>2</sub> m<sup>-2</sup> day<sup>-1</sup>) into the cropped portion of the system. Janssens *et al.* (2001) found that the effect of vegetation cover and productivity in controlling soil CO<sub>2</sub> emission to be greater than that of soil temperature in temperate ecosystems. Banerjee *et al.* (2015) reported that soils under the forest cover type across the present study sites contained more microbial diversity and biomass than their herbland counterparts. If these forest cover types promote microbial populations in agroforestry systems, and have more litter and root exudate input into soil, then microbial activities would be greater, and this would be reflected in higher CO<sub>2</sub> emissions in the forest land cover type.

The temperature sensitivity of soil CO<sub>2</sub> emission in annual cropland was greater than that in forest and open pasture soils, but was not different among the agroforestry systems. The greater temperature sensitivity in annual cropland is consistent with Conant *et al.* (2008) and Arevalo *et al.* (2010). It is likely that increased soil temperature in annual croplands due to cultivation and low vegetation cover led to the loss of carbon from cropped fields in the study region. Further, the widespread conversion of forest and open pasture to annual cropland in many

areas across the globe (Breuer *et al.*, 2006; Clary, 2012) suggests a large and ongoing ecosystem carbon loss. The higher temperature sensitivity for annual crop is also concerning when one considers that the climate is expected to be warmer in the future, as this could create a positive feedback (Boone *et al.*, 1998). Differences in the temperature sensitivity of soil CO<sub>2</sub> emission among the various land uses also suggest modification in soil C pools could occur due to woody species establishment or maintenance within agricultural landscapes. Fang *et al.* (2006) indicated the retention of more labile forms of soil organic carbon could increase the sensitivity of CO<sub>2</sub> emissions to changes in temperature; however, others suggest that the decomposition of recalcitrant organic matter is more temperature sensitive than labile forms (Agren, 2000; Agren & Bosatta, 2002; Fierer *et al.*, 2005; Knorr *et al.*, 2005; Davidson & Jannsens, 2006). Analysis of data from the 36 sites in a soil physical fractionation study showed that forest and perennial pasture land cover types were able to accumulate and store larger amounts of labile carbon than annual cropland (Baah-Acheamfour *et al.*, 2014, 2015).

Carbon quality may not be the sole factor influencing the temperature response of soil CO<sub>2</sub> emissions in agroecosystems (Janssens *et al.*, 2004). Field-based estimates of temperature sensitivity are not only related to temperature responses, but may also be affected by complex interactions between temperature and a range of other factors, such as soil water content, the availability of carbon and nitrogen substrates, and microbial population size (Davidson *et al.*, 1998; Davidson & Jannsens, 2006). In this study, the variability of changes in soil temperature and water content was more pronounced in annual cropland. Such changes are known to simulate the response of CO<sub>2</sub> emission to temperature changes (Pang *et al.*, 2013). There are other possible mechanisms influencing temperature response of soil CO<sub>2</sub> emissions, such as the degree of physical protection of soil carbon. Lack of soil disturbance in both forest and open pasture due

to the absence of tillage could cause organic carbon to become incorporated into soil aggregates, eventually leading to carbon protected by soil minerals from degradation (Six *et al.*, 2000).

Aggregate-protected soil carbon is reported to respond to temperature changes more slowly than non-aggregate-protected soil carbon (Jastrow *et al.*, 1996).

#### **4.2. Soil CH<sub>4</sub> uptake**

Soils across the study sites had a net CH<sub>4</sub> emission (i.e., uptake); the largest CH<sub>4</sub> uptake was reported in midsummer, which is not surprising given that methanotrophs oxidize more CH<sub>4</sub> in drier soils (Mosier *et al.*, 2006). Soil water content accounted for about 38% of the variability in CH<sub>4</sub> emissions across study sites. As soil water content and water-filled pore space decreased in midsummer, more CH<sub>4</sub> was consumed by methanotrophs (Van den Pol-van Dasselaar *et al.*, 1998). While methanotrophy can occur anaerobically (Smemo & Yavitt, 2011), CH<sub>4</sub> oxidizing communities in oxic surface soil layers are thought to consume a significant proportion of CH<sub>4</sub> produced at depth (Raghoebarsing *et al.*, 2005). Methanogens and methanotrophs co-exist in the same area in soils and the net balance between these two microbial groups changes depending on soil water content (Mu *et al.*, 2013). Other studies (e.g., MacDonald *et al.*, 1997; Van den Pol-van Dasselaar *et al.*, 1998; Gacengo *et al.*, 2009), however, have shown increasing soil CH<sub>4</sub> uptake rates not only with lower soil water content, but also with decreasing soil bulk density and increasing sand content. Results suggest CH<sub>4</sub> diffusivity into the soil could be a primary constraint on CH<sub>4</sub> oxidation rates (Christiansen & Gundersen, 2011)

Methane uptake was greater in the silvopasture compared to the hedgerow and shelterbelt systems, as well as in forests than in adjacent herblands. Any increased CH<sub>4</sub> uptake in

silvopasture relative to the other agroforestry systems could come from lower soil disturbance and greater macroporosity (e.g., MacDonald *et al.*, 1997; Van den Pol-van Dasselaar *et al.*, 1998; Gacengo *et al.*, 2009) in soils under silvopasture, as reflected in the reduced soil bulk density and greater sand concentration. Since CH<sub>4</sub> diffusivity into the soil is a primary constraint on CH<sub>4</sub> oxidation rates (Christiansen & Gundersen, 2011), it is likely that the lower soil CH<sub>4</sub> uptake in herbland than in forested land was mainly a product of soil cultivation — causing a reduction in pore continuity due to tillage. Other studies have shown the activity of CH<sub>4</sub>-oxidizing bacteria is reduced due to high levels of soil mineral nitrogen, which limit the activity of CH<sub>4</sub>-oxidizing bacteria (Sainju *et al.*, 2014; Liu *et al.*, 2006; Müller *et al.*, 2006). Methane uptake rates in the hedgerow and shelterbelt systems may have been further reduced by fertilizer applied to the monoculture annual crops, which included up to 120 kg ha<sup>-1</sup> y<sup>-1</sup> of nitrogen (AARD, 2004). While we did not observe any fertilization application in silvopasture systems, they contained larger pools of mineral nitrogen during the early (Baah-Acheamfour *et al.*, 2015) and late growing seasons (Baah-Acheamfour *et al.*, 2014) compared to the other agroforestry systems, a result attributed to the abundance of nitrogen-fixing legumes and inputs from cattle (Reeder & Shuman, 2002). Despite the high mineral nitrogen concentrations in silvopasture soils, we did not observe particularly low CH<sub>4</sub> uptake, perhaps because the state of the soil system (including high temperatures, low water levels, and low bulk density) muted the inhibitory effect of mineral nitrogen concentration on the activity of CH<sub>4</sub>-oxidizing bacteria.

#### **4.3. Soil N<sub>2</sub>O emission**

Soil N<sub>2</sub>O emissions exhibited strong seasonal patterns, with the largest occurring during early to midsummer (June through July). However, the accumulated N<sub>2</sub>O emission from early to late May was low in both years, though increased N<sub>2</sub>O emission in cold, thawing soils has been reported across the study area (Nyborg *et al.*, 1997) and other cold, thawing soils in Canada (Amadi *et al.*, 2016). A large amount of seasonal N<sub>2</sub>O emissions (> 65%) can occur over time scales of hours to weeks in response to management practices (e.g., fertilization, tillage, irrigation) and climatic events (e.g., precipitation, soil thawing) (Venterea *et al.* 2012). Our sampling frequency may have been too low to adequately capture all the large pulse events of GHG emissions during the sampling period (Reeves *et al.*, 2015). Moreover, we could not use a more frequent sampling schedule because of the large geographic area (approximately 60,000 km<sup>2</sup>) covered by the sample sites.

Overall, the hedgerow and shelterbelt systems represented a significant source of soil N<sub>2</sub>O in the middle of the growing season (June and July) in both years. Enhanced N<sub>2</sub>O emissions from June through July coincided with crop production (including fertilizer application) in the annual crop components of hedgerow and shelterbelt systems. Tillage has also been reported to increase soil organic nitrogen mineralization and nitrification, and thus the high nitrate content and higher daily N<sub>2</sub>O emissions (Ball *et al.*, 1999), especially when combined with nitrogen fertilization. It is also possible that temperature-induced increases in soil respiration during this period (June and July) reduced soil oxygen concentrations, which in turn leading to an increase in soil anaerobic conditions. Denitrification of nitrate under anaerobic conditions enhances N<sub>2</sub>O production and could explain why emissions were high during periods of high temperature (Butterbach-Bahl *et al.*, 2013). Emission of soil N<sub>2</sub>O was reduced under silvopasture; N<sub>2</sub>O emissions may be lower in this system because application of nitrogen-based fertilizer in the

forest and open pasture components may be equally low compared to intensively fertilized annual crops within the hedgerow and shelterbelt systems.

#### **4.4. Global warming potential of microbe-mediated soil GHG emission (GWP<sub>m</sub>)**

The GWP<sub>m</sub> should be considered relative to only the net GHG emissions from the soil through microbial activity. Emissions of soil-based CO<sub>2</sub> via heterotrophic respiration had the greatest impact on GWP<sub>m</sub> because CO<sub>2</sub> equivalents from both CH<sub>4</sub> and N<sub>2</sub>O were small. The contributions of CH<sub>4</sub> to total GWP<sub>m</sub> were less than 1%, whereas the contribution of N<sub>2</sub>O was up to 15%. This result suggests that avoiding microbially mediated CO<sub>2</sub> losses associated with cultivation are key to increasing C storage in agroforestry soils for mitigating global warming (Paustian *et al.*, 2000; Lal, 2004; Six *et al.*, 2004; Smith *et al.*, 2008). It was estimated that the silvopasture system had a lower GWP<sub>m</sub> than the other two agroforestry systems, which was largely due to the reduced N<sub>2</sub>O emission and a greater amount of CH<sub>4</sub> uptake in the silvopasture system. The GWP<sub>m</sub> was greater for the hermland than for the forest land, suggesting that soil cultivation in the former not only enhanced GHG emissions but also increased the climate change impact of the ecosystem. In fact, effects of soil cultivation and other management activities in hermland may persist beyond the growing season, which could cause large GHG losses via other processes such as erosion and leaching.

## **5. Conclusions**

The results suggest general control on GHG emissions by soil temperature and water content, but also show that these relationships are modified by land use type. Within agroforestry systems, increased CH<sub>4</sub> uptake and reduced N<sub>2</sub>O emission in areas under trees may reduce the net balance of GHG emissions, and lower GWP as compared to agricultural systems without any trees. Thus, GHG accounting in agroforestry systems need to consider fluxes of CH<sub>4</sub> and N<sub>2</sub>O together with carbon sequestration in order to realize the full GHG mitigation potential of woody species in agro-ecosystems. The silvopasture system contributed less to soil-based GWP than the other two agroforestry systems, which reflects reduced N<sub>2</sub>O emission and a greater amount of CH<sub>4</sub> uptake in both natural forests and adjacent open pasture. Planting or retaining woody species in the agricultural landscape can be an effective measure to reduce soil GHG emissions and mitigate climate change. Global warming potential as calculated in this study did not integrate CO<sub>2</sub> uptake by vegetation or changes in soil organic carbon during the sampling period; further studies in these systems would be necessary to more fully understand the climate change impact in a whole-ecosystem context

**Table 5-1.** Chemical and physical properties of soils (means with standard errors in parentheses) at the 0–10 cm depth under agroforestry systems and land cover types (forest and herbland) across study sites.

Cover type	pH	BD <sup>a</sup> (Mg m <sup>-3</sup> )	NH <sub>4</sub> -N (g m <sup>-2</sup> )	NO <sub>3</sub> -N (g m <sup>-2</sup> )	CEC (cmol kg <sup>-1</sup> )	TOC (Mg ha <sup>-1</sup> )	TN (Mg ha <sup>-1</sup> )	DOC (mg kg <sup>-1</sup> )	DON (mg kg <sup>-1</sup> )	Clay (%)	Silt (%)	Sand (%)
<b>Agroforestry system</b>												
	5.40 (0.12)	1.24 (0.05)	2.64 (0.71)	3.55 (0.57)	42.30 (2.23)	31.70 (12)	2.20 (0.71)	2.99 (0.06)	0.68 (0.06)	26.15 (1.78)	43.07 (1.98)	30.78 (2.12)
Hedgerow	5.90 (0.13)	1.28 (0.04)	1.87 (0.71)	4.18 (0.60)	38.95 (2.23)	16.60 (15)	1.10 (0.90)	3.68 (0.13)	0.85 (0.06)	25.87 (1.86)	45.06 (1.87)	29.06 (2.14)
Shelterbelt	5.75 (0.12)	1.19 (0.05)	1.21 (0.74)	0.9 (0.62)	40.05 (2.24)	49.5 (12)	2.90 (0.40)	3.74 (0.25)	0.80 (0.04)	21.64 (1.79)	41.02 (1.81)	37.34 (2.19)
Silvopasture												
<b>Land cover type</b>												
	5.81 (0.12)	1.12 (0.03)	1.41 (0.61)	1.10 (0.51)	45.76 (1.89)	52.6 (10)	3.10 (0.62)	4.88 (0.08)	0.84 (0.04)	25.05 (1.48)	43.43 (1.50)	31.52 (1.73)
Forest	5.56 (0.10)	1.33 (0.03)	2.30 (0.58)	4.65 (0.40)	35.11 (1.95)	34.8 (11)	2.90 (0.94)	2.04 (0.58)	0.73 (0.03)	24.08 (1.47)	42.75 (1.59)	33.17 (1.73)
Herbland												

<sup>a</sup> BD, bulk density; NH<sub>4</sub>-N, ammonium; NO<sub>3</sub>-N, nitrate; CEC, cation exchange capacity; TOC, total organic carbon; TN, total nitrogen; DON, dissolved organic carbon; DON, dissolved organic nitrogen.

**Table 5-2.** Analysis of variance (*F* and *P* values) of the effect of agroforestry system, land cover type (forest and herbland) within each agroforestry system, and their interaction, on soil temperature and soil water content (0-10 cm depth) from May to September in each of 2013 and 2014 <sup>a</sup>.

Factor	2013			2014		
	<i>Df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>
<b>Soil temperature (°C )</b>						
Agroforestry (AF)	2	<b>21.65</b>	<b>&lt;0.01</b>	2	1.26	0.28
Land cover type (L)	1	<b>530.99</b>	<b>&lt;0.01</b>	1	<b>74.48</b>	<b>&lt;0.01</b>
AF x L	2	<b>41.16</b>	<b>&lt;0.01</b>	2	<b>8.25</b>	<b>&lt;0.01</b>
Date of sampling (D)	19	<b>138.6</b>	<b>&lt;0.01</b>	19	<b>56.44</b>	<b>&lt;0.01</b>
AF x D	38	<b>2.31</b>	<b>0.07</b>	32	<b>2.33</b>	<b>0.07</b>
L x D	19	<b>6.34</b>	<b>&lt;0.01</b>	16	<b>3.28</b>	<b>&lt;0.01</b>
AF x L x D	28	<b>1.92</b>	<b>&lt;0.01</b>	31	0.64	0.93
<b>Soil water content (cm<sup>3</sup>/cm<sup>3</sup>)</b>						
Agroforestry (AF)	2	<b>10.21</b>	<b>&lt;0.01</b>	2	<b>37.46</b>	<b>&lt;0.01</b>
Land cover type (L)	1	0.87	0.35	1	<b>125.16</b>	<b>&lt;0.01</b>
AF x L	2	<b>3.03</b>	<b>0.04</b>	2	1.48	0.26
Date of sampling (D)	19	<b>10.1</b>	<b>&lt;0.01</b>	19	<b>9.34</b>	<b>&lt;0.01</b>
AF x D	38	<b>9.62</b>	<b>&lt;0.01</b>	32	<b>10.15</b>	<b>&lt;0.01</b>
L x D	19	<b>8.28</b>	<b>&lt;0.01</b>	16	<b>7.24</b>	<b>&lt;0.01</b>
AF x L x D	28	<b>11.04</b>	<b>&lt;0.01</b>	31	0.85	0.69

<sup>a</sup> A separate analysis was done for each sampling year (2013 and 2014); F-stats with *p* values < 0.10 indicate significance and are shown in bold.

**Table 5-3.** Analysis of variance (*F* and *P* values) of the effect of agroforestry system, land cover type (forest and herbland) within each agroforestry system, and their interaction, on seasonal changes in soil greenhouse gas emissions from May to September in each of 2013 and 2014<sup>a</sup>.

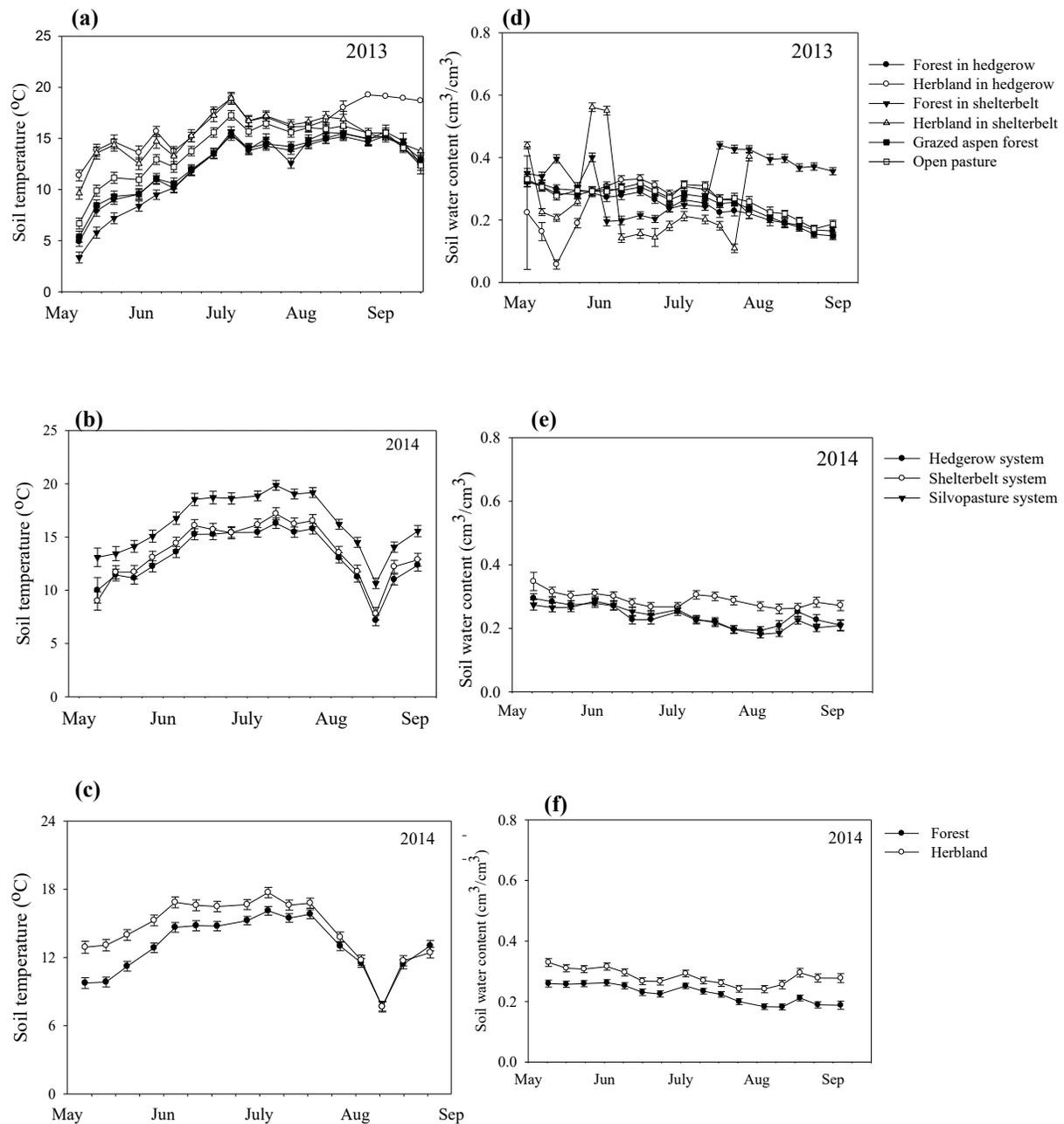
Factor	2013			2014		
	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>
<b>CO<sub>2</sub> flux (kg C ha<sup>-1</sup> d<sup>-1</sup>)</b>						
Agroforestry (AF)	2	<b>5.35</b>	<b>&lt;0.01</b>	2	<b>6.93</b>	<b>&lt;0.01</b>
Land cover type (L)	1	0.01	0.99	1	<b>6.32</b>	<b>0.01</b>
AF x L	2	<b>4.74</b>	<b>&lt;0.01</b>	2	0.32	0.72
Date of sampling (D)	4	<b>81.5</b>	<b>&lt;0.01</b>	4	<b>104.12</b>	<b>&lt;0.01</b>
AF x D	8	0.68	0.71	8	<b>1.92</b>	<b>0.05</b>
L x D	4	0.68	0.61	4	<b>11.95</b>	<b>&lt;0.01</b>
AF x L x D	8	1.36	0.21	8	0.77	0.62
<b>CH<sub>4</sub> flux (g C ha<sup>-1</sup> d<sup>-1</sup>)</b>						
Agroforestry (AF)	2	<b>5.56</b>	<b>&lt;0.01</b>	2	<b>9.17</b>	<b>&lt;0.01</b>
Land cover type (L)	1	<b>49.17</b>	<b>&lt;0.01</b>	1	<b>86.26</b>	<b>&lt;0.01</b>
AF x L	2	1.94	0.14	2	0.8	0.44
Date of sampling (D)	4	<b>3.35</b>	<b>&lt;0.01</b>	4	<b>2.36</b>	<b>&lt;0.01</b>
AF x D	8	1.1	0.60	8	<b>1.76</b>	<b>0.08</b>
L x D	4	0.69	0.60	4	<b>4.42</b>	<b>&lt;0.01</b>
AF x L x D	8	0.39	0.92	8	0.73	0.66
<b>N<sub>2</sub>O flux (g N ha<sup>-1</sup> d<sup>-1</sup>)</b>						
Agroforestry (AF)	2	<b>3.25</b>	<b>0.04</b>	2	<b>3.57</b>	<b>0.03</b>
Land cover type (L)	1	<b>8.6</b>	<b>0.03</b>	1	<b>6.82</b>	<b>0.09</b>
AF x L	2	0.78	0.46	2	1.28	0.27
Date of sampling (D)	4	<b>4.19</b>	<b>0.03</b>	4	<b>2.54</b>	<b>0.04</b>
AF x D	8	<b>2.65</b>	<b>0.07</b>	8	<b>1.82</b>	<b>0.09</b>
L x D	4	<b>4.04</b>	<b>&lt;0.01</b>	4	<b>3.31</b>	<b>0.01</b>
AF x L x D	8	0.76	0.63	8	1.62	0.12

<sup>a</sup> A separate analysis was done for each sampling year (2013 and 2014); F-stats with *p* values < 0.10 indicate significance and are shown in bold.

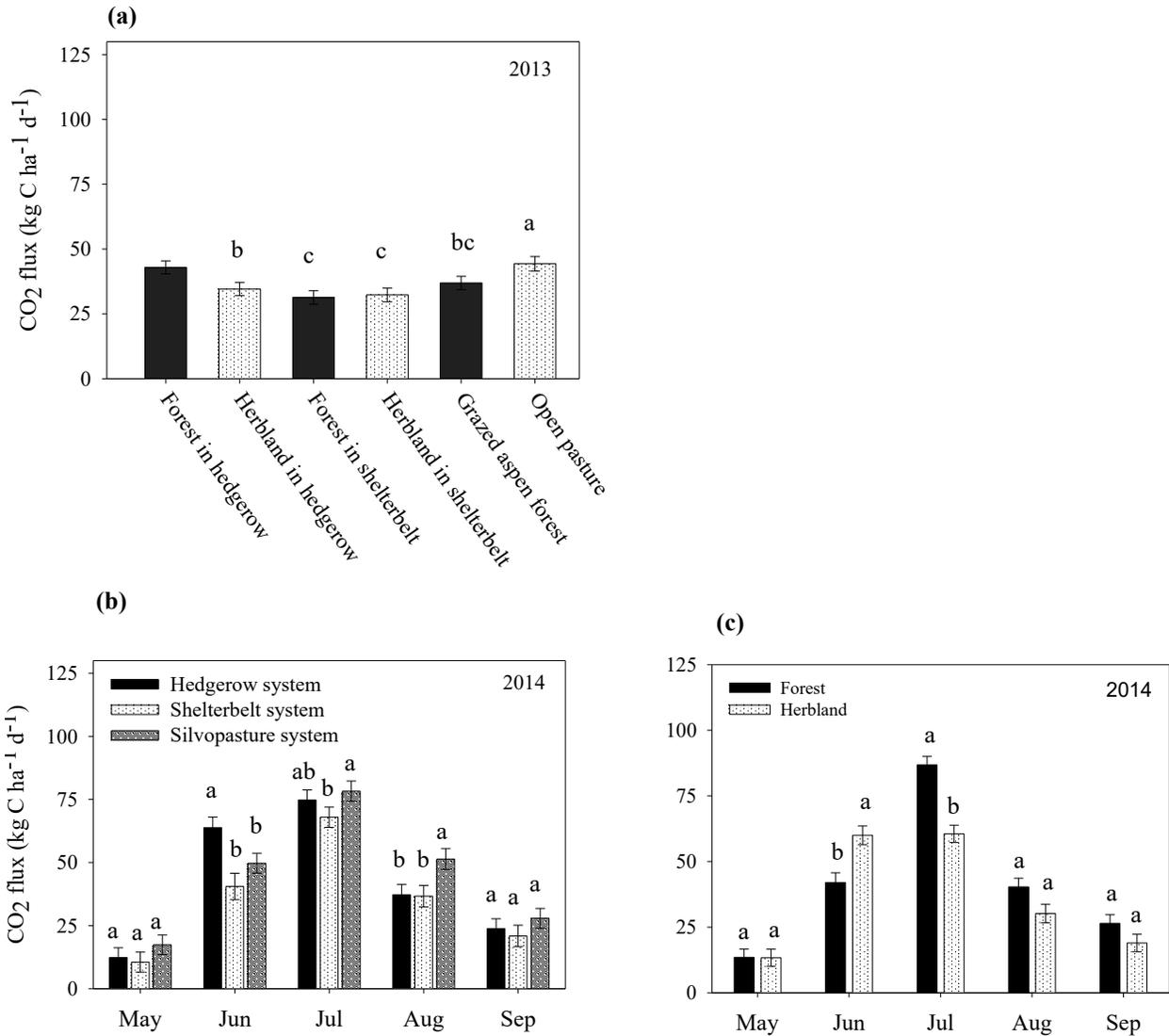
**Table 5-4.** Analysis of variance ( $F$  and  $P$  values) of the dependence of soil CO<sub>2</sub> emissions ( $Flux$ , kg C ha<sup>-1</sup> d<sup>-1</sup>) on soil temperature ( $T$ , °C) using the regression coefficient ( $Flux = ae^{bT}$ ) and temperature sensitivity coefficient ( $Q_{10} = e^{10b}$ ) as affected by agroforestry system and land cover type (forest and herbland) within each agroforest system, and their interactions<sup>a</sup>.

Significance	2013			2014		
	$df$	$F$	$P$	$df$	$F$	$P$
Agroforestry (AF)	2	1.72	0.12	2	0.08	0.92
Land cover type (L)	1	<b>4.42</b>	<b>0.02</b>	1	1.42	0.24
AF x L	2	<b>2.65</b>	<b>0.06</b>	2	0.61	0.55

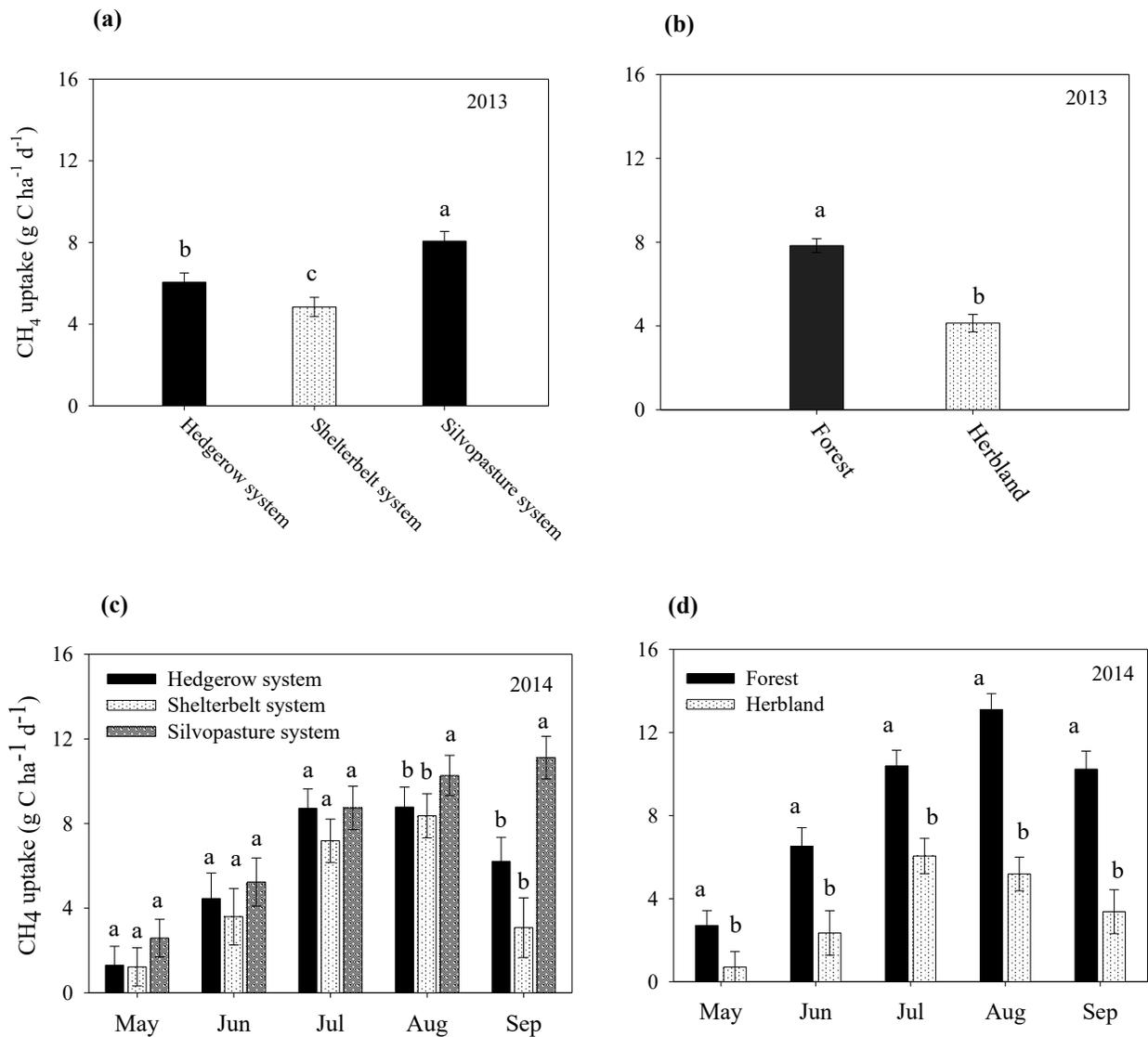
<sup>a</sup> A separate analysis was done for each sampling year (2013 and 2014); F-stats with  $p$  values < 0.10 indicate significance and are shown in bold



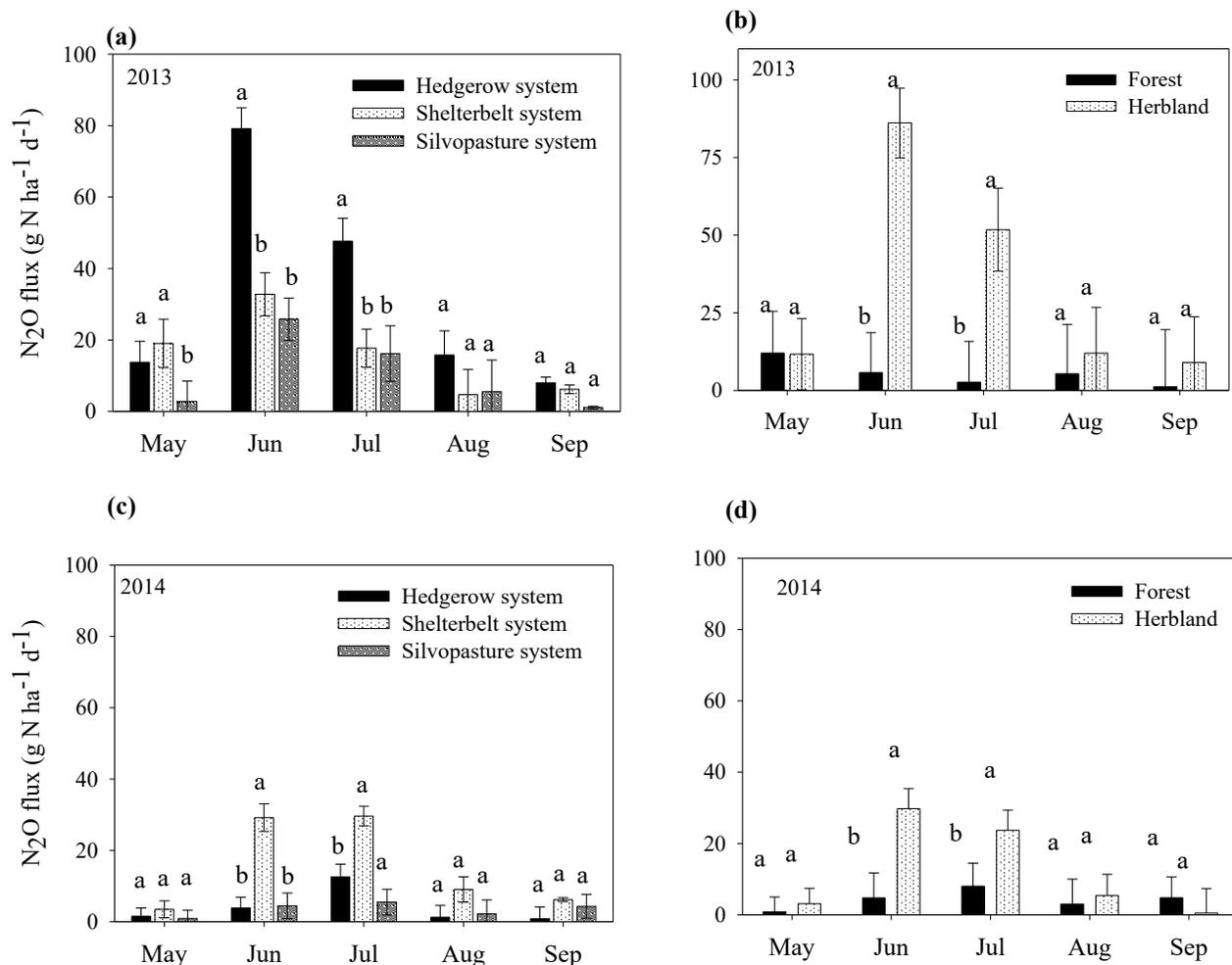
**Fig. 5-1.** Changes in soil temperature and water content within the 0-10 cm soil layer from May to September in 2013 and 2014 as affected by different agroforestry systems and land cover types in central Alberta, Canada: a) and b) interaction effect of agroforestry system and land cover type on soil temperature and water content in 2013, c) and d) effect of agroforestry system on soil temperature and water content in 2014, and e) and f) effect of land cover type on soil temperature and water content in 2014. Vertical bars are SE of the means ( $n=12$ ).



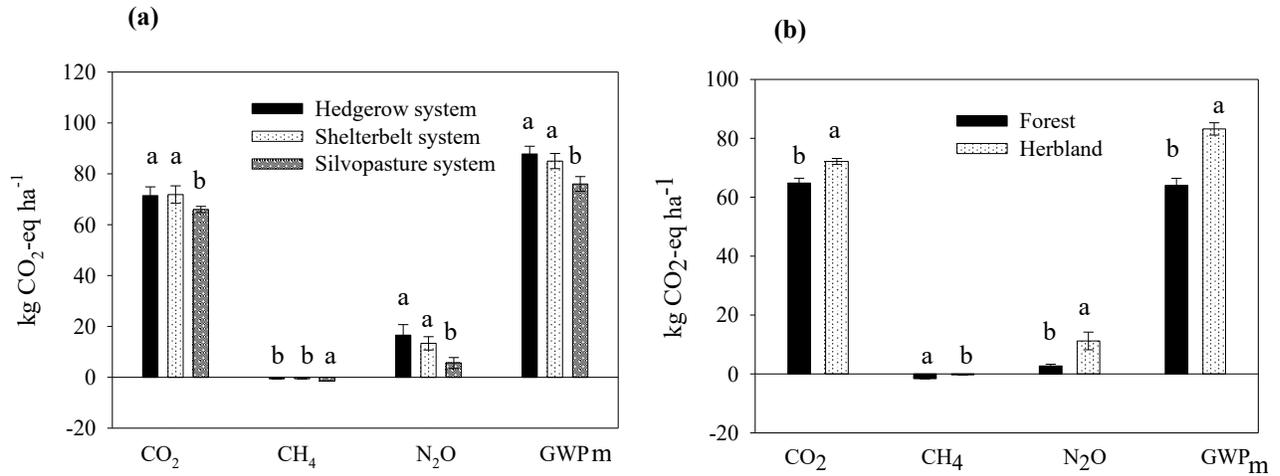
**Fig. 5-2.** Measured soil CO<sub>2</sub> emissions from May to September in 2013 and 2014, as affected by different agroforestry systems and land cover types in central Alberta, Canada: (a) interaction effect of agroforestry system and land cover type on CO<sub>2</sub> emission in 2013, (b) interaction effect of agroforestry system and date of sampling on CO<sub>2</sub> emissions in 2014, and (c) interaction effect of land cover type and date of sampling on CO<sub>2</sub> emissions in 2014. Means (±SE) followed by different letters within a month are significantly different at  $p < 0.10$ .



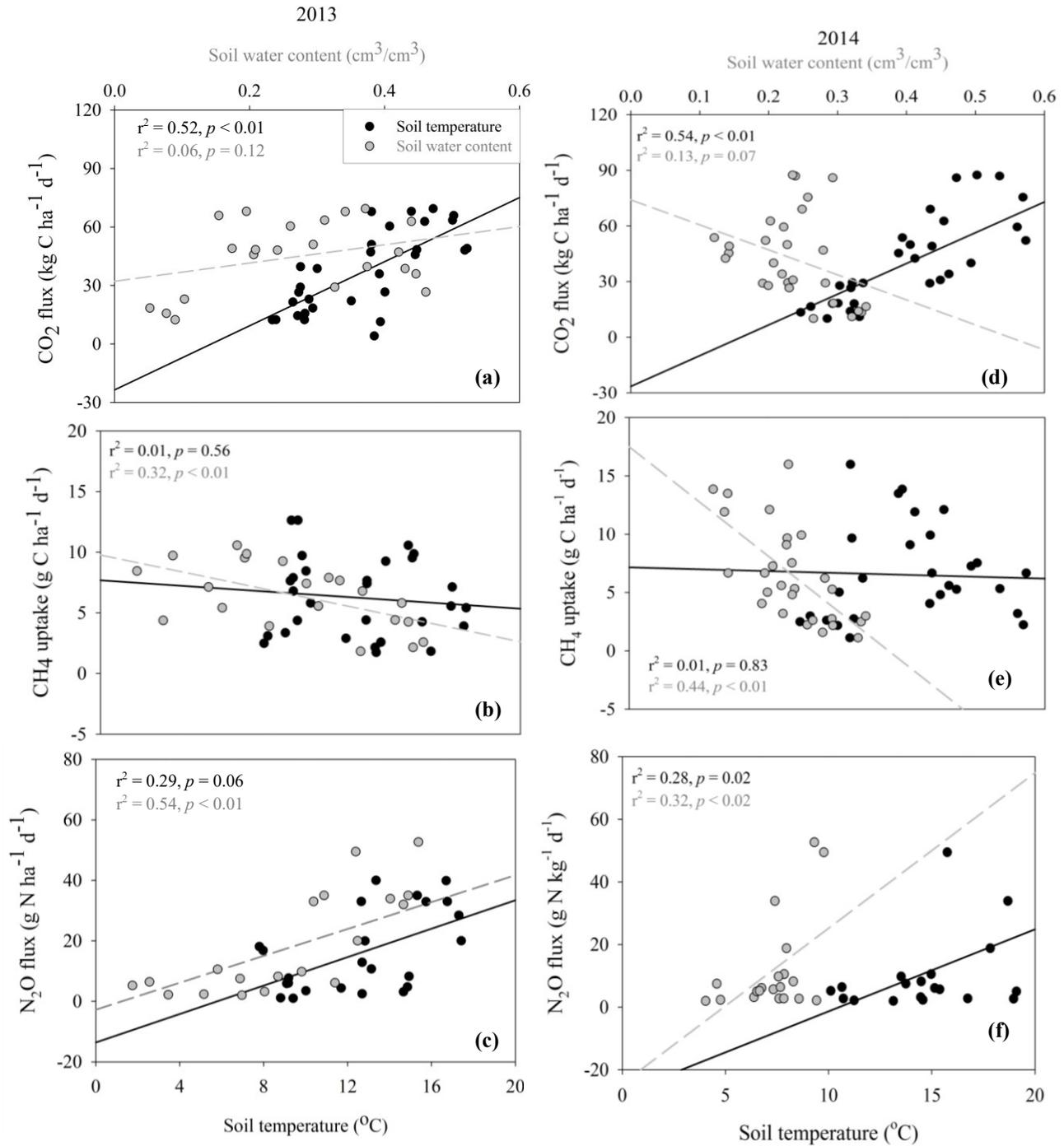
**Fig. 5-3.** Measured soil CH<sub>4</sub> uptakes from May to September in 2013 and 2014 as affected by different land use systems in central Alberta, Canada: (a) effect of agroforestry system on CH<sub>4</sub> uptake in 2013, (b) effect of land cover type on CH<sub>4</sub> uptake in 2013, (c) interaction effect of agroforestry system and date of sampling on CH<sub>4</sub> uptake in 2014, and (d) interaction effect of land cover type and date of sampling on CH<sub>4</sub> uptake in 2014. Means ( $\pm$ SE) followed by different letters within a month are significantly different at  $p < 0.10$ .



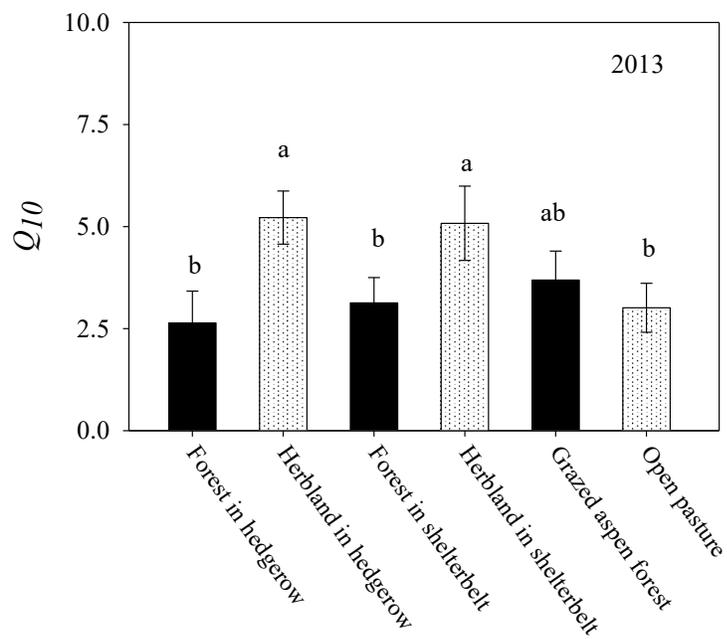
**Fig. 5-4.** Measured soil N<sub>2</sub>O emissions from May to September in 2013 and 2014 as affected by agroforestry system and land cover type in central Alberta, Canada: (a) interaction effect of agroforestry system and date of sampling on N<sub>2</sub>O emission in 2013, (b) interaction effect of land cover type and date of sampling on N<sub>2</sub>O emission in 2013, (c) interaction effect of agroforestry system and date of sampling on N<sub>2</sub>O emission in 2014, and (d) interaction effect of land cover type and date of sampling on N<sub>2</sub>O emission in 2014. Means ( $\pm$ SE) followed by different letters within a month are significantly different at  $p < 0.10$ .



**Fig. 5-5.** Global warming potential of microbe-mediated soil GHG emission (GWP<sub>m</sub>): (a) effect of agroforestry system, (b) effect of land cover type. The calculation of GWP<sub>m</sub> was based on CO<sub>2</sub> emitted via heterotrophic respiration and CH<sub>4</sub>, and N<sub>2</sub>O fluxes by converting each gas concentration to CO<sub>2</sub> equivalent over a 100-yr time scale, using a conversion factor of 1 for CO<sub>2</sub>, 25 for CH<sub>4</sub>, and 298 for N<sub>2</sub>O.



**Fig. 5-6.** The relationship between each of soil  $\text{CO}_2$  emissions,  $\text{CH}_4$  uptake, and  $\text{N}_2\text{O}$  emissions and each of soil temperature ( $^{\circ}\text{C}$ ) and soil water content ( $\text{cm}^3/\text{cm}^3$ ) within the 0-10 soil layer in 2013 ((a), (b), (c)) and 2014 ((d), (e), (f)). Data are based on mean values of each sampling date. Within figures, regressions of soil temperature with each of the greenhouse emissions are represented by black lines, whereas those of soil water content are given in gray lines.



**Fig. 5-7.** Temperature sensitivity ( $Q_{10}$ ) for soil CO<sub>2</sub> emissions as affected by different agroforestry systems and land cover types in central Alberta, Canada. Treatment effects were significant only in 2013. Means ( $\pm$ SE) followed by different letters within a month are significantly different at  $p < 0.10$ .

## References

- AARD (2004). Alberta fertilizer guide [online] Available from [http://www1.agric.gov.ab.ca/\\$department/deptdocs.nsf/all/agdex3894](http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex3894) [accessed 29 December 2015]
- Abdalla M, Jones M, Smith P, Williams M (2009) Nitrous oxide fluxes and denitrification sensitivity to temperature in Irish pasture soils. *Soil Use and Management*, 25, 376-388.
- Ågren GI (2000) Temperature dependence of old soil organic matter. *AMBIO: A Journal of the Human Environment*, 29, 55-55.
- Ågren GI, Bosatta E (2002) Reconciling differences in predictions of temperature response of soil organic matter. *Soil Biology and Biochemistry*, 34, 129-132.
- Albrecht A, Kandji ST (2003) Carbon sequestration in tropical agroforestry systems. *Agriculture, Ecosystems & Environment*, 99, 15-27.
- Amadi CC, Van Rees KC, Farrell RE (2016) Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in shelterbelts compared with adjacent cropped fields. *Agriculture, Ecosystems and Environment*, 223, 123-134.
- Arevalo C, Bhatti JS, Chang SX, Jassal RS, Sidders D (2010) Soil respiration in four different land use systems in north central Alberta, Canada. *Journal of Geophysical Research-Biogeosciences*, 115, G01003.

Baah-Acheamfour M, Carlyle CN, Bork EW, Chang SX (2014) Trees increase soil carbon and its stability in three agroforestry systems in central Alberta, Canada. *Forest Ecology and Management*, 328, 131-139.

Baah-Acheamfour M, Chang SX, Carlyle CN, Bork EW (2015) Carbon pool size and stability are affected by trees and grassland cover types within agroforestry systems of western Canada. *Agriculture, Ecosystems & Environment*, 213, 105-113.

Ball BC, Scott A, Parker JP (1999) Field N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> fluxes in relation to tillage, compaction and soil quality in Scotland. *Soil and Tillage Research*, 53, 29-39.

Banerjee S, Baah-Acheamfour M, Carlyle CN, Bissett A, Richardson AE, Siddique T, Bork EW, Chang SX (2015) Determinants of bacterial communities in Canadian agroforestry systems. *Environmental Microbiology*, 18, 1805–1816.

Bergeron M, Lacombe S, Bradley RL, Whalen J, Cogliastro A, Jutras M, Arp P (2011) Reduced soil nutrient leaching following the establishment of tree-based intercropping systems in eastern Canada. *Agroforestry Systems*, 83, 321-330.

Boone RD, Nadelhoffer KJ, Canary JD, Kaye JP (1998) Roots exert a strong influence on the temperature sensitivity of soil respiration. *Nature*, 396, 570-572.

Brady N, Weil R (2002) Nitrogen and sulfur economy of soils. In: *The Nature and Properties of Soils* (13th Ed.), pp. 542–638. Upper Saddle River, NJ: Prentice-Hall, Inc.

- Breuer L, Huisman J, Keller T, Frede H (2006) Impact of a conversion from cropland to grassland on C and N storage and related soil properties: Analysis of a 60-year chronosequence. *Geoderma*, 133, 6-18.
- Butterbach-Bahl K, Baggs EM, Dannenmann M, Kiese R, Zechmeister-Boltenstern S (2013) Nitrous oxide emissions from soils: how well do we understand the processes and their controls? *Philosophical transactions of the Royal Society of London. Series B, Biological Sciences*, 368, 20130122.
- Cao G, Tang Y, Mo W, Wang Y, Li Y, Zhao X (2004) Grazing intensity alters soil respiration in an alpine meadow on the Tibetan plateau. *Soil Biology and Biochemistry*, 36, 237-243.
- Christiansen JR, Gundersen P (2011). Stand age and tree species affect N<sub>2</sub>O and CH<sub>4</sub> exchange from afforested soils. *Biogeosciences*, 8, 2535-2546.
- Clary J (2012) Determinants of perennial and annual grass distribution in Mediterranean-climate California. *Plant Ecology*, 213, 1203-1208.
- Conant RT, Drijber RA, Haddix ML, Parton WJ, Paul EA, Plante AF, Six J, Steinweg JM (2008) Sensitivity of organic matter decomposition to warming varies with its quality. *Global Change Biology*, 14, 868-877.
- Curiel Yuste J, Baldocchi D, Gershenson A, Goldstein A, Misson L, Wong S (2007) Microbial soil respiration and its dependency on carbon inputs, soil temperature and moisture. *Global Change Biology*, 13, 2018-2035.

- Davidson EA, Belk E, Boone RD (1998) Soil water content and temperature as independent or confounded factors controlling soil respiration in a temperate mixed hardwood forest. *Global Change Biology*, 4, 217–227.
- Davidson EA, Janssens IA (2006) Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, 440, 165-173.
- Desjardins R, Smith W, Grant B, Campbell C, Riznek R (2005) Management strategies to sequester carbon in agricultural soils and to mitigate greenhouse gas emissions. In: *Increasing Climate Variability and Change*. pp. 283-297. Springer.
- Dougherty MC, Thevathasan NV, Gordon AM, Lee H, Kort J (2009) Nitrate and *Escherichia coli* NAR analysis in tile drain effluent from a mixed tree intercrop and monocrop system. *Agriculture, Ecosystems & Environment*, 131, 77-84.
- Environment Canada. 2013. Canada's emission trend. [online] <http://www.ec.gc.ca/indicateurs-indicators/default.asp?lang=en&n=CCED3397-1>, [accessed in May 2015].
- Environment Canada. 2015. Alberta weather condition [online] [http://weather.gc.ca/forecast/canada/index\\_e.html?id=AB](http://weather.gc.ca/forecast/canada/index_e.html?id=AB), [accessed in February 2015]
- Evers AK, Bambrick A, Lacombe S et al. (2010) Potential greenhouse gas mitigation through temperate tree-based intercropping systems. *Open Agriculture Journal*, 4, 49-57.
- Fang C, Smith P, Smith J (2006) Is resistant soil organic matter more sensitive to temperature than the labile organic matter? *Biogeosciences*, 3, 65-68.

- FAO (2014) FAOSTAT Emissions Database [online] <http://www.wri.org/resources/datasets/cait-country-greenhouse-gas-emissions-data>, [accessed in February 2015].
- Fierer N, Craine JM, McLauchlan K, Schimel JP (2005) Litter quality and the temperature sensitivity of decomposition. *Ecology*, 86, 320-326.
- Gacengo CN, Wood CW, Shaw JN, Raper RL, Balkcom KS (2009) Agroecosystem management effects on greenhouse gas emissions across a coastal plain catena. *Soil Science*, 174, 229-237.
- Green S, Vogeler I, Clothier B, Mills T, Van Den Dijssel C (2003) Modelling water uptake by a mature apple tree. *Soil Research*, 41, 365-380.
- Horwath WR, Pregitzer KS, Paul EA (1994) 14C allocation in tree-soil systems. *Tree Physiology*, 14, 1163-1176.
- Janssens I, Carrara A, Ceulemans R (2004) Annual  $Q_{10}$  of soil respiration reflects plant phenological patterns as well as temperature sensitivity. *Global Change Biology*, 10, 161-169.
- Janssens I, Lankreijer H, Matteucci G et al. (2001) Productivity overshadows temperature in determining soil and ecosystem respiration across European forests. *Global Change Biology*, 7, 269-278.
- Jastrow J (1996) Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. *Soil Biology & Biochemistry*, 28, 665-676.

- Jensen LS, Salo T, Palmason F et al. (2005) Influence of biochemical quality on C and N mineralisation from a broad variety of plant materials in soil. *Plant and Soil*, 273, 307-326.
- Kelting DL, Burger JA, Edwards GS (1998). Estimating root respiration, microbial respiration in the rhizosphere, and root-free soil respiration in forest soils. *Soil Biology and Biochemistry*, 30, 961-968.
- Knorr W, Prentice I, House J, Holland E (2005) Long-term sensitivity of soil carbon turnover to warming. *Nature*, 433, 298-301.
- Lal R (2004) Soil carbon sequestration to mitigate climate change. *Geoderma*, 123, 1-22.
- Lambert M, Fréchette J (2005) Analytical techniques for measuring fluxes of CO<sub>2</sub> and CH<sub>4</sub> from hydroelectric reservoirs and natural water bodies. In: *Greenhouse Gas Emissions—Fluxes and Processes*. pp. 37-60. Springer,
- Liebig M, Morgan J, Reeder J, Ellert B, Gollany H, Schuman G (2005) Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada. *Soil and Tillage Research*, 83, 25-52.
- Liu XJ, Mosier AR, Halvorson AD, Zhang FS (2006) The impact of nitrogen placement and tillage on NO, N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> fluxes from a clay loam soil. *Plant and Soil*, 280, 177-188.
- MacDonald J, Skiba U, Sheppard L, Ball B, Roberts J, Smith K, Fowler D (1997) The effect of nitrogen deposition and seasonal variability on methane oxidation and nitrous oxide

- emission rates in an upland spruce plantation and moorland. *Atmospheric Environment*, 31, 3693-3706.
- Martins MR, Jantalia CP, Polidoro JC, Batista JN, Alves BJ, Boddey RM, Urquiaga S (2015) Nitrous oxide and ammonia emissions from N fertilization of maize crop under no-till in a Cerrado soil, *Soil and Tillage Research*, 151, 75-81.
- Meinzer FC, Goldstein G, Andrade JL (2001) Regulation of water flux through tropical forest canopy trees: do universal rules apply? *Tree physiology*, 21, 19-26.
- Mosier A, Schimel D, Valentine D, Bronson K, Parton W (1991) Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. *Nature*, 350, 330-332.
- Mosier AR, Halvorson AD, Reule CA, Liu XJ (2006) Net global warming potential and greenhouse gas intensity in irrigated cropping systems in northeastern Colorado. *Journal of Environmental Quality*, 35, 1584-1598.
- Mu Z, Huang A, Ni J, Li J, Liu Y, Shi S, Xie D, Hatano R (2013) Soil greenhouse gas fluxes and net global warming potential from intensively cultivated vegetable fields in southwestern China. *Journal of Soil Science and Plant Nutrition*, 13, 566-578.
- Müller T, Walter B, Wirtz A, Burkovski A (2006) Ammonium toxicity in bacteria. *Current Microbiology*, 52, 400-406.
- Mutuo PK, Cadisch G, Albrecht A, Palm C, Verchot L (2005) Potential of agroforestry for carbon sequestration and mitigation of greenhouse gas emissions from soils in the tropics, *Nutrient Cycling in Agroecosystems*, 71, 43-54.

- Myhre G, Shindell D, Breon F-M et al. (2013) Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group 1 to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (eds Stocker TF, Qin D, Plattner G-K, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM), pp. 659–740. Cambridge University Press, Cambridge, UK and New York, NY, USA.
- Nelson RG (2002) Resource assessment and removal analysis for corn stover and wheat straw in the Eastern and Midwestern United States—rainfall and wind-induced soil erosion methodology, *Biomass and Bioenergy*, 22, 349-363.
- Nyborg M, Laidlaw JW, Solberg ED, Malhi SS (1997) Denitrification and nitrous oxide emissions from a Black Chernozemic soil during spring thaw in Alberta. *Canadian Journal of Soil Science*, 77, 153-160.
- Ohashi M, Gyokusen K, Saito A (2000) Contribution of root respiration to total soil respiration in a Japanese cedar (*Cryptomeria japonica* D. Don) artificial forest, *Ecological Research*, 15, 323-33.
- Pang X, Bao W, Zhu B, Cheng W (2013) Responses of soil respiration and its temperature sensitivity to thinning in a pine plantation. *Agricultural and Forest Meteorology*, 171, 57-64.
- Paustian K, Six J, Elliott E, Hunt H (2000) Management options for reducing CO<sub>2</sub> emissions from agricultural soils. *Biogeochemistry*, 48, 147-163.

- Peichl M, Thevathasan N, Gordon AM, Huss J, Abohassan RA (2006) Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. *Agroforestry Systems*, 66, 243-257.
- Raghoebarsing AA, Smolders AJ, Schmid MC et al. (2005) Methanotrophic symbionts provide carbon for photosynthesis in peat bogs. *Nature*, 436, 1153-1156.
- Reeder J, Schuman G (2002) Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands. *Environmental Pollution*, 116, 457-463.
- Reeves S, Wang W (2015) Optimum sampling time and frequency for measuring N<sub>2</sub>O emissions from a rain-fed cereal cropping system. *Science of The Total Environment*, 530, 219-226.
- Ruser R, Flessa H, Russow R, Schmidt G, Buegger F, Munch J (2006) Emission of N<sub>2</sub>O, N<sub>2</sub> and CO<sub>2</sub> from soil fertilized with nitrate: effect of compaction, soil moisture and rewetting. *Soil Biology and Biochemistry*, 38, 263-274.
- Sainju UM, Jabro JD, Stevens WB (2008) Soil carbon dioxide emission and carbon content as affected by irrigation, tillage, cropping system, and nitrogen fertilization, *Journal of Environmental Quality*, 37, 98-106.
- Sainju UM, Barsotti JL, Wang J (2014) Net global warming potential and greenhouse gas intensity affected by cropping sequence and nitrogen fertilization. *Soil Science Society of America Journal*, 78, 248-261.

- Sainju UM, Stevens WB, Caesar-TonThat T, Liebig MA (2012) Soil greenhouse gas emissions affected by irrigation, tillage, crop rotation, and nitrogen fertilization, *Journal of Environmental Quality*, 41, 1774-1786.
- SAS Institute Inc (2014) SAS User's Guide: Statistics SAS/C Online Doc, Release 9.30. SAS Inc., Cary, NC.
- Schoeneberger M, Bentrup G, de Gooijer H, Soolanayakanahally R, Sauer T, Brandle J, Zhou X, Current D (2012) Branching out: agroforestry as a climate change mitigation and adaptation tool for agriculture, *Journal of Soil and Water Conservation*, 67,128A-36A.
- Silva JP, Lasso A, Lubberding HJ, Peña MR, Gijzen HJ (2015) Biases in greenhouse gases static chambers measurements in stabilization ponds: Comparison of flux estimation using linear and non-linear models. *Atmospheric Environment*, 109, 130-138.
- Six J, Elliott E, Paustian K (2000) Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biology and Biochemistry*, 32, 2099-2103.
- Six J, Ogle SM, Conant RT, Mosier AR, Paustian K (2004) The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. *Global Change Biology*,10, 155-1560.
- Smemo K, Yavitt J (2011) Anaerobic oxidation of methane: an underappreciated aspect of methane cycling in peatland ecosystems? *Biogeosciences*, 8, 779-793.

- Smith P, Haberl H, Popp A, Erb KH, Lauk C, Harper R, Tubiello FN, Siqueira-Pinto A, Jafari M, Sohi S, Masera O (2013) How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals?. *Global Change Biology*, 19, 2285-2302.
- Smith P, Martino D, Cai Z et al. (2008) Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society of London. Series B, Biological sciences*, 363, 789-813.
- Soil Classification Working Group (1998) *The Canadian System of Soil Classification*. NRC Research Press, Ottawa, Canada.
- Stolk P, Jacobs C, Moors E, Hensen A, Velthof G, Kabat P (2009) Significant non-linearity in nitrous oxide chamber data and its effect on calculated annual emissions. *Biogeosciences Discussions*, 6, 115-141.
- Syakila A, Kroeze C (2011) The global nitrous oxide budget revisited. *Greenhouse Gas Measurement and Management*, 1, 17-26.
- Teepe R, Brumme R, Beese F (2000) Nitrous oxide emissions from frozen soils under agricultural, fallow and forest land. *Soil Biology and Biochemistry*, 32, 1807-1810.
- Van den Pol-van Dasselaar, A, Van Beusichem M, Oenema O (1998) Effects of soil moisture content and temperature on methane uptake by grasslands on sandy soils, *Plant and Soil*, 204, 213-222.

Venterea RT, Halvorson AD, Kitchen N, Liebbig MA, Cavigelli MA, Del Grosso SJ, Motavalli PP, Nelson KA, Spokas KA, Singh BP, Stewart CE (2012) Challenges and opportunities for mitigating nitrous oxide emissions from fertilized cropping systems. *Frontiers in Ecology and the Environment*, 10, 562-570.

Vargas R, Collins SL, Thomey ML et al (2012) Precipitation variability and fire influence the temporal dynamics of soil CO<sub>2</sub> efflux in an arid grassland. *Global Change Biology*, 18, 1401-1411.

Willison T, Webster C, Goulding K, Powlson D (1995) Methane oxidation in temperate soils: effects of land use and the chemical form of nitrogen fertilizer. *Chemosphere*, 30, 539-546.

Wotherspoon A, Thevathasan NV, Gordon AM, Voroney RP (2014) Carbon sequestration potential of five tree species in a 25-year-old temperate tree-based intercropping system in southern Ontario, Canada. *Agroforestry Systems*, 88, 631-643.

Zhang A, Cui L, Pan G, Li L, Hussain Q, Zhang X, Zheng J, Crowley D (2010) Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from Tai Lake plain, China. *Agriculture, Ecosystems & Environment*, 139, 469-475.

## **Chapter 6. Forest and grassland cover types increase CO<sub>2</sub> emissions from agricultural soils by enhancing autotrophic but not heterotrophic respiration**

### **1. Introduction**

The integration of trees and shrubs into crop and livestock production such as in the form of agroforestry has been suggested as a practice capable of offsetting carbon dioxide (CO<sub>2</sub>) emissions due to its ability to act as a large sink for atmospheric carbon (C) (Schoeneberger, 2009). Uncertainties exist, however, regarding the impact of trees on net emissions of C from soils. Soil respiration is one of the largest sources of terrestrial C released to the atmosphere; about 80 Pg of sequestered C per year is lost through this pathway (Raich and Schlesinger, 1992; Boone et al., 1998). Soil C losses represent the sum of autotrophic respiration from live roots and their microbial symbionts (especially mycorrhizal fungi) and heterotrophic respiration associated with the oxidation of plant detritus, root exudates and humified organic matter by soil heterotrophs decomposing organic matter (Boone et al., 1998). The C lost through autotrophic respiration is tied to the consumption of organic compounds synthesized by primary producers themselves, and is independent of total C levels in the soil (Horwath et al., 1994; Ohashi et al., 2000), and instead reflects live belowground phytomass. Heterotrophic respiration primarily leads to a decline in C accumulation in the soil, which in turn, reduces net ecosystem productivity (Hanson et al., 2000). Unfortunately, most soil respiration measurements in agroforestry systems are interpreted without simultaneous information on autotrophic and heterotrophic components.

Autotrophic and heterotrophic respiration can each account for about 50% of total soil respiration (Hanson et al. 2000, Bond-Lamberty et al. 2004, and Subke et al. 2006) , but this value varies widely among ecosystems, and can range from 10 to 98% in forest and grassland

ecosystems (Hanson et al., 2000; Högberg et al., 2001; Epron, 2009, Gomez-Casanovas et al., 2012). Such large variation is related to differences in biotic and abiotic conditions within the ecosystem, including heterogeneity of vegetation coverage, root abundance and distribution, microclimate conditions, soil organism population size, substrate availability, and the time scales used (Mo et al., 2008; Van Der Heijden et al., 2008), or may simply be an artifact of sampling methodology. For example, Son and Kim (1996) used soil trenching to isolate and partition autotrophic and heterotrophic respiration in pitch pine (*Pinus rigida*) and Japanese larch (*Larix leptolepis*) forests, and estimated autotrophic contribution comprised only 3% of total soil respiration. Using the same method, Lee et al. (2010) found autotrophic respiration was 34% of total respiration in a similar ecosystem and climatic regime. One potential problem with the study by Son and Kim (1996) is that most respiration measurements were taken immediately after trenching, which can result in a large amount of dead root mass and lead to an overestimation of heterotrophic respiration (Hanson et al., 2000).

It is difficult to predict the response of soil respiration or its components to variation in ecosystem conditions, although an understanding of these responses is critical for evaluating how changes in these factors affect ecosystem-atmosphere C exchange. The sensitivity of soil respiration to temperature changes will largely determine the effect of autotrophic and heterotrophic activity on net C flux from soils to the atmosphere when climate warms in the future (Boone et al., 1998). While it has been suggested that soil organic matter decomposition by heterotrophs is more sensitive to temperature changes than autotrophic respiration (Kirschbaum, 2000), the opposite has also been reported (Högberg et al., 2001). Carbon lost through autotrophic respiration does not originate from soil C (Horwath et al., 1994; Ohashi et al., 2000); however, a high temperature sensitivity of autotrophic respiration among plants that

allocate abundant photosynthate to roots can limit C sequestration by soils like its heterotrophic counterpart (Gomez-Casanovas et al., 2012). Consequently, it is necessary to simultaneously measure changes in soil C due to heterotrophic and autotrophic respiration to determine whether an ecosystem is gaining or losing C.

The main objective of this study was to use a large-scale field experiment to test the relative contribution of autotrophic and heterotrophic components to total soil respiration in hedgerow, shelterbelt, and silvopasture agroforestry systems, as well as their component land cover types (forest vs. herbland). In addition, a sensitivity analysis was performed to understand how the various systems will impact fluxes of C from soils to the atmosphere in response to future temperature changes. Within each agroforestry system, areas under forest were hypothesized to have greater autotrophic respiration but lower heterotrophic respiration compared to herbland. Moreover, this response is projected to occur because of a greater supply of vegetative inputs (aboveground and roots) during the growing season as well as lower soil temperatures under perennial vegetation, which in turn slow down the microbial processes responsible for soil organic matter mineralization. Finally, I predicted that the silvopasture would have greater autotrophic respiration than both hedgerow and shelterbelt systems because of the continuous presence of live vegetation from the combination of perennial forest and grassland cover types in the former.

## **2. Materials and Methods**

### **2.1. Site description**

Research sites were located across a 270 km long north-south soil/climate gradient spanning the prairie and parkland natural regions of central Alberta, Canada. Mean (30-yr) annual precipitation across the study area was 480 mm, and mean annual air temperature 2.6 °C (Environment Canada, 2015). Precipitation during the 2013 and 2014 sampling season (April 1 to September 30) was 306 and 407 mm, respectively. Mean air temperature in 2013 was 13.8 °C, and 13.2°C in 2014. Soils were classified as Luvisols (Soil Classification Working Group, 1998), Dark Gray Chernozems, and Black Chernozems in the northern, central, and southern portions of the study area, respectively.

The field study used a split-plot design, which included agroforestry system (hedgerow, shelterbelt and silvopasture) as the whole plot, and land cover types (forest vs. herbland) as the split-plot (i.e. subplot). Herblands within the hedgerow and shelterbelt systems were comprised of annual cropland, while the silvopasture system included perennial grassland (as the herbland component) growing in conjunction with patches of aspen forest. A total of 36 agroforestry plots (12 hedgerow sites, 12 shelterbelt sites, and 12 silvopasture sites) that included 72 subplots (36 each for forest and herbland) were sampled. Four permanent sampling points for autotrophic and heterotrophic measurements were established within each agroforestry system (or whole plot), with two sampling points randomly set up within each of the forest and herbland land use types. Sampling points in forested subplots were established in the center of the treed area to minimize the edge effects, whereas those in the herbland were located one tree height (~ 30 m) from the edge of the treed zone to avoid the influence of trees. Wire cages, secured to the ground with metal stakes, were used to protect equipment used for soil respiration measurement in the silvopastoral system from damage by cattle (*Bos taurus*).

## 2.2. Partitioning of autotrophic and heterotrophic components

Data on total soil respiration in this study were collected from static chamber measurements in Chapter 5. To account for the contribution of autotrophic and heterotrophic components to total respiration, I used a modified root exclusion (trenching) method (Kelting et al. 1998). This method is based on the assumption that root severance around the circumference of plastic soil collars should reduce autotrophic respiration to negligible levels within three months after trenching. To achieve this, soil collars made of PVC (Schedule SDR35), 21.2 cm in diameter and 35 cm long, were inserted into the soil at each of two permanent sampling points set up within the forest and herbland subplots of each agroforestry system in September, 2012. Collars were installed to leave 2 or 3 cm above the soil surface, and were beveled on the bottom to facilitate insertion with minimal soil compaction. Soil collars were maintained free from vegetation by spraying with Roundup<sup>®</sup> (Glyphosate, 540 g L<sup>-1</sup>, diluted 10:1 with water) at a monthly application rate of 5 L ha<sup>-1</sup> (Saurette et al., 2008). Glyphosate can be directly and rapidly degraded by microbes (Haney et al., 2000) without adversely affecting microbial activity in soils when used under recommended conditions (Stratton and Stewart, 1992). In addition, aboveground live vegetation was gently removed by hand from inside the collars at regular intervals but mulch or LFH material was retained. Collars in the annual cropland component of hedgerow and shelterbelt systems were removed during spring tillage and promptly reinstalled as close as possible to the original location.

Soil heterotrophic respiration was measured monthly in each sampling point from mid-May through September, during 2013 and 2014 with a Li-Cor 8100A soil respiration system (LI-8100A, LI-COR Biosciences, Lincoln, Nebraska, USA), equipped with an infrared gas analyzer

and a 20 cm inside diameter flow-through chamber. Measurements were made on the same day, and in most cases, within 30 min of static chamber measurement. Gas flux was calculated as changes in CO<sub>2</sub> concentration over time, recorded by a data logger at 5s intervals between 1 and 3 min after placing the chamber over the collar. Triplicate measurements were averaged to obtain a mean heterotrophic respiration measurement for each collar. Shorter collars (11.4 cm long) were also inserted at 12 agroforestry sites (4 each in the hedgerow, shelterbelt and silvopasture systems) to measure total soil respiration with the Li-Cor 8100A system. The purpose of these collars was to verify the similarity between collar and static chamber measurements of total soil respiration. About 8-9 cm of the short soil collars was inserted into the soil to provide a solid foundation and reduce lateral diffusion of CO<sub>2</sub> in the soil below the chamber.

Root severance can increase soil water content, which can affect soil temperature and heterotrophic respiration rates (Baggs et al., 2006). To test if this occurred in this study, I measured soil temperature and volumetric water content at the 0-10 cm depth inside and outside of the soil collars with Omega T-handled Type E thermocouples (Model 8100-201, Omega Engineering, Montreal, QC) and time domain reflectometry (TDR) probes (Model Theta Probe ML2X, Delta-T Devices, Cambridge, England), respectively, at the time of each gas sampling. Soil temperature and water content did not differ inside and outside the collars (data not shown), indicating temperature and moisture were not affected by root severance.

### **2.3. Calculations**

Soil autotrophic respiration was calculated as the difference between total soil respiration from the static chamber system and heterotrophic respiration from the Li-Cor 8100A system, and

assumed autotrophic respiration was reduced to negligible levels eight months after collar installation (Saurette et al., 2008). The sensitivity of both autotrophic and heterotrophic respiration to temperature changes was determined by fitting a first-order exponential model (Eqn. 1) to mean soil CO<sub>2</sub> efflux for each sampling date to soil temperatures measured within the 0-10 cm soil layer.

$$Flux = \beta_o \exp^{\beta_1 T} \quad [1]$$

where  $\beta_o$  and  $\beta_1$  are fitted constants and  $T$  is temperature. The temperature sensitivity coefficient  $Q_{10}$  (which denotes the rate of change in CO<sub>2</sub> emissions for each 10 °C of change in soil temperature) was calculated using Eqn. [2]:

$$Q_{10} = \exp^{10\beta_1} \quad [2]$$

#### 2.4. Statistical analysis

The relationship between measurements of total soil respiration using the Li-Cor 8100A soil respiration system and fluxes computed with the static chamber system was assessed, with a regression based on paired measurements over the two years. Values of autotrophic and heterotrophic respiration, and the ratio of heterotrophic to total respiration were compared among land uses with ANOVA using Proc MIXED in SAS (SAS Institute Inc., 2014), and included repeated measures since observations were made over time on the same experimental unit. Residual plots and influential plots were evaluated to ensure model assumptions of normality and

equal variance were satisfied. No parameters required transformation. In running the repeated measures ANOVA, agroforestry system and land cover type were considered fixed effects, whereas sampling date was treated as the repeated measure fixed variable. Replications within each agroforestry system were treated as random. Least significant difference (LSD) tests were used to separate treatment means and treatment-by-sampling date effects when ANOVA indicated significant effects at  $\alpha < 0.10$ , unless stated otherwise. A liberal alpha value was used to reduce the risk of a Type II error given that gas samples were collected from treatments across a large geographic area that had substantial variation in soil properties and vegetation composition.

### **3. Results**

#### **3.1. Total CO<sub>2</sub> effluxes measured with a Li-Cor 8100A and the static chamber method**

Correspondence between the two methods of assessing soil CO<sub>2</sub> exchange was high ( $r^2 = 0.84$ ; Fig. 6-1); however, the slope of the line was less than one, indicating that the Li-Cor CO<sub>2</sub> readings were slightly lower than the static chamber values for low gas flux, but higher than the static chamber when gas flux was high. Overall, deviations between the static chamber and automated system data remained non-significant, with both the y-intercept (12.6) and slope (0.89) of the regression line remaining statistically similar to zero and one, respectively, at the 5% significance level.

#### **3.2. Land cover effects on autotrophic and heterotrophic respiration**

In both 2013 and 2014, soil autotrophic respiration was affected by agroforestry system, land cover type, sampling date, and the interaction between agroforestry system and sampling date (Table 6-1). Autotrophic respiration differed among agroforestry systems from June through August in 2013; autotrophic was lower in the hedgerow during June than the other systems, but greater in silvopasture than hedgerow and shelterbelts during July and August (Fig. 6-2a). In the same period, soil autotrophic respiration was 33% lower under herbland relative to adjacent forest land cover (Fig. 6-2b). In 2014, the silvopasture again showed greater autotrophic respiration than the hedgerow and shelterbelt systems, particularly in July and August (Fig. 6-2c), while the hedgerow was greater than the others in June, but lower in September (Fig. 6-2c). Autotrophic respiration was again 31% lower in herbland compared to forest in 2014 (Fig. 6-2d).

Heterotrophic respiration was affected by most treatments and their interactions in 2013 and 2014 (Table 6-1). Heterotrophic respiration in 2013 differed among agroforestry systems only in June, July, and August (Fig. 6-3a), a pattern that varied markedly over time. While heterotrophic respiration was greatest in the hedgerow and lowest in the shelterbelt in June, by July the silvopasture had increased above the other systems. One month later in August, the shelterbelt exceeded both of the other systems in heterotrophic respiration. The mean heterotrophic respiration from annual croplands (averaged across all sites and sampling dates) was significantly greater than both the forest and open pasture in 2013 (Fig. 6-3b). Heterotrophic respiration in 2014 peaked in June through July then declined thereafter (Fig. 6-3c). Across most sampling dates (all but September), heterotrophic respiration within the shelterbelt was generally lower than in the hedgerow and silvopasture systems, with the latter systems peaking in June and

July, respectively (Fig. 6-3c). Additionally, heterotrophic respiration in forests was lower than in herbland during June and August, while the opposite pattern occurred during July (Fig. 6-3d).

The mean growing season contribution of heterotrophic to total respiration (expressed as  $R_H/R_T$ ) was affected by agroforestry system in 2013 but not 2014 (Table 6-2); ratios were significantly greater in the hedgerow and shelterbelt than the silvopasture system in 2013. A similar effect of land cover type was observed in both years, with a lower  $R_H/R_T$  in forest compared to adjacent herbland (Table 6-2).

### **3.3. Response of soil respiration components to temperature changes**

A first-order exponential relationship between soil respiration and soil temperature showed that in 2013 and 2014, the temperature sensitivity ( $Q_{10}$ ) of both autotrophic and heterotrophic respiration was affected by agroforestry systems as well as land cover type, but not their interaction (Table 6-3). On average, autotrophic respiration was more sensitive to temperature change ( $Q_{10} = 4.52$ ) than heterotrophic respiration ( $Q_{10} = 3.83$ ) across all sites and sampling dates. The  $Q_{10}$  value calculated for autotrophic respiration in 2013 and 2014 were consistently greater in the silvopasture than in the hedgerow and shelterbelt systems (Table 6-4). Forests also had greater  $Q_{10}$  values than herbland soils, but only in 2013 (Table 6-4). In contrast, temperature sensitivity of heterotrophic respiration in the silvopasture remained below that of the hedgerow and shelterbelt systems during both years. Temperature sensitivity of autotrophic respiration was greater in forest than herbland soils in 2013, but the reverse occurred in 2014 (Table 6-4).

## **4. Discussion**

Heterotrophic respiration was 54% of total respiration across all sites and sampling dates in the current study, indicating heterotrophic respiration dominated total respiration. Further, hedgerow and shelterbelt establishment was estimated to result in greater heterotrophic respiration than silvopasture system, but this response was recorded only in 2013. Temperature and precipitation effects on the rate of heterotrophic respiration vary seasonably (Suseela et al., 2012), and this may have caused the differences between years. Air temperature during the 2013 sampling season was generally higher than that during 2014, but the opposite was found for precipitation (Environment Canada, 2015). Though precipitation was low in 2013, rain events might have adequately wetted the uppermost soil layers, where most of the heterotrophic activity occurred (Yuste et al., 2003). This phenomenon is expected to be more predominant in annual cropland within the hedgerow and shelterbelt systems and appeared to increase heterotrophic respiration, potentially due to high microbial activity.

Heterotrophic respiration primarily causes C accumulated in the soil to be lost, which in turn reduces net ecosystem productivity (Boone et al., 1998). Soil  $R_H/R_T$  has been shown to vary between 63 to 83% in the region encompassed by this study (Saurette et al., 2008; Arevalo et al., 2010). According to literature reviews (Hanson et al., 2000; Subke et al., 2006), heterotrophic and autotrophic respiration generally each account for about half of total soil respiration. Relative contributions of autotrophic and heterotrophic respiration to soil respiration may converge at approximately 50% when soil organic C is in dynamic equilibrium (Nakane et al., 1983). Over the two growing seasons examined, the fraction of total soil respiration derived from heterotrophic respiration in the silvopasture was as high as 51%. Hence, the silvopasture system may be near such equilibrium in soil organic C compared to the hedgerow and shelterbelt

systems, possibly because of the permanence of vegetation in both embedded land cover types (i.e., forest and perennial pasture) and lack of ongoing tillage. The greater heterotrophic respiration observed in the hedgerow and shelterbelt systems is attributed to the annually cropped areas, where decomposition of organic matter may be greater (Curiel et al., 2003). Compared to forested areas and perennial pasture, soil management practices in annual croplands such as tillage, fertilizer inputs, and harvesting are known to increase heterotrophic respiration more than autotrophic respiration via increases in activities of soil microorganisms (Bond-Lamberty and Thomson, 2010). Herblands also have been found to contain a large pool of readily decomposable C within the light fraction of soil organic matter (Baah-Acheamfour et al., 2015). This result, together with the high levels of soil disturbance in the topsoil of the annually cropped herblands, may explain its greater heterotrophic respiration relative to autotrophic respiration.

Activities of soil microorganisms are mainly controlled by temperature, soil water content, organic matter availability, soil texture and aggregation (Curiel et al., 2003; Banerjee et al., 2015), although these ecosystem factors could further be modified by the type and abundance of vegetation. Results suggest the establishment and retention of woody vegetation in agriculture can lead to a larger reduction in heterotrophic respiration, despite enhancing autotrophic respiration presumably due to greater total plant biomass in this vegetation type. The lower  $R_H/R_T$  ratios evident in forest compared to herbland were similar to those reported by Arevalo et al. (2010), where the  $R_H/R_T$  over two growing seasons was 25% in native aspen forest, and 35% within an agricultural land use system lacking trees. In a study of soil respiration in shelterbelt and adjacent cropped fields, Amadi et al. (2016) reported high soil  $CO_2$  efflux in areas with trees relative to adjacent cropped fields without trees. The authors attributed the result to the enhanced

root activity due to shelterbelt establishment and increased microbial activity arising from continuous litter input by woody vegetation. Lee and Jose (2003) indicated microbial biomass, soil organic matter, and soil pH were the major factors affecting soil respiration in a 7-year-old cottonwood (*Populus deltoids*), while fine root production and soil organic matter were the major factors affecting soil respiration in loblolly pine (*Pinus taeda*) plantation of similar age. High root activity in forest is responsible for the observed increases in CO<sub>2</sub> emission but not microbial respiration. Moderation in soil micro-environment (e.g. high moisture, humidity, and low temperature) via tree and shrub establishment could be responsible for the low R<sub>H</sub>/R<sub>T</sub> ratios in forest. Although encouraging the planting or maintenance of trees within the agricultural landscape can beneficially increase ecosystem C sequestration (Garrity, 2004, Schoeneberger, 2009; Swallow et al., 2009), there is also potential to limit the C sequestration benefit of this management practice if C lost through autotrophic respiration is not separated from the net C flux of the agro-ecosystem. Logically, one would expect litter quality to be lower and decomposition rates slower in forest than herbland areas (Cambardella and Elliott, 1994).

Heterotrophic respiration in herblands also proved more sensitive to changes in soil temperature than in forest only in 2014, indicating the effect of land cover types on the sensitivity of microbial respiration to projected temperature changes can vary widely between growing seasons (Suseela et al., 2012). According to Fierer et al. (2005), high Q<sub>10</sub> values of soil heterotrophic respiration suggest the presence of more labile C forms, which are less stable and sensitive to ongoing temperature changes. This suggestion was not supported by our 2014 results, or by a similar laboratory mesocosm study in which the recalcitrant forms of organic C were more sensitive to changes in temperature than labile forms (Högberg et al., 2010). The high values of Q<sub>10</sub> for autotrophic respiration in the silvopasture system suggest net C flux from soils

may depend more on the relative abundance of roots and associated fungi as well as their contribution to total soil respiration, while the opposite may be true in the hedgerow and shelterbelt systems (Yuste et al., 2003; Tang et al., 2005). In contrast, the silvopasture emitted greater amounts of soil CO<sub>2</sub> through autotrophic respiration from live roots and their microbial symbionts in July and August. This observation is not necessarily indicative of soil C losses, but rather shows greater consumption of organic compounds synthesized by primary producers themselves, and may be independent of C in the soil (Horwath et al., 1994; Ohashi et al., 2000). Moreover, high Q<sub>10</sub> values of autotrophic respiration could also limit soil C sequestration particularly in perennial vegetation that allocates more C to roots (i.e. forest and grassland). The re-allocation of photosynthates from above-ground to roots within vegetation enhances root respiration (Gomez-Casanovas et al., 2012), which in turn is regulated by temperature (Hawkes et al., 2008).

Results also indicate a strong time dependency in autotrophic and heterotrophic respiration, demonstrating complex interactions between temperature and a range of other factors, such as soil water content, substrate availability and microbial population size (Yuste et al., 2003). For example, autotrophic respiration was high from June to August, coincident with a period of high photosynthetic rates in response to increased summer temperature (Sampson et al., 2007; Högberg et al., 2010; Kuzyakov et al., 2010). The hedgerow and shelterbelt system had annual cropland in which live root production is highly seasonal, occurring mostly from June through August, but remaining relatively low the rest of the year. Notably, peak cropland canopy closure typically occurs in July and appeared to reduce heterotrophic respiration, potentially due to greater insulation of soil, which would cool soil temperature and reduce microbial activity. These results are similar to Casanovas et al. (2012) who showed the contribution of autotrophic

to total soil respiration was only 20% late in the growing season, a value down from 60% early in the growing season. Limited live vegetation from May through mid-June, and again in late summer (late August through September), would result in lower autotrophic respiration across the annual cropland land uses.

## **5. Conclusions**

The presence of perennial vegetation in an annually cropped agricultural landscape altered the autotrophic and heterotrophic components of soil respiration, which is known to affect net C flux from soils. Compared to herbland, most of the CO<sub>2</sub> released from soils under forest was contributed by autotrophic rather than heterotrophic activity. Planting or retaining trees in an otherwise cropped landscape may lead to a larger reduction in heterotrophic respiration, though the practice could increase autotrophic sensitivity to temperature when the climate warms in the future. Conversely, herblands had autotrophic respiration less dependent on temperature, but had stronger heterotrophic responses, though the latter were not consistent across years. Overall, the establishment of silvopastures has greater potential to induce soil C sequestration than the hedgerow and shelterbelt systems, because it leads to a larger reduction in heterotrophic respiration, and lower response of heterotrophic activity to temperature.

**Table 6-1.** Analysis of variance (*F* and *P* values) for the effect of agroforestry system, land cover type (forest and herbland) within each system, and their interaction on soil autotrophic ( $R_A$ ) and heterotrophic ( $R_H$ ) respiration from June to September in 2013 and 2014.<sup>a</sup>

Significance	2013			2014		
	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>
<b><math>R_A</math> (kg C ha<sup>-1</sup> d<sup>-1</sup>)</b>						
Agroforestry (AF)	2	<b>3.87</b>	<b>0.05</b>	2	<b>12.51</b>	<b>&lt;0.01</b>
Land cover type (L)	1	<b>7.64</b>	<b>&lt;0.01</b>	1	<b>15.94</b>	<b>&lt;0.01</b>
AF x L	2	0.74	0.48	2	0.48	0.63
Date of sampling (D)	4	<b>9.95</b>	<b>&lt;0.01</b>	4	<b>8.54</b>	<b>&lt;0.01</b>
AF x D	8	<b>2.21</b>	<b>0.07</b>	8	<b>2.15</b>	<b>0.07</b>
L x D	4	2.29	0.10	4	1.66	0.17
AF x L x D	8	0.69	0.51	8	1.55	0.21
<b><math>R_H</math> (kg C ha<sup>-1</sup> d<sup>-1</sup>)</b>						
Agroforestry (AF)	2	<b>3.15</b>	<b>0.06</b>	2	<b>14.39</b>	<b>&lt;0.01</b>
Land cover type (L)	1	<b>30.80</b>	<b>&lt;0.01</b>	1	<b>9.64</b>	<b>&lt;0.01</b>
AF x L	2	<b>9.95</b>	<b>&lt;0.01</b>	2	1.25	0.30
Date of sampling (D)	4	<b>13.88</b>	<b>&lt;0.01</b>	4	<b>16.32</b>	<b>&lt;0.01</b>
AF x D	8	<b>2.99</b>	<b>0.03</b>	8	<b>2.73</b>	<b>0.03</b>
L x D	4	0.94	0.43	4	<b>2.65</b>	<b>0.04</b>
AF x L x D	8	0.47	0.62	8	0.73	0.66

<sup>a</sup> A separate analysis was done for each sampling year (2013 and 2014); F-stats with *p* values < 0.10

indicate significance and are shown in bold.

**Table 6-2.** Ratio of heterotrophic to total respiration ( $R_H/R_T$ ; means with standard errors in parentheses) within soils under different agroforestry systems and land cover types (forest and herbland). The interaction effects of agroforestry system by land cover type were not significant ( $p > 0.10$ ); therefore, only main treatment effects are listed ( $n = 12$ ).

Treatment	$R_H/R_T$ (%)	
	2013	2014
<b>Agroforestry</b>		
Hedgerow	68 (2.4) a	49 (2.1) a
Shelterbelt	64 (3.4) a	45 (4.1) a
Silvopasture	54 (2.9) b	48 (3.1) a
LSD <sub>0.10</sub>	3.56	0.20
Prob > F	0.03	0.81
<b>Land cover type</b>		
Forest	52 (2.8) b	46 (3.2) b
Herbland	60 (3.0) a	57 (2.6) a
LSD <sub>0.10</sub>	23.62	21.21
Prob > F	<0.01	<0.01

Within a column, means with different letters differ among the agroforestry systems or between land cover types significantly at  $p < 0.10$  according to the Fisher's protected multiple comparison test.

**Table 6-3.** Analysis of variance (*F* and *P* values) for the temperature sensitivity ( $Q_{10}$ ) of soil autotrophic ( $R_A$ ) and heterotrophic respirations ( $R_H$ ) as affected by agroforestry system, land cover type (forest vs. herbland) and their interactions from June to September in 2013 and 2014.<sup>a</sup>

Factor	2013			2014		
	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>
$R_A$ (Kg C ha <sup>-1</sup> d <sup>-1</sup> )						
Agroforestry (AF)	2	<b>4.87</b>	<b>&lt;0.01</b>	2	<b>5.28</b>	<b>&lt;0.01</b>
Land cover type (L)	1	<b>5.64</b>	<b>&lt;0.01</b>	1	<b>9.16</b>	<b>&lt;0.01</b>
AF x L	2	0.68	0.71	2	0.74	0.55
$R_H$ (Kg C ha <sup>-1</sup> d <sup>-1</sup> )						
Agroforestry (AF)	2	<b>6.42</b>	<b>&lt;0.01</b>	2	<b>5.23</b>	<b>&lt;0.01</b>
Land cover type (L)	1	<b>5.67</b>	<b>&lt;0.01</b>	1	<b>8.69</b>	<b>&lt;0.01</b>
AF x L	2	0.92	0.52	2	1.02	0.23

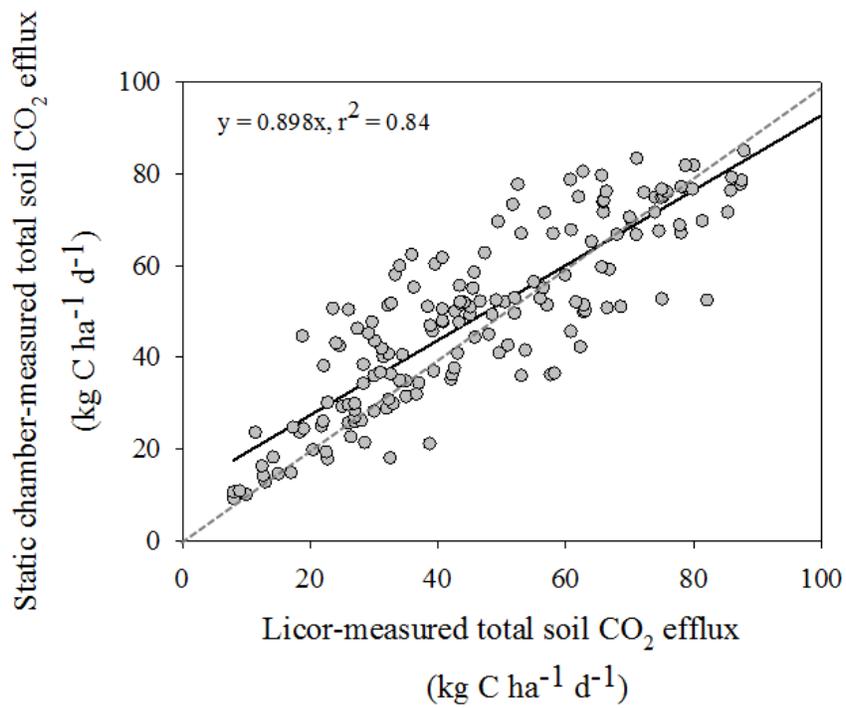
<sup>a</sup> A separate analysis was done for each sampling year (2013 and 2014); F-stats with *p* values < 0.10

indicate significance and are shown in bold.

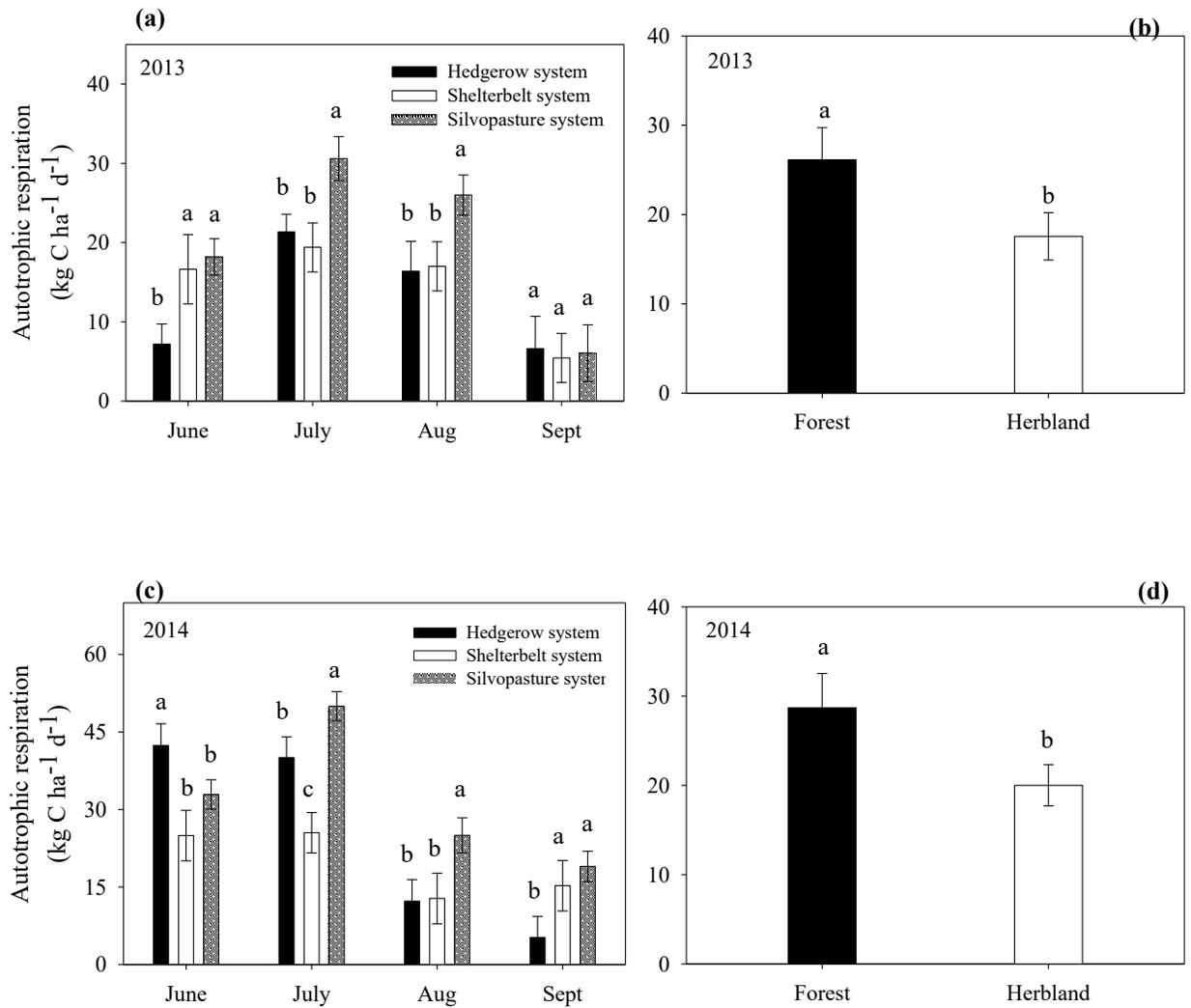
**Table 6-4.** Temperature sensitivity ( $Q_{10}$ ; means with standard errors in parentheses) of autotrophic and heterotrophic soil respirations in soils under different agroforestry systems and land cover types (forest and herbland). Interactions between agroforestry system and land cover type were not significant for any parameters listed ( $p > 0.10$ ), therefore, only the main treatment effects are shown ( $n = 12$ ).

Treatment	$Q_{10}$ - Autotrophic respiration (kg C ha <sup>-1</sup> d <sup>-1</sup> ) <sup>a</sup>				$Q_{10}$ - Heterotrophic respiration (kg C ha <sup>-1</sup> d <sup>-1</sup> )			
	A	B	r <sup>2</sup>	Q <sub>10</sub>	a	b	r <sup>2</sup>	Q <sub>10</sub>
<b>2013</b>								
<b>Agroforestry</b>								
Hedgerow	12.88	0.142	0.53	4.14 (0.3) b	26.89	0.133	0.66	3.78 (0.6) a
Shelterbelt	20.22	0.157	0.62	4.80 (0.5) b	24.61	0.122	0.48	3.68 (0.4) a
Silvopasture	14.61	0.176	0.54	5.81 (0.1) a	27.67	0.119	0.53	3.28 (0.6) b
<b>Land cover type</b>								
Forest	23.26	0.158	0.58	4.85 (0.4) a	26.91	0.136	0.58	3.89 (0.9) a
Herbland	12.15	0.135	0.56	3.86 (0.5) b	28.47	0.124	0.65	3.46 (0.7) b
<b>2014</b>								
<b>Agroforestry</b>								
Hedgerow	24.98	0.148	0.44	4.40 (0.9) b	24.47	0.148	0.56	4.39 (0.5) a
Shelterbelt	19.62	0.161	0.55	3.86 (1.1) b	26.82	0.156	0.53	4.75 (0.6) a
Silvopasture	31.72	1.352	0.47	5.01 (0.2) a	18.75	0.132	0.49	3.74 (0.4) b
<b>Land cover type</b>								
Forest	17.56	1.516	0.52	4.55 (1.0) a	25.22	0.129	0.58	3.02 (0.4) b
Herbland	28.69	0.138	0.49	3.97 (0.8) a	27.33	0.131	0.51	3.85 (0.6) a

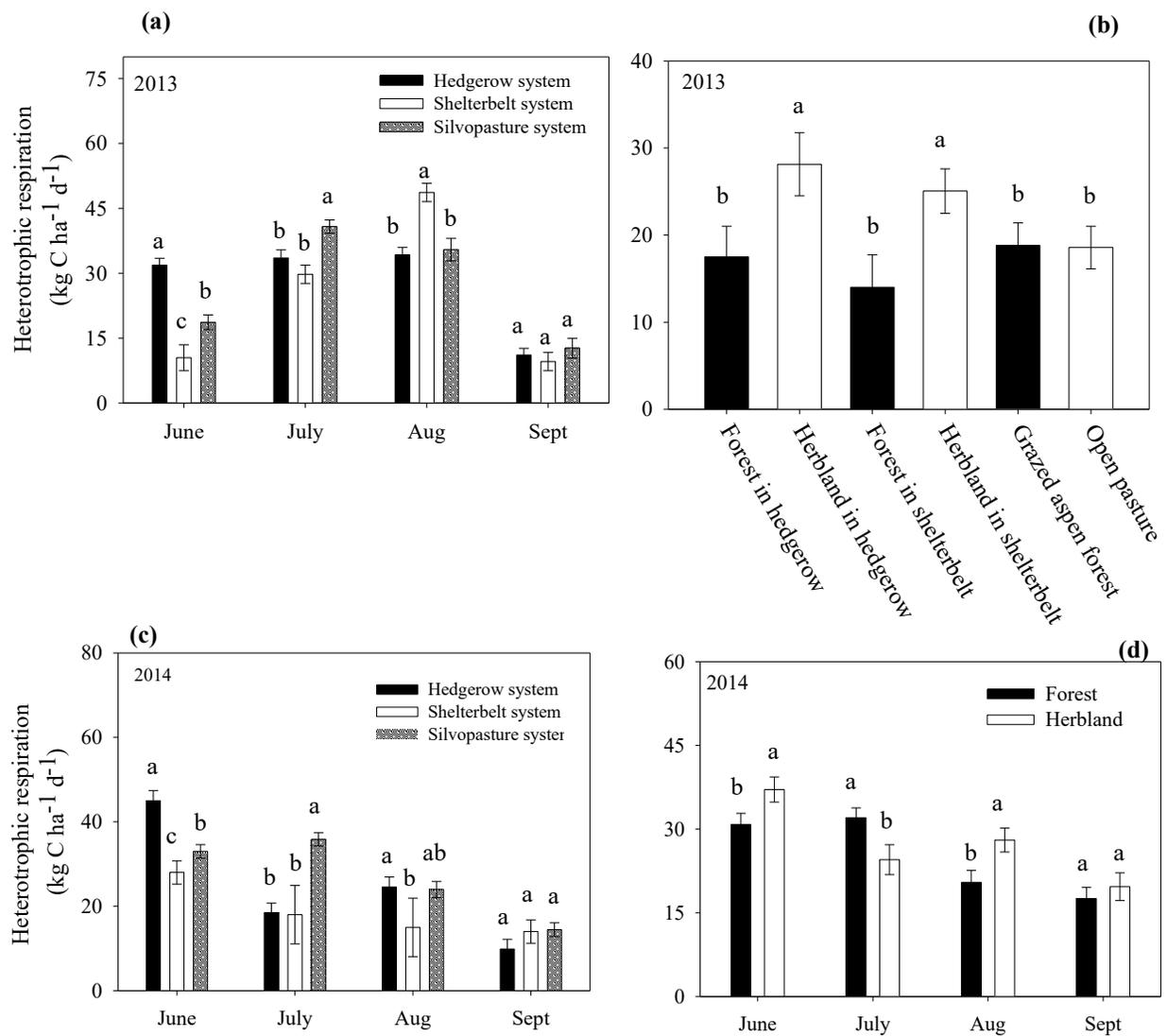
<sup>a</sup> Within a column, means with different letters differ among the agroforestry systems or between land cover types significantly at  $p < 0.10$  according to the Fisher's protected multiple comparison test.



**Fig. 6-1.** Comparison of total soil CO<sub>2</sub> effluxes measured using the static chamber and the Li-Cor 8100A automated chamber techniques. The dashed line is the 1:1 line and the solid line is the regression between the static and automated chamber data pairs.



**Fig. 6-2.** Soil autotrophic respiration ( $R_A$ ) from June to September in 2013 and 2014 as affected by different land uses in central Alberta, Canada: a) interaction effects of agroforestry system and date of sampling on  $R_A$  in 2013, b) effect of land cover type on  $R_A$  in 2013, c) interaction effects of agroforestry system and date of sampling on  $R_A$  in 2014, and d) effect of land cover type on  $R_A$  in 2014. Means ( $\pm$  SE) followed by different letters within a month are significantly different at  $p < 0.10$ .



**Fig. 6-3.** Measured soil heterotrophic respiration ( $R_H$ ) from June to September in 2013 and 2014, as affected by different land uses across central Alberta, Canada: a) interaction effects of agroforestry system and date of sampling on  $R_H$  in 2013, b) interaction effects of agroforestry system and land cover type on  $R_H$  in 2013, c) interaction effects of agroforestry system and date of sampling on  $R_H$  in 2014, and d) interaction effects of land cover type and date of sampling on  $R_H$  in 2014. Means ( $\pm$ SE) followed by different letters within a month are significantly different at  $p < 0.10$ .

## References

- Arevalo C, Bhatti JS, Chang SX, Jassal RS, Sidders D. 2010. Soil respiration in four different land use systems in north central Alberta, Canada. *J Geophys Res: Biogeosciences*, 115.
- Amadi CC, Van Rees KC, Farrell RE. 2016. Soil–atmosphere exchange of carbon dioxide, methane and nitrous oxide in shelterbelts compared with adjacent cropped fields. *Agric Ecosyst Environ* 223:123-34.
- Baah-Acheamfour M, Carlyle CN, Bork EW, Chang SX. 2014. Trees increase soil carbon and its stability in three agroforestry systems in central Alberta, Canada. *For Ecol Manage* 328:131-9.
- Baah-Acheamfour M, Chang SX, Carlyle CN, Bork EW. 2015. Carbon pool size and stability are affected by trees and grassland cover types within agroforestry systems of western Canada. *Agric Ecosyst Environ* 213:105-13.
- Baggs E. 2006. Partitioning the components of soil respiration: a research challenge. *Plant Soil* 284:1-5.
- Banerjee S, Baah-Acheamfour M, Carlyle CN, Bissett A, Richardson AE, Siddique T, Bork EW, Chang SX. 2015. Determinants of bacterial communities in Canadian agroforestry systems. *Environ Microbiol*, <http://dx.doi.org/10.1111/1462-2920.12986>.
- Bond-Lamberty B, Thomson A. 2010. Temperature-associated increases in the global soil respiration record. *Nature* 464:579-82.

- Bond-Lamberty B, Wang C, Gower ST. 2004. A global relationship between the heterotrophic and autotrophic components of soil respiration? *Glob Change Biol* 10:1756-66.
- Boone RD, Nadelhoffer KJ, Canary JD, Kaye JP. 1998. Roots exert a strong influence on the temperature sensitivity of soil respiration. *Nature* 396:570-2.
- Cambardella C, Elliott E. 1994. Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. *Soil Sci Soc Am J* 58:123-30.
- Curiel Yuste J, Janssens IA, Carrara A, Meiresonne L, Ceulemans R. 2003. Interactive effects of temperature and precipitation on soil respiration in a temperate maritime pine forest. *Tree Physiol* 23:1263-70.
- Environment Canada. 2015. Alberta weather condition [online]  
[http://weather.gc.ca/forecast/canada/index\\_e.html?id=AB](http://weather.gc.ca/forecast/canada/index_e.html?id=AB), [accessed in May 2015]
- Epron D, Marsden C, M'Bou AT, Saint-André L, d'Annunzio R, Nouvellon Y. 2009. Soil carbon dynamics following afforestation of a tropical savannah with Eucalyptus in Congo. *Plant Soil* 323:309-22.
- Fierer N, Craine JM, McLauchlan K, Schimel JP. 2005. Litter quality and the temperature sensitivity of decomposition. *Ecology* 86:320-6.
- Garrity DP. 2004. Agroforestry and the achievement of the Millennium Development Goals. *Agrofor Syst* 61:5-17.
- Gomez-Casanovas N, Matamala R, Cook DR, Gonzalez-Meler MA. 2012. Net ecosystem exchange modifies the relationship between the autotrophic and heterotrophic

- components of soil respiration with abiotic factors in prairie grasslands. *Glob Change Biol* 18:2532-45.
- Goulden ML, Munger JW, FAN S, Daube BC, Wofsy SC. 1996. Measurements of carbon sequestration by long-term eddy covariance: Methods and a critical evaluation of accuracy. *Glob Change Biol* 2:169-82.
- Haney RL, Senseman SA, Hons FM, Zuberer DA. 2000. Effect of glyphosate on soil microbial activity and biomass. *Weed Science* 48:89-93.
- Hanson P, Edwards N, Garten C, Andrews J. 2000. Separating root and soil microbial contributions to soil respiration: a review of methods and observations. *Biogeochemistry* 48:115-46.
- Högberg P, Nordgren A, Buchmann N, Taylor AF, Ekblad A, Högberg MN et al. 2001. Large-scale forest girdling shows that current photosynthesis drives soil respiration. *Nature* 411:789-92.
- Horwath WR, Pregitzer KS, Paul EA. 1994. 14C Allocation in tree-soil systems. *Tree Physiol* 14:1163-76.
- Hawkes CV, Hartley IP, Ineson P, Fitter AH. 2008. Soil temperature affects carbon allocation within arbuscular mycorrhizal networks and carbon transport from plant to fungus. *Glob Change Biol* 14:1181-90.
- Kelting DL, Burger JA, Edwards GS. 1998. Estimating root respiration, microbial respiration in the rhizosphere, and root-free soil respiration in forest soils. *Soil Biol Biochem* 30:961-8.

- Kirschbaum MU. 2000. Will changes in soil organic carbon act as a positive or negative feedback on global warming?. *Biogeochemistry* 48:21-51.
- Kuzyakov Y, Gavrichkova O. 2010. REVIEW: Time lag between photosynthesis and carbon dioxide efflux from soil: a review of mechanisms and controls. *Glob Change Biol* 16:3386-406.
- Lee KH, Jose S. 2013. Soil respiration, fine root production, and microbial biomass in cottonwood and loblolly pine plantations along a nitrogen fertilization gradient. *For Ecol Manage* 185:263-73.
- Lee N, Koo J, Noh NJ, Kim J, Son Y. 2010. Autotrophic and heterotrophic respiration in needle fir and *Quercus*-dominated stands in a cool-temperate forest, central Korea. *J Plant Res* 123:485-95.
- Mo J, Zhang W, Zhu W, Gundersen P, Fang Y, Li D, Wang H. 2008. Nitrogen addition reduces soil respiration in a mature tropical forest in southern China. *Glob Change Biol* 14:403-412.
- Nakane K, Yamamoto M, Tsubota H. 1983. Estimation of root respiration rate in a mature forest ecosystem. *Nihon Seitai Gakkaishi* 33:397-408.
- Ohashi M, Gyokusen K, Saito A. 2000. Contribution of root respiration to total soil respiration in a Japanese cedar (*Cryptomeria japonica* D. Don) artificial forest. *Ecol Res* 15:323-33.
- Raich JW, Potter CS. 1995. Global patterns of carbon dioxide emissions from soils. *Glob Biogeochem Cycles* 9:23-36.

- Raich JW, Tufekciogul A. 2000. Vegetation and soil respiration: correlations and controls. *Biogeochemistry* 48:71-90.
- Raich J, Schlesinger WH. 1992. The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. *Tellus B* 44:81-99.
- SAS Institute Inc. 2014. SAS User's Guide: Statistics SAS/C Online Doc, Release 9.30. SAS Inc., Cary, NC.
- Sampson D, Janssens I, Curiel Yuste J, Ceulemans R. 2007. Basal rates of soil respiration are correlated with photosynthesis in a mixed temperate forest. *Glob Change Biol* 13:2008-17.
- Saurette DD, Chang SX, Thomas BR. 2008. Autotrophic and heterotrophic respiration rates across a chronosequence of hybrid poplar plantations in northern Alberta. *Can J Soil Sci* 88:261-72.
- Schoeneberger MM. 2009. Agroforestry: working trees for sequestering carbon on agricultural lands. *Agrofor Syst* 75:27-37.
- Soil Classification Working Group. 1998. The Canadian System of Soil Classification. NRC Research Press, Ottawa, Canada, p. 187.
- Son Y, Kim H. 1996. Soil respiration in *Pinus rigida* and *Larix leptolepis* plantations. *J Korean For Soc* (Korea Republic).
- Stratton GW, Stewart KE. 1992. Glyphosate effects on microbial biomass in a coniferous forest soil. *Environ Toxicol Water Qual* 7:223-36.

- Subke J, Inghima I, Francesca Cotrufo M. 2006. Trends and methodological impacts in soil CO<sub>2</sub> efflux partitioning: a metaanalytical review. *Glob Change Biol* 12:921-43.
- Suseela V, Conant RT, Wallenstein MD, Dukes JS. 2012. Effects of soil moisture on the temperature sensitivity of heterotrophic respiration vary seasonally in an old-field climate change experiment. *Glob Change Biol* 18:336-48.
- Swallow BM, Kallesoe MF, Iftikhar UA, van Noordwijk M, Bracer C, Scherr SJ et al. 2009. Compensation and rewards for environmental services in the developing world: framing pan-tropical analysis and comparison. *Ecol Soc* 14:26.
- Tang J, Baldocchi DD. 2005. Spatial-temporal variation in soil respiration in an oak-grass savanna ecosystem in California and its partitioning into autotrophic and heterotrophic components. *Biogeochemistry* 73:183-207.
- Van Der Heijden, Marcel GA, Bardgett RD, Van Straalen NM. 2008. The unseen majority: soil microbes as drivers of plant diversity and productivity in terrestrial ecosystems. *Ecol Lett* 11:296-310.
- Yuste, JC, Janssens IA, Carrara A, Meiresonne L, Ceulemans R. 2003. Interactive effects of temperature and precipitation on soil respiration in a temperate maritime pine forest. *Tree Physiol* 23:1263-1270.

## **Chapter 7. Conclusions and future research**

### **1. Overview of the study objectives**

Western Canada's prairie region is extensively cultivated for agricultural production, which is a large source of greenhouse gas (GHG) emissions (Liebig et al., 2005). Agroforestry systems are common land uses across Canada, which integrate trees into the agricultural landscape and could play a substantial role in sequestering carbon (C) and mitigating increases in atmospheric GHG concentrations (Kort and Turnock, 1998; Schoeneberger, 2009). Although there are a number of studies on C sequestration and GHGs emissions comparing among forest types and different land management practices in Canada (e.g., Nyborg et al., 1997; Malhi et al., 2011; Arevalo et al., 2012), few studied the contribution of planted shelterbelt, natural hedgerow and silvopasture systems to C sequestration and GHG emissions reduction (e.g., Amichev et al., 2015; Amadi et al., 2016), although such agroforestry systems are widely practiced across Canada.

Unfortunately, most of these studies have been conducted over small individual land holdings that make extrapolation of research results to calculate the C sink that can be achieved with agroforestry practices at a provincial or regional level difficult. In general, the potential of agroforestry systems to become a 'managed' sink for atmospheric GHGs in the region has received much less attention, resulting in different data gaps that need to be filled in order to promote the wider adoption of this unique land use system by landowners.

This thesis research tested in a large-scale field study (between 54° 43' and 52° 28' N latitude, and 113° 44' and 113° 17' W longitude) the impacts of forest and herbland (areas without trees) components of three agroforestry systems (hedgerow, shelterbelt, and silvopasture) of western Canada, with the aims of assessing the benefit of planting or retaining woody species in

agriculture and drawing lessons that can better direct future research in this area. The research revolved around four main questions: 1) how much C is stored in bulk soils under the forest and herbland components of the different agroforestry systems? 2) does the integration of perennial vegetation into the agricultural landscape affect the distribution of C among soil fractions, and how stable is the C? 3) can the integration of perennial vegetation into the agricultural landscape provide benefits to minimize soil CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions? 4) which of the studied land use systems would cause more C accumulation in the soil to be lost?.

## **2. Summary of the research results and management implications**

During the study, which took place in central Alberta, Canada from 2012 to 2015, all the studied agroforestry systems proved to have potential to accumulate and store large amounts of soil C and mitigate GHG emissions compared to agricultural land use alone without perennial vegetation. However, the amount of C stored in soils of these systems and quantity of GHGs emitted to the atmosphere varies depending on the land cover type and management practices. In Chapters 3 and 4, the silvopasture remained the most effective agroforestry system for storing C in the bulk soil because of the continuous presence of live vegetation from the combination of perennial forest and grassland cover types. This result complements existing knowledge about the long-term effects of growing trees on ecosystem and soil C storage in a pastureland (Maia et al., 2007; Haile et al., 2008; Pinno and Belanger, 2008). Compared to the hedgerow and shelterbelt, most of the C stored in soils under the silvopasture system should be within the coarse and light fractions, which are considered less stable. The stability of the C will largely determine the effects of a warmer world on net C flux from soils to the atmosphere, and the

hedgerow and shelterbelt systems should be expected to provide more stable offset potential of atmospheric C when the climate warms in the future. The large soil C accumulation in silvopasture system indicates this system directly reduces CO<sub>2</sub> levels in the atmosphere more than the other two systems despite being often established on low quality sites (i.e., coarse-textured soils). Currently, there is 20 million ha of rangeland and pastureland grazed by livestock in the Canadian prairie, much of which already has trees and shrubs (Statistics Canada, 2011). The extent to which silvopasture occurs in the region confirms a large and ongoing ecosystem C accumulation (Bhatti et al., 2002). Because of this, the conservation of existing silvopasture systems and avoiding losses associated with cultivation are key to maximizing C stores. Within each agroforestry system, areas under forest had the most soil C sequestration benefits over annual croplands, while grassland covers within silvopasture maintained C levels similar to that of forests in all agroforestry systems. The implication of these results for C mitigation in agroecosystems is quite straightforward; if trees are beneficial in storing C, then we should be promoting tree planting within cultivated agricultural landscapes. .

Overall, opportunities appear to exist for reducing GHG emissions and mitigate climate change by promoting the establishment of perennial vegetation in the agricultural landscape. Over two seasons (May to September) in 2013 and 2014, forest soils had greater CO<sub>2</sub> emission, higher CH<sub>4</sub> uptake, and lower N<sub>2</sub>O emission than adjacent herbland soils. As a result, forested areas had a smaller global warming potential than their herbland counterpart based on all three GHGs. Although CO<sub>2</sub> emission was greater in the silvopasture than the other two agroforestry systems, soils in the silvopasture also had greater CH<sub>4</sub> uptake and lower N<sub>2</sub>O emission rates as compared with the other two agroforestry systems. The result that CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions differ among the studied land uses could inspire management decisions to reduce GHGs

emissions from agricultural lands by tailoring reduction to specific land use system. Greater fluxes of CO<sub>2</sub> and lower emissions of non-CO<sub>2</sub> GHGs (i.e., CH<sub>4</sub> and N<sub>2</sub>O) in shelterbelts than in adjacent cropped fields in the study region have also been reported in previous studies (Amadi et al., 2016). Reduced CH<sub>4</sub> and N<sub>2</sub>O emission in areas under trees in agroforestry is important given the high global warming potentials of CH<sub>4</sub> (25 times over a 100-year time horizon) and N<sub>2</sub>O (300 times) (Forster et al., 2007). Silvopasture system can become a ‘hot spot’ for CH<sub>4</sub> emissions through indirect emissions from livestock (Beauchemin and McGinn 2006; Beauchemin *et al.* 2010). High CH<sub>4</sub> uptake in the forest cover type could help offset some of the emissions from livestock in silvopasture. Current promotion of agroforestry in Canada to reduce GHG emissions and, therefore, mitigate climate change needs additional consideration beyond just the benefit of C sequestration. Also, it is important to educate farmers on the benefits of forest and grassland retention as well as tree planting, which extend beyond increased C storage.

Soil C losses represented the sum of autotrophic respiration from live roots and their microbial symbionts (especially mycorrhizal fungi) and heterotrophic respiration associated with the oxidation of plant detritus, root exudates and humified organic matter by soil heterotrophs decomposing organic matter (Boone et al., 1998). However, C mitigation will be enhanced if the overall impact leads to a larger reduction in heterotrophic respiration rather than by autotrophic respiration. In Chapter 6, I expand on the investigation in Chapter 5 by quantifying the relative contribution of autotrophic and heterotrophic components to total soil respiration in the studied land use systems. Results showed that most of the soil CO<sub>2</sub> released from forest and herbland cover types in each agroforestry system was contributed by live roots and soil heterotrophs, respectively. Thus, areas under herbland in all agroforestry systems are expected to reduce net ecosystem productivity more than forest. Although the data in Chapter 5 revealed CO<sub>2</sub> emission

in the shelterbelt was low, the average contribution of heterotrophic respiration to total soil respiration was high, which reflects the estimates from the annual cropland area of this system. Another important finding in this investigation is establishing silvopasture system can provide greater potential to induce soil C sequestration than the hedgerow and shelterbelt systems, because it leads to a larger reduction in heterotrophic respiration. However, management decisions may need to avoid labile C losses associated with intensive grazing and other management practices in this system.

There are a number of overarching questions and debates in the scientific community in relation to the sensitivity of soil respiration to temperature changes. While it has been suggested that soil organic matter decomposition by heterotrophs is more sensitive to temperature changes than autotrophic respiration (Kirschbaum, 2000), the opposite has also been reported (Högberg et al., 2001). The sensitivity of autotrophic and heterotrophic respiration to temperature changes will depend on the type of land cover; the temperature sensitivity of autotrophic and heterotrophic respiration was high in forest and herbland, respectively. High temperature sensitivity of heterotrophic respiration in annual cropland could cause more soil C to be lost when the climate warms in the future, highlighting the importance of hedgerow and shelterbelt land cover types to protect this C.

### **3. Recommendations and future research needs**

Further investigations are needed on the nature of the contribution from the deep rooting systems. Even though this study focused only on the upper 0-30 cm of mineral soil, our results clearly demonstrate that integrating trees into agricultural landscapes is important in the context

of increasing soil C accumulation. Inclusion of trees within the agricultural landscape could also affect properties of deeper soil (Haile et al., 2008; Howlett et al., 2011), not only because their roots extend to deeper depth but also as a result of the role of deeper soils in regulating long-term nutrient dynamics and C stabilization. Key questions include to what degree individual plant species impact C storage in the soil profile, and whether planting of fast-growing species such as hybrid poplars has advantages over slow-growing species, such as Norway spruce? Research on this subject could be useful in the context of selecting woody species of practical significance and developing agroforestry systems aimed at increasing long-term soil C storage. A root system that can extend deep into the soil profile would in theory be advantageous in storing C and capturing nutrients that might otherwise be lost beyond the crop rooting zone.

Despite mechanisms in some Canadian jurisdictions to reward landowners for practices that reduce GHG (Alberta Agriculture and Rural Development (AARD) 2015), agroforestry does not qualify due in part to a lack of data that would support C offset policies. Results from this thesis research and other studies in Canada (e.g., Amadi et al., 2016; Amichev et al., 2015; Winans et al., 2016) suggest agroforestry could be for payment under the various GHG emissions trading programs, much like those associated with reduced and zero tillage in annual cropland. It will be worthwhile to scale results from this and other related studies up to the provincial level, to see how much C is being sequestered in agroforestry systems across the province. Once quantified, agroforestry researchers can then work with economists to develop more effective market mechanisms that tie the proper management of these systems, either through their creation (e.g., shelterbelts) or maintenance (e.g., hedgerows or silvopastures), to future C offset and GHG mitigation policies. The development of formal programs to compensate producers for the environmental goods and services that agroforestry systems

provide could have a large impact on the adoption of agroforestry systems in Canada.

Landowners can diversify their income, by harvesting and replanting the trees for pulp wood production or saw lumber. While practitioners wait for agroforestry systems to be recognized as a mitigation projection in the studied region, other defunct support programs such as the Prairie Shelterbelt Program need to be reinstated, so in the future the adoption of agroforestry may not be affected.

#### **4. References**

AARD. 2015. Agricultural carbon offsets: Information for Alberta's offset market [Online]

Available from <http://worldwidewords.agriculture.alberta.ca/agcarbonoffsets> [accessed 10 November 2015].

Amadi, C.C., Van Rees, K.C., Farrell, R.E., 2016. Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in shelterbelts compared with adjacent cropped fields. *Agric. Ecosyst. Environ.* 223, 123-134.

Amichev, B.Y., Bentham, M.J., Cerkowniak, D., Kort, J., Kulshreshtha, S., Laroque, C.P., Piwowar, J.M., Van Rees, K.C., 2015. Mapping and quantification of planted tree and shrub shelterbelts in Saskatchewan, Canada. *Agrofor. Syst.* 89, 49-65.

Arevalo, C., Chang, S.X., Bhatti, J.S., Sidders, D., 2012. Mineralization potential and temperature sensitivity of soil organic carbon under different land uses in the parkland region of Alberta, Canada. *Soil Sci. Soc. Am. J.* 76, 241-251.

- Beauchemin, K., McGinn, S., 2006. Methane emissions from beef cattle: Effects of fumaric acid, essential oil, and canola oil. *J. Anim. Sci.* 84, 1489-1496.
- Beauchemin, K.A., Janzen, H.H., Little, S.M., McAllister, T.A., McGinn, S.M., 2010. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. *Agr. Syst.* 103, 371-379.
- Bhatti, J., Apps, M., Tarnocai, C., 2002. Estimates of soil organic carbon stocks in central Canada using three different approaches. *Can. J. Forest Res.* 32, 805-812.
- Boone, R.D., Nadelhoffer, K.J., Canary, J.D., Kaye, J.P., 1998. Roots exert a strong influence on the temperature sensitivity of soil respiration. *Nature* 396, 570-572.
- Forster, P.M.D., Shine, K.P., Stuber, N., 2007. It is premature to include non-CO<sub>2</sub> effects of aviation in emission trading schemes. *Atmos. Environ.* 41, 3941-3941.
- Haile, S.G., Nair, P., Nair, V.D., 2008. Carbon storage of different soil-size fractions in Florida silvopastoral systems. *J. Environ. Qual.* 37, 1789-1797.
- Högberg, P., Nordgren, A., Buchmann, N., Taylor, A.F., Ekblad, A., Högberg, M.N., Nyberg, G., Ottosson-Löfvenius, M., Read, D.J., 2001. Large-scale forest girdling shows that current photosynthesis drives soil respiration. *Nature* 411, 789-792.
- Howlett, D.S., Mosquera-Losada, M.R., Nair, P., Nair, V.D., Rigueiro-Rodríguez, A., 2011. Soil carbon storage in silvopastoral systems and a treeless pasture in northwestern Spain. *J. Environ. Qual.* 40, 825-832.

- Kirschbaum, M.U., 2000. Will changes in soil organic carbon act as a positive or negative feedback on global warming? *Biogeochemistry* 48, 21-51.
- Kort, J., Turnock, R., 1998. Carbon reservoir and biomass in Canadian prairie shelterbelts. *Agrofor. Syst.* 44, 175-186.
- Liebig, M., Morgan, J., Reeder, J., Ellert, B., Gollany, H., Schuman, G., 2005. Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada. *Soil Till. Res.* 83, 25-52.
- Maia, S.M.F., Xavier, F.A.S., Oliveira, T.S., Mendonça, E.S., Araújo Filho, J.A., 2007. Organic carbon pools in a Luvisol under agroforestry and conventional farming systems in the semi-arid region of Ceará, Brazil. *Agrofor. Syst.* 71, 127-138.
- Malhi, S.S., Nyborg, M., Solberg, E.D., McConkey, B., Dyck, M., Puurveen, D., 2011. Long-term straw management and N fertilizer rate effects on quantity and quality of organic C and N and some chemical properties in two contrasting soils in Western Canada. *Biol. Fert. Soils* 47, 785-800.
- Nyborg, M., Laidlaw, J., Solberg, E., Malhi, S., 1997. Denitrification and nitrous oxide emissions from a Black Chernozemic soil during spring thaw in Alberta. *Can. J. Soil Sci.* 77, 153-160.
- Pinno, B.D., Belanger, N., 2008. Ecosystem carbon gains from afforestation in the Boreal Transition ecozone of Saskatchewan (Canada) are coupled with the devolution of Black Chernozems. *Agric. Ecosyst. Environ.* 123, 56-62.

Schoeneberger, M.M., 2009. Agroforestry: working trees for sequestering carbon on agricultural lands. *Agrofor. Syst.* 75, 27-37.

Statistics Canada, 2011. Demographic Changes in Canadian Agriculture. Statistics Canada Catalogue no. 96-325-X. Ottawa. Version updated November 2014. [Online] <http://www5.statcan.gc.ca/subject-sujet/result-resultat?pid=920&id=-920&lang=eng&type=OLC&pageNum=1&more=0> [accessed 13 May 2015].

Winans, K.S., Whalen, J.K., Rivest, D., Cogliastro, A., Bradley, R.L., 2016. Carbon Sequestration and Carbon Markets for Tree-Based Intercropping Systems in Southern Quebec, Canada. *Atmosphere* 7, 17.

## Bibliography

- AARD, 2004. Alberta fertilizer guide [online] Available from [http://www1.agric.gov.ab.ca/\\$department/deptdocs.nsf/all/agdex3894](http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex3894) [accessed 29 December 2015]
- AARD, 2015. Agricultural carbon offsets: Information for Alberta's offset market [Online] Available from <http://worldwidewords.agriculture.alberta.ca/agcarbonoffsets> [accessed 10 November 2015].
- Adams, B.W., Ehlert, G., Stone, C., Alexander, M., Lawrence, D., Willoughby, D., Hincz, C., Burkinshaw, A., Carlson, J., France, K., 2009. Rangeland Health Assessment for Grassland, Forest and Tame Pasture. Alberta Sustainable Resources Department. Public Lands Division. Edmonton. AB. Pub. No. T/044.
- Abdalla, M., Jones, M., Smith, P., Williams, M., 2009. Nitrous oxide fluxes and denitrification sensitivity to temperature in Irish pasture soils. *Soil Use and Management* 25, 376-388.
- Agren, G.I., Bosatta, E., 2002. Reconciling differences in predictions of temperature response of soil organic matter. *Soil Biology and Biochemistry* 34, 129-132.
- Alam, M., Olivier, A., Paquette, A., Dupras, J., Revéret, J., Messier, C., 2014. A general framework for the quantification and valuation of ecosystem services of tree-based intercropping systems. *Agroforestry Systems* 88, 679-691.
- Alberta Agriculture and Rural Development, 2010. Alberta Fertilizer Guide [online]. [http://www1.agric.gov.ab.ca/\\$department/deptdocs.nsf/all/agdex3894/\\$file/541-1.pdf?OpenElement](http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex3894/$file/541-1.pdf?OpenElement) [site last visited in August 2013].

- Albrecht, A., Kandji, S.T., 2003. Carbon sequestration in tropical agroforestry systems. *Agriculture, Ecosystems & Environment* 99, 15-27.
- Aldy, J.E., Stavins, R.N., 2012. The promise and problems of pricing carbon: Theory and experience. *The Journal of Environment & Development* 1070496512442508.
- Altieri, M.A., 1999. The ecological role of biodiversity in agroecosystems. *Agriculture, Ecosystems & Environment* 74, 19-31.
- Alvarez, R., Alvarez, C., 2000. Soil organic matter pools and their associations with carbon mineralization kinetics. *Soil Science Society of America Journal* 64, 184-189.
- Amadi, C.C., Van Rees, K.C., Farrell, R.E., 2016. Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in shelterbelts compared with adjacent cropped fields. *Agriculture, Ecosystems & Environment* 223, 123-134.
- Amichev, B.Y., Bentham, M.J., Cerkowniak, D., Kort, J., Kulshreshtha, S., Laroque, C.P., Piwowar, J.M., Van Rees, K.C., 2015. Mapping and quantification of planted tree and shrub shelterbelts in Saskatchewan, Canada. *Agroforestry Systems* 89, 49-65.
- Amichev, B.Y., Johnston, M., Van Rees, K.C., 2010. Hybrid poplar growth in bioenergy production systems: Biomass prediction with a simple process-based model (3PG). *Biomass and Bioenergy* 34, 687-702.
- Amonette, J.E., Kim, J., Russell, C.K., 2004. Enhancement of soil carbon sequestration: A catalytic approach. *American Chemical Society* 49, 366.

- Arevalo, C.B., Bhatti, J.S., Chang, S.X., Sidders, D., 2009. Ecosystem carbon stocks and distribution under different land-uses in north central Alberta, Canada. *Forest Ecology and Management* 257, 1776-1785.
- Arevalo, C., Bhatti, J.S., Chang, S.X., Jassal, R.S., Sidders, D., 2010. Soil respiration in four different land use systems in north central Alberta, Canada. *Journal of Geophysical Research: Biogeosciences* 115, .
- Arevalo, C., Chang, S.X., Bhatti, J.S., Sidders, D., 2012. Mineralization potential and temperature sensitivity of soil organic carbon under different land uses in the parkland region of Alberta, Canada. *Soil Science Society of America Journal* 76, 241-251.
- Ayres, E., Steltzer, H., Berg, S., Wallenstein, M.D., Simmons, B.L., Wall, D.H., 2009. Tree species traits influence soil physical, chemical, and biological properties in high elevation forests. *Plos One* 4, e5964.
- Baah-Acheamfour, M., Carlyle, C.N., Bork, E.W., Chang, S.X., 2014. Trees increase soil carbon and its stability in three agroforestry systems in central alberta, canada. *Forest Ecology and Management* 328, 131-139.
- Baah-Acheamfour, M., Chang, S.X., Carlyle, C.N., Bork, E.W., 2015. Carbon pool size and stability are affected by trees and grassland cover types within agroforestry systems of western Canada. *Agriculture, Ecosystems & Environment* 213, 105-113.
- Baggs, E., 2006. Partitioning the components of soil respiration: A research challenge. *Plant and Soil* 284, 1-5.

- Baisden, W., Amundson, R., Cook, A., Brenner, D., 2002. Turnover and storage of C and N in five density fractions from California annual grassland surface soils. *Global Biogeochemical Cycles* 16, .
- Ball, B.C., Scott, A., Parker, J.P., 1999. Field N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> fluxes in relation to tillage, compaction and soil quality in Scotland. *Soil and Tillage Research* 53, 29-39.
- Bambrick, A.D., Whalen, J.K., Bradley, R.L., Cogliastro, A., Gordon, A.M., Olivier, A., Thevathasan, N.V., 2010. Spatial heterogeneity of soil organic carbon in tree-based intercropping systems in Québec and Ontario, Canada. *Agroforestry Systems* 79, 343-353.
- Banerjee, S., Baah-Acheamfour, M., Carlyle, C.N., Bissett, A., Richardson, A.E., Siddique, T., Bork, E.W., Chang, S.X., 2015. Determinants of bacterial communities in Canadian agroforestry systems. *Environmental microbiology* , 18, 1805–1816.
- Baron, V., Mapfumo, E., Dick, A., Naeth, M., Okine, E., Chanasyk, D., 2002. Grazing intensity impacts on pasture carbon and nitrogen flow. *Journal of Range Management* 535-541.
- Beauchemin, K., McGinn, S., 2006. Methane emissions from beef cattle: Effects of fumaric acid, essential oil, and canola oil. *Journal of animal science* 84, 1489-1496.
- Beauchemin, K.A., Janzen, H.H., Little, S.M., McAllister, T.A., McGinn, S.M., 2010. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. *Agricultural Systems* 103, 371-379.

- Beauchemin, K.A., McGinn, S.M., 2006. Enteric methane emissions from growing beef cattle as affected by diet and level of intake. *Canadian journal of animal science* 86, 401-408.
- Bergeron, M., Lacombe, S., Bradley, R.L., Whalen, J., Cogliastro, A., Jutras, M., Arp, P., 2011. Reduced soil nutrient leaching following the establishment of tree-based intercropping systems in eastern Canada. *Agroforestry Systems* 83, 321-330.
- Berndes, G., Hoogwijk, M., van den Broek, R., 2003. The contribution of biomass in the future global energy supply: A review of 17 studies. *Biomass and Bioenergy* 25, 1-28.
- Bhatti, J., Apps, M., Tarnocai, C., 2002. Estimates of soil organic carbon stocks in central Canada using three different approaches. *Canadian Journal of Forest Research* 32, 805-812.
- Binkley, D., Richter, D., 1987. Nutrient cycles and H budgets of forest ecosystems. *Advances in Ecological Research* 16, 1-51.
- Bond-Lamberty, B., Thomson, A., 2010. Temperature-associated increases in the global soil respiration record. *Nature* 464, 579-582.
- Bond-Lamberty, B., Wang, C., Gower, S.T., 2004. A global relationship between the heterotrophic and autotrophic components of soil respiration? *Global Change Biology* 10, 1756-1766.
- Boone, R.D., Nadelhoffer, K.J., Canary, J.D., Kaye, J.P., 1998. Roots exert a strong influence on the temperature sensitivity of soil respiration. *Nature* 396, 570-572.

- Borden, K.A., Isaac, M.E., Thevathasan, N.V., Gordon, A.M., Thomas, S.C., 2014. Estimating coarse root biomass with ground penetrating radar in a tree-based intercropping system. *Agroforestry Systems* 88, 657-669.
- Brady, N., Weil, R., 1999. *The nature and properties of soil* 12th ed.
- Bronick, C.J., Lal, R., 2005. Manuring and rotation effects on soil organic carbon concentration for different aggregate size fractions on two soils in northeastern Ohio, USA. *Soil and Tillage Research* 81, 239-252.
- Burke, I.C., Yonker, C., Parton, W., Cole, C., Schimel, D., Flach, K., 1989. Texture, climate, and cultivation effects on soil organic matter content in US grassland soils. *Soil Science Society of America Journal* 53, 800-805.
- Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., Zechmeister-Boltenstern, S., 2013. Nitrous oxide emissions from soils: How well do we understand the processes and their controls? *Philosophical transactions of the Royal Society of London. Series B, Biological Sciences* 368, 20130122.
- Cambardella, C., Elliott, E., 1994. Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. *Soil Science Society of America Journal* 58, 123-130.
- Campbell, C., Biederbeck, V., Wen, G., Zentner, R., Schoenau, J., Hahn, D., 1999. Seasonal trends in selected soil biochemical attributes: Effects of crop rotation in the semiarid prairie. *Canadian Journal of Soil Science* 79, 73-84.

- Cao, G., Tang, Y., Mo, W., Wang, Y., Li, Y., Zhao, X., 2004. Grazing intensity alters soil respiration in an alpine meadow on the Tibetan plateau. *Soil Biology and Biochemistry* 36, 237-243.
- Cardinael, R., Thevathasan, N., Gordon, A., Clinch, R., Mohammed, I., Sidders, D., 2012. Growing woody biomass for bioenergy in a tree-based intercropping system in southern Ontario, Canada. *Agroforestry Systems* 86, 279-286.
- Carter, M.R., Gregorich, E.G., 2006. *Soil Sampling and Methods of Analysis*, second ed. Francis & Taylor Group, LLC, CRC Press. Boca Raton, FL 334887–2742.
- Chanasyk, D.S., Naeth, M.A., 1995. Grazing impacts on bulk density and soil strength in the foothills fescue grasslands of Alberta, Canada. *Canadian Journal of Soil Science* 75, 551-557.
- Chen, Y., Li, Y., Awada, T., Han, J., Luo, Y., 2012. Carbon sequestration in the total and light fraction soil organic matter along a chronosequence in grazing exclosures in a semiarid degraded sandy site in china. *Journal of Arid Land* 4, 411-419.
- Christensen, B.T., 2001. Physical fractionation of soil and structural and functional complexity in organic matter turnover. *European Journal of Soil Science* 52, 345-353.
- Christiansen, J.R., Gundersen, P., 2011. Stand age and tree species affect N<sub>2</sub>O and CH<sub>4</sub> exchange from afforested soils. *Biogeosciences* 8, 2535-2546.
- Clary, J., 2012. Determinants of perennial and annual grass distribution in Mediterranean-climate California. *Plant Ecology* 213, 1203-1208.

- Clinch, R.L., Thevathasan, N.V., Gordon, A.M., Volk, T.A., Sidders, D., 2009. Biophysical interactions in a short rotation willow intercropping system in southern Ontario, Canada. *Agriculture, Ecosystems & Environment* 131, 61-69.
- Cole, C.V., Flach, K., Lee, J., Sauerbeck, D., Stewart, B., 1993. Agricultural Sources and Sinks of Carbon. In: Anonymous *Terrestrial Biospheric Carbon Fluxes Quantification of Sinks and Sources of CO<sub>2</sub>*. Springer, , pp. 111-122.
- Compton, J.E., Boone, R.D., 2000. Long-term impacts of agriculture on soil carbon and nitrogen in new England forests. *Ecology* 81, 2314-2330.
- Conant, R.T., Drijber, R.A., Haddix, M.L., Parton, W.J., Paul, E.A., Plante, A.F., Six, J., Steinweg, J.M., 2008. Sensitivity of organic matter decomposition to warming varies with its quality. *Global Change Biology* 14, 868-877.
- Costa, M.H., Foley, J.A., 2000. Combined effects of deforestation and doubled atmospheric CO<sub>2</sub> concentrations on the climate of Amazonia. *Journal of Climate* 13, 18-34.
- Creamer, C., Filley, T., Boutton, T., Oleynik, S., Kantola, I., 2011. Controls on soil carbon accumulation during woody plant encroachment: Evidence from physical fractionation, soil respiration, and  $\delta^{13}\text{C}$  of respired CO<sub>2</sub>. *Soil Biology and Biochemistry* 43, 1678-1687.
- Creamer, C.A., Filley, T.R., Boutton, T.W., 2013. Long-term incubations of size and density separated soil fractions to inform soil organic carbon decay dynamics. *Soil Biology and Biochemistry* 57, 496-503.

- Crossman, N.D., Burkhard, B., Nedkov, S., 2012. Quantifying and mapping ecosystem services. *International Journal of Biodiversity Science, Ecosystem Services & Management* 8, 1-4.
- Crow, S.E., Swanston, C.W., Lajtha, K., Brooks, J.R., Keirstead, H., 2007. Density fractionation of forest soils: Methodological questions and interpretation of incubation results and turnover time in an ecosystem context. *Biogeochemistry* 85, 69-90.
- Curiel Yuste, J., Baldocchi, D., Gershenson, A., Goldstein, A., Misson, L., Wong, S., 2007. Microbial soil respiration and its dependency on carbon inputs, soil temperature and moisture. *Global Change Biology* 13, 2018-2035.
- Curiel Yuste, J., Janssens, I.A., Carrara, A., Meiresonne, L., Ceulemans, R., 2003. Interactive effects of temperature and precipitation on soil respiration in a temperate maritime pine forest. *Tree physiology* 23, 1263-1270.
- d'Annunzio, R., Conche, S., Landais, D., Saint-André, L., Joffre, R., Barthes, B.G., 2008. Pairwise comparison of soil organic particle-size distributions in native savannas and eucalyptus plantations in Congo. *Forest Ecology and Management* 255, 1050-1056.
- Dalal, R., Mayer, R., 1986. Long term trends in fertility of soils under continuous cultivation and cereal cropping in southern Queensland. IV. loss of organic carbon from different density functions. *Soil Research* 24, 301-309.
- Davidson, E.A., Janssens, I.A., 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature* 440, 165-173.

- Davidson, E., Belk, E., Boone, R.D., 1998. Soil water content and temperature as independent or confounded factors controlling soil respiration in a temperate mixed hardwood forest. *Global Change Biology* 4, 217-227.
- de Dieu Nzila, J., Bouillet, J., Laclau, J., Ranger, J., 2002. The effects of slash management on nutrient cycling and tree growth in eucalyptus plantations in the Congo. *Forest Ecology and Management* 171, 209-221.
- Desjardins, R., Kulshreshtha, S., Junkins, B., Smith, W., Grant, B., Boehm, M., 2001. Canadian greenhouse gas mitigation options in agriculture. *Nutrient Cycling in Agroecosystems* 60, 317-326.
- Desjardins, R., Smith, W., Grant, B., Campbell, C., Riznek, R., 2005. Management strategies to sequester carbon in agricultural soils and to mitigate greenhouse gas emissions. *Climatic Change* 70, 283-297.
- Dixon, R., 1995. Agroforestry systems: Sources of sinks of greenhouse gases? *Agroforestry Systems* 31, 99-116.
- Don, A., Scholten, T., Schulze, E., 2009. Conversion of cropland into grassland: Implications for soil organic-carbon stocks in two soils with different texture. *Journal of Plant Nutrition and Soil Science* 172, 53-62.
- Donkor, N., Gedir, J., Hudson, R., Bork, E., Chanasyk, D., Naeth, M., 2002. Impacts of grazing systems on soil compaction and pasture production in Alberta. *Canadian Journal of Soil Science* 82, 1-8.

- Dorodnikov, M., Kuzyakov, Y., Fangmeier, A., Wiesenberg, G.L., 2011. C and N in soil organic matter density fractions under elevated atmospheric CO<sub>2</sub>: Turnover vs. stabilization. *Soil Biology and Biochemistry* 43, 579-589.
- Dougherty, M.C., Thevathasan, N.V., Gordon, A.M., Lee, H., Kort, J., 2009. Nitrate and escherichia coli NAR analysis in tile drain effluent from a mixed tree intercrop and monocrop system. *Agriculture, Ecosystems & Environment* 131, 77-84.
- Duncan, K., 2008. Agricultural practices that reduce greenhouse gases (GHGs) and generate co-benefits. *WIT Transactions on Ecology and the Environment* 110.
- Duval, M.E., Galantini, J.A., Iglesias, J.O., Canelo, S., Martinez, J.M., Wall, L., 2013. Analysis of organic fractions as indicators of soil quality under natural and cultivated systems. *Soil and Tillage Research* 131, 11-19.
- Elliott, E., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. *Soil Science Society of America Journal* 50, 627-633.
- Environment Canada. 2012. Alberta Weather Condition [online]  
[http://weather.gc.ca/forecast/canada/index\\_e.html?id=AB](http://weather.gc.ca/forecast/canada/index_e.html?id=AB), [site last visited in May 2013]
- Environment Canada. 2013. Canada's emission trend. [online] <http://www.ec.gc.ca/indicateurs-indicators/default.asp?lang=en&n=CCED3397-1> [accessed 18 May 2014].
- Environment Canada. 2015. Alberta weather condition [online]  
[http://weather.gc.ca/forecast/canada/index\\_e.html?id=AB](http://weather.gc.ca/forecast/canada/index_e.html?id=AB), [accessed in February 2015]

- Epron, D., Marsden, C., M'Bou, A.T., Saint-André, L., d'Annunzio, R., Nouvellon, Y., 2009. Soil carbon dynamics following afforestation of a tropical savannah with eucalyptus in Congo. *Plant and Soil* 323, 309-322.
- Epstein, H., Lauenroth, W., Burke, I., 1997. Effects of temperature and soil texture on ANPP in the US great plains. *Ecology* 78, 2628-2631.
- Evers, A., Bambrick, A., Lacombe, S., Dougherty, M.C., Peichl, M., Gordon, A.M., Thevathasan, N.V., Whalen, J., Bradley, R.L., 2010. Potential greenhouse gas mitigation through temperate tree-based intercropping systems. *Open Agric.J* 4, 49-57.
- Fang, C., Smith, P., Smith, J., 2006. Is resistant soil organic matter more sensitive to temperature than the labile organic matter? *Biogeosciences* 3, 65-68.
- FAO (2014) FAOSTAT Emissions Database [online] <http://www.wri.org/resources/datasets/cait-country-greenhouse-gas-emissions-data>, [accessed in February 2015].
- Fierer, N., Craine, J.M., McLauchlan, K., Schimel, J.P., 2005. Litter quality and the temperature sensitivity of decomposition. *Ecology* 86, 320-326.
- Follett, R.F., Kimble, J.M., 2000. *The Potential of US Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect*. CRC Press, .
- Fontaine, S., Barot, S., Barré, P., Bdioui, N., Mary, B., Rumpel, C., 2007. Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature* 450, 277-280.

- Forster, Piers M de F, Shine, K.P., Stuber, N., 2006. It is premature to include non-CO<sub>2</sub> effects of aviation in emission trading schemes. *Atmospheric Environment* 40, 1117-1121.
- Fortier, J., Gagnon, D., Truax, B., Lambert, F., 2010. Biomass and volume yield after 6 years in multiclonal hybrid poplar riparian buffer strips. *Biomass and Bioenergy* 34, 1028-1040.
- Fortier, J., Truax, B., Gagnon, D., Lambert, F., 2015. Biomass carbon, nitrogen and phosphorus stocks in hybrid poplar buffers, herbaceous buffers and natural woodlots in the riparian zone on agricultural land. *Journal of environmental management* 154, 333-345.
- Fortier, J., Truax, B., Gagnon, D., Lambert, F., 2013. Mature hybrid poplar riparian buffers along farm streams produce high yields in response to soil fertility assessed using three methods. *Sustainability* 5, 1893-1916.
- Frame, J., 2005. *Forage Legumes for Temperate Grasslands*. Science Publishers Inc., Enfield, NH, USA. 309 pp.
- Freedman, B., Keith, T., 1996. Planting trees for carbon credits: A discussion of context, issues, feasibility, and environmental benefits. *Environmental Reviews* 4, 100-111.
- Gacengo, C.N., Wood, C.W., Shaw, J.N., Raper, R.L., Balkcom, K.S., 2009. Agroecosystem management effects on greenhouse gas emissions across a coastal plain catena. *Soil Science* 174, 229-237.
- Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B., Cosby, B.J., 2003. The nitrogen cascade. *Bioscience* 53, 341-356.

- Gebauer, R.L., Ehleringer, J.R., 2000. Water and nitrogen uptake patterns following moisture pulses in a cold desert community. *Ecology* 81, 1415-1424.
- Golluscio, R.A., Austin, A.T., Martínez, G.C.G., Gonzalez-Polo, M., Sala, O.E., Jackson, R.B., 2009. Sheep grazing decreases organic carbon and nitrogen pools in the patagonian steppe: Combination of direct and indirect effects. *Ecosystems* 12, 686-697.
- Gomez-Casanovas, N., Matamala, R., Cook, D.R., Gonzalez-Meler, M.A., 2012. Net ecosystem exchange modifies the relationship between the autotrophic and heterotrophic components of soil respiration with abiotic factors in prairie grasslands. *Global Change Biology* 18, 2532-2545.
- Gordon, A., Thevathasan, N., 2005. How much carbon can be stored in canadian agroecosystems using a silvopastoral approach. *Silvopastoralism and sustainable land management*. CABI Publishing, Wallingford, UK 210-218.
- Goulden, M.L., Munger, J.W., FAN, S., Daube, B.C., Wofsy, S.C., 1996. Measurements of carbon sequestration by long-term eddy covariance: Methods and a critical evaluation of accuracy. *Global Change Biology* 2, 169-182.
- Government of Alberta. 2013. Central parkland range plant community guide. Range plant communities and range health assessment guidelines for the central parkland subregion of Alberta. Second approximation, Pub. No. T/125. Prepared by Kupsch, T., France, K., Loonen, H., Burkinshaw, A., Willoughby, M.G., and Ron L. McNeil Landwise Inc., Range Management Branch, Alberta Environment and Sustainable Resource Development, Edmonton, AB.

- Grant, C., Lafond, G., 1993. The effects of tillage systems and crop sequences on soil bulk density and penetration resistance on a clay soil in southern Saskatchewan. *Canadian Journal of Soil Science* 73, 223-232.
- Green, S., Vogeler, I., Clothier, B., Mills, T., Van Den Dijssel, C., 2003. Modelling water uptake by a mature apple tree. *Soil Research* 41, 365-380.
- Gupta, N., Kukal, S., Bawa, S., Dhaliwal, G., 2009. Soil organic carbon and aggregation under poplar based agroforestry system in relation to tree age and soil type. *Agroforestry Systems* 76, 27-35.
- Haile, S.G., Nair, P., Nair, V.D., 2008. Carbon storage of different soil-size fractions in Florida silvopastoral systems. *Journal of environmental quality* 37, 1789-1797.
- Hanson, P., Edwards, N., Garten, C., Andrews, J., 2000. Separating root and soil microbial contributions to soil respiration: A review of methods and observations. *Biogeochemistry* 48, 115-146.
- Harris, D., Horwáth, W.R., van Kessel, C., 2001. Acid fumigation of soils to remove carbonates prior to total organic carbon or carbon-13 isotopic analysis. *Soil Science Society of America Journal* 65, 1853-1856.
- Hassink, J., 1997. The capacity of soils to preserve organic C and N by their association with clay and silt particles. *Plant and Soil* 191, 77-87.
- Heitschmidt, R.K., Stuth, J., 1991. *Grazing management, an ecological perspective*. Timber Press, Inc. Portland, Oregon. 18 pp.

- Högberg, P., Nordgren, A., Buchmann, N., Taylor, A.F., Ekblad, A., Högberg, M.N., Nyberg, G., Ottosson-Löfvenius, M., Read, D.J., 2001. Large-scale forest girdling shows that current photosynthesis drives soil respiration. *Nature* 411, 789-792.
- Horwath, W.R., Pregitzer, K.S., Paul, E.A., 1994. 14C allocation in tree-soil systems. *Tree physiology* 14, 1163-1176.
- Howlett, D.S., Mosquera-Losada, M.R., Nair, P., Nair, V.D., Rigueiro-Rodríguez, A., 2011. Soil carbon storage in silvopastoral systems and a treeless pasture in northwestern Spain. *Journal of environmental quality* 40, 825-832..
- Huntington, T., Johnson, C., Johnson, A., Siccama, T., Ryan, D., 1989. Carbon, organic matter, and bulk density relationships in a forested spodosol. *Soil Science* 148, 380-386.
- Hutchinson, J., Campbell, C., Desjardins, R., 2007. Some perspectives on carbon sequestration in agriculture. *Agricultural and Forest Meteorology* 142, 288-302.
- IPCC. 2000. Land-use, land-use change, and forestry. A special report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.
- IPCC. 2013. Climate change: The physical science basis. Working group I contribution to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, United Kingdom.
- Isaac, M.E., Carlsson, G., Ghoulam, C., Makhani, M., Thevathasan, N.V., Gordon, A.M., 2014. Legume performance and nitrogen acquisition strategies in a tree-based agroecosystems. *Agroecology and sustainable food systems* 38, 686-703.

- Isaac, M.E., Harmand, J., Lesueur, D., Lelon, J., 2011. Tree age and soil phosphorus conditions influence N<sub>2</sub>-fixation rates and soil N dynamics in natural populations of acacia Senegal. *Forest Ecology and Management* 261, 582-588.
- Issah, G., Kimaro, A., Kort, J., Knight, J., 2014. Quantifying biological nitrogen fixation of agroforestry shrub species using <sup>15</sup>N dilution techniques under greenhouse conditions. *Agroforestry Systems* 88, 607-617.
- Janssens, I., Carrara, A., Ceulemans, R., 2004. Annual Q<sub>10</sub> of soil respiration reflects plant phenological patterns as well as temperature sensitivity. *Global Change Biology* 10, 161-169.
- Janssens, I., Lankreijer, H., Matteucci, G., Kowalski, A., Buchmann, N., Epron, D., Pilegaard, K., Kutsch, W., Longdoz, B., Grünwald, T., 2001. Productivity overshadows temperature in determining soil and ecosystem respiration across European forests. *Global Change Biology* 7, 269-278.
- Jastrow, J., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. *Soil Biology and Biochemistry* 28, 665-676.
- Jensen, L.S., Salo, T., Palmason, F., Breland, T.A., Henriksen, T.M., Stenberg, B., Pedersen, A., Lundström, C., Esala, M., 2005. Influence of biochemical quality on C and N mineralization from a broad variety of plant materials in soil. *Plant and Soil* 273, 307-326.

- Jobbágy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications* 10, 423-436.
- John, B., Yamashita, T., Ludwig, B., Flessa, H., 2005. Storage of organic carbon in aggregate and density fractions of silty soils under different types of land use. *Geoderma* 128, 63-79.
- Johnson, D.W., 1992. Effects of Forest Management on Soil Carbon Storage. In: Anonymous Natural Sinks of CO<sub>2</sub>. Springer, , pp. 83-120.
- Johnson, D.W., Curtis, P.S., 2001. Effects of forest management on soil C and N storage: Meta analysis. *Forest Ecology and Management* 140, 227-238.
- Jose, S., 2009. Agroforestry for ecosystem services and environmental benefits: An overview. *Agroforestry Systems* 76, 1-10.
- Kaiser, E., Kohrs, K., Kücke, M., Schnug, E., Heinemeyer, O., Munch, J., 1998. Nitrous oxide release from arable soil: Importance of N-fertilization, crops and temporal variation. *Soil Biology and Biochemistry* 30, 1553-1563.
- Kalra, Y.P., Maynard, D.C., 1991. *Methods Manual for Forest Soil and Plant Analysis*. Information Report - Northwest Region, Forestry Canada viii + 116 pp.
- Kang, B., 1997. Alley cropping—soil productivity and nutrient recycling. *Forest Ecology and Management* 91, 75-82.

- Kang, B., Caveness, F., Tian, G., Kolawole, G., 1999. Longterm alley cropping with four hedgerow species on an alfisol in southwestern Nigeria—effect on crop performance, soil chemical properties and nematode population. *Nutrient Cycling in Agroecosystems* 54, 145-155.
- Kass, D.C., Sylvester-Bradley, R., Nygren, P., 1997. The role of nitrogen fixation and nutrient supply in some agroforestry systems of the Americas. *Soil Biology and Biochemistry* 29, 775-785.
- Kebreab, E., Clark, K., Wagner-Riddle, C., France, J., 2006. Methane and nitrous oxide emissions from Canadian animal agriculture: A review. *Canadian Journal of Animal Science* 86, 135-157.
- Kelting, D.L., Burger, J.A., Edwards, G.S., 1998. Estimating root respiration, microbial respiration in the rhizosphere, and root-free soil respiration in forest soils. *Soil Biology and Biochemistry* 30, 961-968.
- Kirschbaum, M.U., 2000. Will changes in soil organic carbon act as a positive or negative feedback on global warming? *Biogeochemistry* 48, 21-51.
- Kleber, M., Mertz, C., Zikeli, S., Knicker, H., Jahn, R., 2004. Changes in surface reactivity and organic matter composition of clay subfractions with duration of fertilizer deprivation. *European Journal of Soil Science* 55, 381-391.
- Knorr, W., Prentice, I., House, J., Holland, E., 1998. Long-term sensitivity of soil carbon turnover to warming. *Paleoceanography* 13, 150-160.

- Kort, J., Turnock, R., 1998. Carbon reservoir and biomass in Canadian prairie shelterbelts. *Agroforestry Systems* 44, 175-186.
- Kort, J., Richardson, J., Soolanayakanahally, R., Schroeder, W., 2014. Innovations in temperate agroforestry: The 13th north American agroforestry conference. *Agroforestry Systems* 88, 563-567.
- Kulshreshtha, S.N., 2010. Agroforestry development on the Canadian prairies.
- Kuzyakov, Y., Gavrichkova, O., 2010. REVIEW: Time lag between photosynthesis and carbon dioxide efflux from soil: A review of mechanisms and controls. *Global Change Biology* 16, 3386-3406.
- Labrecque, M., Teodorescu, T.I., 2005. Field performance and biomass production of 12 willow and poplar clones in short-rotation coppice in southern Québec (Canada). *Biomass and Bioenergy* 29, 1-9.
- Lafond, G.P., Walley, F., May, W., Holzapfel, C., 2011. Long term impact of no-till on soil properties and crop productivity on the Canadian prairies. *Soil and Tillage Research* 117, 110-123.
- Lal R (2004) Soil carbon sequestration to mitigate climate change. *Geoderma*, 123, 1-22.
- Lal, R., 2009. Challenges and opportunities in soil organic matter research. *European Journal of Soil Science* 60, 158-169.

- Lal, R., 2002. Soil carbon dynamics in cropland and rangeland. *Environmental pollution* 116, 353-362.
- Lal, R., 2001. World cropland soils as a source or sink for atmospheric carbon. *Advances in Agronomy* 71, 145-191.
- Lambert M, Fréchet J (2005) Analytical techniques for measuring fluxes of CO<sub>2</sub> and CH<sub>4</sub> from hydroelectric reservoirs and natural water bodies. In: *Greenhouse Gas Emissions—Fluxes and Processes*. pp. 37-60. Springer,
- LaRade, S., Bork, E., 2011. Short communication: Aspen forest overstory relations to understory production. *Canadian Journal of Plant Science* 91, 847-851.
- Lee, N., Koo, J., Noh, N.J., Kim, J., Son, Y., 2010. Autotrophic and heterotrophic respiration in needle fir and quercus-dominated stands in a cool-temperate forest, central Korea. *Journal of Plant Research* 123, 485-495.
- Lenka, N.K., Dass, A., Sudhishri, S., Patnaik, U., 2012. Soil carbon sequestration and erosion control potential of hedgerows and grass filter strips in sloping agricultural lands of eastern India. *Agriculture, Ecosystems & Environment* 158, 31-40.
- Li, Y., Zhang, J., Chang, S.X., Jiang, P., Zhou, G., Shen, Z., Wu, J., Lin, L., Wang, Z., Shen, M., 2014. Converting native shrub forests to chinese chestnut plantations and subsequent intensive management affected soil C and N pools. *Forest Ecology and Management* 312, 161-169.

- Liebig, M., Morgan, J., Reeder, J., Ellert, B., Gollany, H., Schuman, G., 2005. Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada. *Soil and Tillage Research* 83, 25-52.
- Liu, T., McConkey, B., Huffman, T., Smith, S., MacGregor, B., Yemshanov, D., Kulshreshtha, S., 2014. Potential and impacts of renewable energy production from agricultural biomass in Canada. *Applied Energy* 130, 222-229.
- Liu, X.J., Mosier, A.R., Halvorson, A.D., Zhang, F.S., 2006. The impact of nitrogen placement and tillage on NO, N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> fluxes from a clay loam soil. *Plant and Soil* 280, 177-188.
- Lutes, K., Oelbermann, M., Thevathasan, N., Gordon, A., Effect of nitrogen fertilizer on greenhouse gas emissions in two willow clones (*salix miyabeana* and *S. dasyclados*) in southern Ontario, Canada. *Agroforestry Systems* 1-12.
- Lützw, M.v., Kögel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner, B., Flessa, H., 2006. Stabilization of organic matter in temperate soils: Mechanisms and their relevance under different soil conditions—a review. *European Journal of Soil Science* 57, 426-445.
- MacDonald, J., Skiba, U., Sheppard, L., Ball, B., Roberts, J., Smith, K., Fowler, D., 1997. The effect of nitrogen deposition and seasonal variability on methane oxidation and nitrous oxide emission rates in an upland spruce plantation and moorland. *Atmospheric Environment* 31, 3693-3706.

- Mackay, W.C., T.G. Miller, D.R. Moore and R. Woelcke. 1999. Performance characteristics of aeration devices. Prairie Farm Rehabilitation Administration. Prairie Farm Administration Paper, Peace River, AB, Canada.
- Maia, S.M.F., Xavier, F.A.S., Oliveira, T.S., Mendonça, E.S., Araújo Filho, J.A., 2007. Organic carbon pools in a luvisol under agroforestry and conventional farming systems in the semi-arid region of Ceará, Brazil. *Agroforestry Systems* 71, 127-138.
- Malhi, S.S., Nyborg, M., Solberg, E.D., McConkey, B., Dyck, M., Puurveen, D., 2011. Long-term straw management and N fertilizer rate effects on quantity and quality of organic C and N and some chemical properties in two contrasting soils in western Canada. *Biology and Fertility of Soils* 47, 785-800.
- Mann, L., 1986. Changes in soil carbon storage after cultivation. *Soil Science* 142, 279-288.
- Martins, M.R., Jantalia, C.P., Polidoro, J.C., Batista, J.N., Alves, B.J., Boddey, R.M., Urquiaga, S., 2015. Nitrous oxide and ammonia emissions from N fertilization of maize crop under no-till in a cerrado soil. *Soil and Tillage Research* 151, 75-81.
- Matthews, S., Pease, S., Gordon, A., Williams, P., 1993. Landowner perceptions and the adoption of agroforestry practices in southern Ontario, Canada. *Agroforestry Systems* 21, 159-168.
- Mead, D.J., Comforth, I.S., Eds., 1995. Proceedings of the Tree and Soil Workshop, Lincoln University 28 February–2 March 1994. Agronomy Society of New Zealand Special Publication No. 10. Lincoln University Press, Canterbury.

- Meinzer, F.C., Goldstein, G., Andrade, J.L., 2001. Regulation of water flux through tropical forest canopy trees: Do universal rules apply? *Tree physiology* 21, 19-26.
- Moni, C., Derrien, D., Hatton, P., Zeller, B., Kleber, M., 2012. Density fractions versus size separates: Does physical fractionation isolate functional soil compartments? *Biogeosciences* 9, 5181–5197.
- Montagnini, F., Nair, P., 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. *Agroforestry Systems* 61, 281-295.
- Mosier, A., Schimel, D., Valentine, D., Bronson, K., Parton, W., 1991. Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. *Nature* 350, 330-332.
- Mosier, A.R., Halvorson, A.D., Reule, C.A., Liu, X.J., 2006. Net global warming potential and greenhouse gas intensity in irrigated cropping systems in northeastern Colorado. *Journal of environmental quality* 35, 1584-1598.
- Mu, Z., Huang, A., Ni, J., Li, J., Liu, Y., Shi, S., Xie, D., Hatano, R., 2013. Soil greenhouse gas fluxes and net global warming potential from intensively cultivated vegetable fields in southwestern china. *Journal of soil science and plant nutrition* 13, 566-578.
- Mujuru, L., Mureva, A., Velthorst, E., Hoosbeek, M., 2013. Land use and management effects on soil organic matter fractions in rhodic ferralsols and haplic arenosols in bindura and shamva districts of Zimbabwe. *Geoderma* 209, 262-272.
- Müller, T., Walter, B., Wirtz, A., Burkovski, A., 2006. Ammonium toxicity in bacteria. *Current microbiology* 52, 400-406.

- Munroe, J.W., Isaac, M.E., 2014. N<sub>2</sub>-fixing trees and the transfer of fixed-N for sustainable agroforestry: A review. *Agronomy for sustainable development* 34, 417-427.
- Mutuo, P.K., Cadisch, G., Albrecht, A., Palm, C., Verchot, L., 2005. Potential of agroforestry for carbon sequestration and mitigation of greenhouse gas emissions from soils in the tropics. *Nutrient Cycling in Agroecosystems* 71, 43-54.
- Myhre G, Shindell D, Breon F-M et al. (2013) Anthropogenic and Natural Radiative Forcing. In: *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Stocker TF, Qin D, Plattner G-K, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM), pp. 659–740. Cambridge University Press,
- Naeth, M., Bailey, A., Pluth, D., Chanasyk, D., Hardin, R., 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland ecosystems of Alberta. *Journal of Range Management* 7-12.
- Nair, P.K.R., 2011. Agroforestry systems and environmental quality: Introduction. *Journal of environmental quality* 40, 784-790.
- Nair, P.K.R., Buresh, R.J., Mugendi, D.N., Latt, C.R., 1998. Nutrient cycling in tropical agroforestry systems: Myths and Science, in: Buck , L.E., Lassoie, J.P., Fernandes, E.C.M. (eds), *Agroforestry in Sustainable Agricultural Systems*. CRC Press

- Nair, P.K.R., Nair, V.D., Kumar, B.M., Haile, S.G., 2009. Soil carbon sequestration in tropical agroforestry systems: A feasibility appraisal. *Environmental Science & Policy* 12, 1099-1111.
- Nair, P.K.R., Nair, V.D., Kumar, B.M., Showalter, J.M., 2010. Chapter five-carbon sequestration in agroforestry systems. *Advances in Agronomy* 108, 237-307.
- Nakane, K., Yamamoto, M., Tsubota, H., 1983. Estimation of root respiration rate in a mature forest ecosystem. *Nihon Seitai Gakkaishi* 33, 397-408.
- Nascente, A.S., Li, Y.C., Crusciol, C.A.C., 2013. Cover crops and no-till effects on physical fractions of soil organic matter. *Soil and Tillage Research* 130, 52-57.
- Nason, G., Pluth, D., McGill, W., 1988. Volatilization and foliar recapture of ammonia following spring and fall application of nitrogen-15 urea to a douglas-fir ecosystem. *Soil Science Society of America Journal* 52, 821-828.
- Natural Resources Canada. 2014. Energy sources and distribution. [online]  
<http://www.nrcan.gc.ca/energy/alternative-fuels/programs/12358> [accessed 20 April 2016].
- Nelson, R.G., 2002. Resource assessment and removal analysis for cornstover and wheat straw in the eastern and midwestern united States—rainfall and wind-induced soil erosion methodology. *Biomass and Bioenergy* 22, 349-363.

- Nissim, W.G., Pitre, F.E., Teodorescu, T.I., Labrecque, M., 2013. Long-term biomass productivity of willow bioenergy plantations maintained in southern Québec, Canada. *Biomass and Bioenergy* 56, 361-369.
- Nyborg, M., Laidlaw, J., Solberg, E., Malhi, S., 1997. Denitrification and nitrous oxide emissions from a black chernozemic soil during spring thaw in Alberta. *Canadian Journal of Soil Science* 77, 153-160.
- Oades, J.M., 1984. Soil organic matter and structural stability: Mechanisms and implications for management. *Plant and Soil* 76, 319-337.
- Oelbermann, M., Gordon, A.M., 2000. Quantity and quality of autumnal litterfall into a rehabilitated agricultural stream. *Journal of environmental quality* 29, 603-611.
- Oelbermann, M., Raimbault, B.A., 2015. Riparian land-use and rehabilitation: Impact on organic matter input and soil respiration. *Environmental management* 55, 496-507.
- Oelbermann, M., Voroney, R.P., 2011. An evaluation of the century model to predict soil organic carbon: Examples from Costa Rica and Canada. *Agroforestry Systems* 82, 37-50.
- Oelbermann, M., Voroney, R.P., Gordon, A.M., 2004. Carbon sequestration in tropical and temperate agroforestry systems: A review with examples from Costa Rica and southern Canada. *Agriculture, Ecosystems & Environment* 104, 359-377.
- Oelbermann, M., Voroney, R.P., Thevathasan, N.V., Gordon, A.M., Kass, D.C., Schlönvoigt, A.M., 2006. Soil carbon dynamics and residue stabilization in a Costa Rican and southern Canadian alley cropping system. *Agroforestry Systems* 68, 27-36.

- Ohashi, M., Gyokusen, K., Saito, A., 2000. Contribution of root respiration to total soil respiration in a Japanese cedar (*cryptomeria japonica* D. don) artificial forest. *Ecological Research* 15, 323-333.
- Pang, X., Bao, W., Zhu, B., Cheng, W., 2013. Responses of soil respiration and its temperature sensitivity to thinning in a pine plantation. *Agricultural and Forest Meteorology* 171, 57-64.
- Partey, S.T., 2011. Effect of pruning frequency and pruning height on the biomass production of *tithonia diversifolia* (hemsl) A. gray. *Agroforestry Systems* 83, 181-187.
- Paul, E., Follett, R., Leavitt, S., Halvorson, A., Peterson, G., Lyon, D., 1997. Radiocarbon dating for determination of soil organic matter pool sizes and dynamics. *Soil Science Society of America Journal* 61, 1058-1067.
- Paul, K., Polglase, P., Nyakuengama, J., Khanna, P., 2002. Change in soil carbon following afforestation. *Forest Ecology and Management* 168, 241-257.
- Paustian, K., Six, J., Elliott, E., Hunt, H., 2000. Management options for reducing CO<sub>2</sub> emissions from agricultural soils. *Biogeochemistry* 48, 147-163.
- Paustian, K., Cole, C.V., Sauerbeck, D., Sampson, N., 1998. CO<sub>2</sub> mitigation by agriculture: An overview. *Climatic Change* 40, 135-162.
- Peichl, M., Thevathasan, N.V., Gordon, A.M., Huss, J., Abohassan, R.A., 2006. Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. *Agroforestry Systems* 66, 243-257.

- PFRA. 2000. Prairie agricultural landscapes: a land resource review. ISBN 0-622-28574-3.  
Minister of Public Works and Government Services 2000, 179 pp
- Pinno, B.D., Belanger, N., 2008. Ecosystem carbon gains from afforestation in the boreal transition ecozone of saskatchewan (canada) are coupled with the devolution of black chernozems. *Agriculture, Ecosystems & Environment* 123, 56-62.
- Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: Processes and potential. *Global Change Biology* 6, 317-327.
- Potter, K., Daniel, J., Altom, W., Torbert, H., 2001. Stocking rate effect on soil carbon and nitrogen in degraded soils. *Journal of Soil and Water Conservation* 56, 233-236.
- Powell, G.W., Bork, E.W., 2006. Aspen canopy removal and root trenching effects on understory vegetation. *Forest Ecology and Management* 230, 79-90.
- Prescott, C.E., Blevins, L.L., Staley, C., 2004. Litter decomposition in British Columbia forests: Controlling factors and influences of forestry activities. *British Columbia Journal of Ecosystem Management* 5, 44-57.
- Raghoebarsing, A.A., Smolders, A.J., Schmid, M.C., Rijpstra, W.I.C., Wolters-Arts, M., Derksen, J., Jetten, M.S., Schouten, S., Damsté, J.S.S., Lamers, L.P., 2005. Methanotrophic symbionts provide carbon for photosynthesis in peat bogs. *Nature* 436, 1153-1156.
- Raich, J.W., Potter, C.S., 1995. Global patterns of carbon dioxide emissions from soils. *Global Biogeochemical Cycles* 9, 23-36.

- Raich, J.W., Tufekciogul, A., 2000. Vegetation and soil respiration: Correlations and controls. *Biogeochemistry* 48, 71-90.
- Raich, J., Schlesinger, W.H., 1992. The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. *Tellus B* 44, 81-99.
- Ramachandran Nair, P., Mohan Kumar, B., Nair, V.D., 2009. Agroforestry as a strategy for carbon sequestration. *Journal of plant nutrition and soil science* 172, 10-23.
- Reeder, J., Schuman, G., 2002. Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands. *Environmental pollution* 116, 457-463.
- Reeves, S., Wang, W., 2015. Optimum sampling time and frequency for measuring N<sub>2</sub>O emissions from a rain-fed cereal cropping system. *Science of The Total Environment* 530, 219-226.
- Rivest, D., Cogliastro, A., Bradley, R.L., Olivier, A., 2010. Intercropping hybrid poplar with soybean increases soil microbial biomass, mineral N supply and tree growth. *Agroforestry Systems* 80, 33-40.
- Rowe, E.C., Van Noordwijk, M., Suprayogo, D., Cadisch, G., 2005. Nitrogen use efficiency of monoculture and hedgerow intercropping in the humid tropics. *Plant and Soil* 268, 61-74.
- Ruser, R., Flessa, H., Russow, R., Schmidt, G., Buegger, F., Munch, J., 2006. Emission of N<sub>2</sub>O, N<sub>2</sub> and CO<sub>2</sub> from soil fertilized with nitrate: Effect of compaction, soil moisture and rewetting. *Soil Biology and Biochemistry* 38, 263-274.

- Sainju, U.M., Barsotti, J.L., Wang, J., 2014. Net global warming potential and greenhouse gas intensity affected by cropping sequence and nitrogen fertilization. *Soil Science Society of America Journal* 78, 248-261.
- Sainju, U.M., Jabro, J.D., Stevens, W.B., 2008. Soil carbon dioxide emission and carbon content as affected by irrigation, tillage, cropping system, and nitrogen fertilization. *Journal of environmental quality* 37, 98-106.
- Sainju, U.M., Stevens, W.B., Caesar-TonThat, T., Liebig, M.A., 2012. Soil greenhouse gas emissions affected by irrigation, tillage, crop rotation, and nitrogen fertilization. *Journal of environmental quality* 41, 1774-1786.
- Sampson, D., Janssens, I., Curiel Yuste, J., Ceulemans, R., 2007. Basal rates of soil respiration are correlated with photosynthesis in a mixed temperate forest. *Global Change Biology* 13, 2008-2017.
- SAS Institute Inc., 2013. *SAS User's Guide: Statistics SAS/C Online Doc*, Release 9.30. SAS Inc., Cary, NC
- Saurette, D.D., Chang, S.X., Thomas, B.R., 2008. Autotrophic and heterotrophic respiration rates across a chronosequence of hybrid poplar plantations in northern Alberta. *Canadian Journal of Soil Science* 88, 261-272.
- Schnabel, R.R., Franzluebbbers, A.J., Stout, W.L., Sanderson, M.A., Stuedemann, J.A., 2001. The effects of pasture management practices. In: Follet, R.F., Kimble, J.M., and Lal, R.

- (eds.): The potential of U.S. grazing lands to sequester carbon and mitigate the greenhouse effect. 291–322 pp. Lewis Publishers, Boca Raton, FL.
- Schoeneberger, M.M., 2009. Agroforestry: Working trees for sequestering carbon on agricultural lands. *Agroforestry Systems* 75, 27-37.
- Schott, S., 2013. Carbon pricing options for Canada. *Canadian Public Policy* 39, S109-S124.
- Schroeder, P., 1994. Carbon storage benefits of agroforestry systems. *Agroforestry Systems* 27, 89-97.
- Sharrow, S., Ismail, S., 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. *Agroforestry Systems* 60, 123-130.
- Silva, J.P., Lasso, A., Lubberding, H.J., Peña, M.R., Gijzen, H.J., 2015. Biases in greenhouse gases static chambers measurements in stabilization ponds: Comparison of flux estimation using linear and non-linear models. *Atmospheric Environment* 109, 130-138.
- Simpson, J.A. 1999. Effects of shade on maize and soybean productivity in a tree-based intercropping system. PhD Thesis, University of Guelph, Guelph.
- Six, J., Conant, R., Paul, E.A., Paustian, K., 2002. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant and Soil* 241, 155-176.
- Six, J., Elliott, E., Paustian, K., Doran, J., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Science Society of America Journal* 62, 1367-1377.

- Six, J., Ogle, S.M., Conant, R.T., Mosier, A.R., Paustian, K., 2004. The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. *Global Change Biology* 10, 155-160.
- Six, J., Elliott, E., Paustian, K., 2000. Soil macroaggregate turnover and microaggregate formation: A mechanism for C sequestration under no-tillage agriculture. *Soil Biology and Biochemistry* 32, 2099-2103.
- Smith, P., Haberl, H., Popp, A., Erb, K., Lauk, C., Harper, R., Tubiello, F.N., Siqueira Pinto, A., Jafari, M., Sohi, S., 2013. How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? *Global Change Biology* 19, 2285-2302.
- Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J., 2008. Greenhouse gas mitigation in agriculture. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences* 363, 789-813.
- Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic matter: Mechanisms and controls. *Geoderma* 74, 65-105.
- Soil Classification Working Group. 1998. *The Canadian System of Soil Classification*. NRC Research Press, Ottawa, Canada, 187 pp.

- Son, Y., Kim, H., 1996. Soil respiration in pinus rigida and larix leptolepis plantations. Journal of Korean Forestry Society (Korea Republic) .
- Statistics Canada. 2011. Demographic Changes in Canadian Agriculture. Statistics Canada Catalogue no. 96-325-X. Ottawa. Version updated November 2014. [Online] <http://www5.statcan.gc.ca/subject-sujet/result-resultat?pid=920&id=-920&lang=eng&type=OLC&pageNum=1&more=0> [accessed 13 May 2015].
- Stolk, P., Jacobs, C., Moors, E., Hensen, A., Velthof, G., Kabat, P., 2009. Significant non-linearity in nitrous oxide chamber data and its effect on calculated annual emissions. Biogeosciences Discussions 6, 115-141.
- Subke, J., Inghima, I., Francesca Cotrufo, M., 2006. Trends and methodological impacts in soil CO<sub>2</sub> efflux partitioning: A metaanalytical review. Global Change Biology 12, 921-943.
- Swanston, C.W., Myrold, D.D., 1997. Incorporation of nitrogen from decomposing red alder leaves into plants and soil of a recent clearcut in Oregon. Canadian journal of forest research 27, 1496-1502.
- Syakila, A., Kroeze, C., 2011. The global nitrous oxide budget revisited. Greenhouse Gas Measurement and Management 1, 17-26.
- Takimoto, A., Nair, P.R., Nair, V.D., 2008. Carbon stock and sequestration potential of traditional and improved agroforestry systems in the west African Sahel. Agriculture, Ecosystems & Environment 125, 159-166.

- Tang, J., Baldocchi, D.D., 2005. Spatial–temporal variation in soil respiration in an oak–grass savanna ecosystem in California and its partitioning into autotrophic and heterotrophic components. *Biogeochemistry* 73, 183-207.
- Teepe, R., Brumme, R., Beese, F., 2000. Nitrous oxide emissions from frozen soils under agricultural, fallow and forest land. *Soil Biology and Biochemistry* 32, 1807-1810.
- Teklay, T., Chang, S.X., 2008. Temporal changes in soil carbon and nitrogen storage in a hybrid poplar chronosequence in northern Alberta. *Geoderma* 144, 613-619.
- Thevathasan, N.V., Gordon, A.M., Bradley, R., Cogliastro, A., Folkard, P., Grant, R., Kort, J., Liggins, L., Njenga, F., Olivier, A., 2012. Agroforestry Research and Development in Canada: The Way Forward. In: Anonymous Agroforestry-the Future of Global Land use. Springer, , pp. 247-283.
- Thevathasan, N., Gordon, A., 1997. Poplar leaf biomass distribution and nitrogen dynamics in a poplar-barley intercropped system in southern Ontario, Canada. *Agroforestry Systems* 37, 79-90.
- Thevathasan, N., Gordon, A., Simpson, J., Reynolds, P., Price, G., Zhang, P., 2004. Biophysical and ecological interactions in a temperate tree-based intercropping system. *Journal of Crop Improvement* 12, 339-363.
- Tiessen, H., Stewart, J., 1983. Particle-size fractions and their use in studies of soil organic matter: II. cultivation effects on organic matter composition in size fractions. *Soil Science Society of America Journal* 47, 509-514.

- Tonneijck, F.H., van der Plicht, J., Jansen, B., Verstraten, J.M., Hooghiemstra, H., 2006. Radiocarbon dating of soil organic matter fractions in andosols in northern Ecuador. *Radiocarbon* 48, 337.
- Udawatta, R.P., Jose, S., 2011. Carbon Sequestration Potential of Agroforestry Practices in Temperate North America. In: Anonymous Carbon Sequestration Potential of Agroforestry Systems. Springer, , pp. 17-42.
- Valdivia, C., Barbieri, C., Gold, M.A., 2012. Between forestry and farming: Policy and environmental implications of the barriers to agroforestry adoption. *Canadian Journal of Agricultural Economics/Revue canadienne d'agroeconomie* 60, 155-175.
- Van den Pol-van Dasselaar, A, Van Beusichem, M., Oenema, O., 1998. Effects of soil moisture content and temperature on methane uptake by grasslands on sandy soils. *Plant and Soil* 204, 213-222.
- Van Der Heijden, Marcel GA, Bardgett, R.D., Van Straalen, N.M., 2008. The unseen majority: Soil microbes as drivers of plant diversity and productivity in terrestrial ecosystems. *Ecology Letters* 11, 296-310.
- Vargas, R., Collins, S.L., Thomey, M.L., Johnson, J.E., Brown, R.F., Natvig, D.O., Friggens, M.T., 2012. Precipitation variability and fire influence the temporal dynamics of soil CO<sub>2</sub> efflux in an arid grassland. *Global Change Biology* 18, 1401-1411.
- Venterea, R.T., Halvorson, A.D., Kitchen, N., Liebig, M.A., Cavigelli, M.A., Del Grosso, S.J., Motavalli, P.P., Nelson, K.A., Spokas, K.A., Singh, B.P., 2012. Challenges and

- opportunities for mitigating nitrous oxide emissions from fertilized cropping systems. *Frontiers in Ecology and the Environment* 10, 562-570.
- von Lützow, M., Kögel-Knabner, I., Ekschmitt, K., Flessa, H., Guggenberger, G., Matzner, E., Marschner, B., 2007. SOM fractionation methods: Relevance to functional pools and to stabilization mechanisms. *Soil Biology and Biochemistry* 39, 2183-2207.
- Watson, C., Atkinson, D., Gosling, P., Jackson, L., Rayns, F., 2002. Managing soil fertility in organic farming systems. *Soil Use and Management* 18, 239-247.
- Willison, T., Webster, C., Goulding, K., Powlson, D., 1995. Methane oxidation in temperate soils: Effects of land use and the chemical form of nitrogen fertilizer. *Chemosphere* 30, 539-546.
- Winans, K.S., Whalen, J.K., Rivest, D., Cogliastro, A., Bradley, R.L., 2016. Carbon sequestration and carbon markets for tree-based intercropping systems in southern Québec, Canada. *Atmosphere* 7, 17.
- Winans, K., Whalen, J.K., Cogliastro, A., Rivest, D., Ribaud, L., 2014. Soil carbon stocks in two hybrid poplar-hay crop systems in southern Québec, Canada. *Forests* 5, 1952-1966.
- Witt, C., Cassman, K., Olk, D., Biker, U., Liboon, S., Samson, M., Ottow, J., 2000. Crop rotation and residue management effects on carbon sequestration, nitrogen cycling and productivity of irrigated rice systems. *Plant and Soil* 225, 263-278.

- Wotherspoon, A., Thevathasan, N.V., Gordon, A.M., Voroney, R.P., 2014. Carbon sequestration potential of five tree species in a 25-year-old temperate tree-based intercropping system in southern Ontario, Canada. *Agroforestry Systems* 88, 631-643.
- Yamashita, T., Flessa, H., John, B., Helfrich, M., Ludwig, B., 2006. Organic matter in density fractions of water-stable aggregates in silty soils: Effect of land use. *Soil Biology and Biochemistry* 38, 3222-3234.
- Yemshanov, D., McKenney, D., 2008. Fast-growing poplar plantations as a bioenergy supply source for Canada. *Biomass and Bioenergy* 32, 185-197.
- Young, A., 1997. *Agroforestry for Soil Management*. CAB international, 330 pp.
- Zhang, A., Cui, L., Pan, G., Li, L., Hussain, Q., Zhang, X., Zheng, J., Crowley, D., 2010. Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from tai lake plain, China. *Agriculture, Ecosystems & Environment* 139, 469-475.

**Appendix 2-1.** Soil carbon storage potential under agroforestry systems in different locations in Canada .

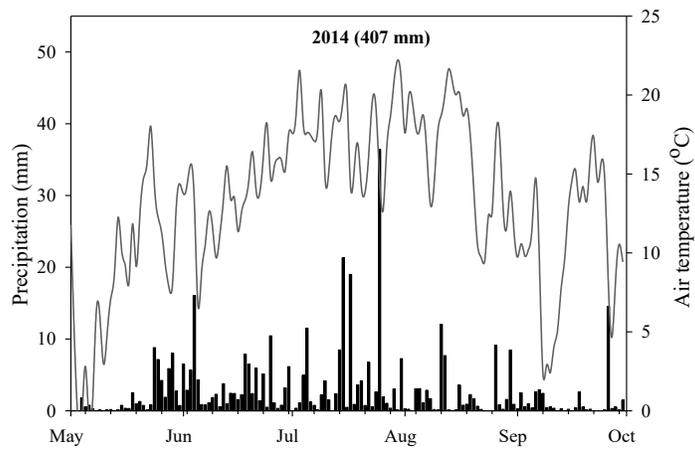
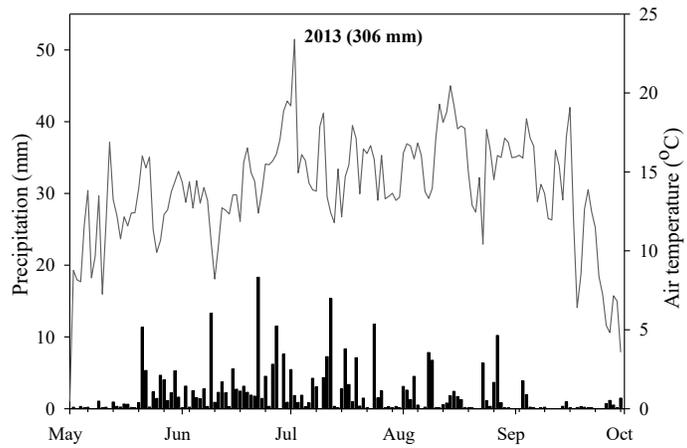
Agroforestry system	Location	Age (year)	Cover type	Species type	Depth (cm)	Soil C (Mg ha <sup>-1</sup> )	References	
Alley cropping	Ontario	13	Forest	Hybrid Poplar	0-20	78.5	Peichl <i>et al.</i> (2006)	
			Herbland	Norway Spruce		66		
				Barley monocrop		65		
	Ontario	21	Forest	Willow intercrop	0-40	46.6	Cardinael <i>et al.</i> (2012)	
			Herbland	Willow sole crop		43.7		
	Ontario	25	Forest	Herbland	Hybrid Poplar	0-40	86.8	Wotherspoon <i>et al.</i> (2014)
					Norway Spruce		78.3	
					Red oak		83.8	
					Black walnut		83.3	
					White cedar		76.8	
					Soybean monocrop		72.3	
	Ontario	18	Forest	Herbland	Hybrid poplar	0-20	101.5	Oelbermann <i>et al.</i> (2006)
					Crop monoculture		97.7	
	Ontario	18	Forest	Herbland	Hybrid poplar	0-40	125.3	
					Crop monoculture		120.2	
	Quebec	4	Forest	Herbland	Hybrid Poplar	0-30	76.9	Bambrick <i>et al.</i> (2010)
					Crop monoculture		80.1	
Quebec	8	Forest	Herbland	Mixed tree species	0-30	77.1		
				Crop monoculture		43.5		
Ontario	21	Forest	Herbland	Hybrid Poplar	0-20	56.9		
				Norway spruce		51.0		
				Crop monoculture		51.0		
Quebec	9	Forest	Herbland	Hybrid Poplar	0-5	21.9	Winans <i>et al.</i> (2014)	
				Crop monoculture		19.4		

**Appendix 2-1. Cont'd**

Agroforestry system	Location	Age (year)	Cover type	Species type	Soil depth (cm)	Soil C (Mg ha <sup>-1</sup> )	References
Alley cropping	Quebec	9	Forest	Hybrid Poplar	0-30	85.7	Winans <i>et al.</i> (2014)
			Herbland	Crop monoculture		113.2	
Shelterbelt	Alberta	30	Forest	Spruce based	0-10	37.9	Baah-Acheamfour <i>et al.</i> (2014)
			Herbland	Cropland		31.9	
	Alberta	30	Forest	Spruce based	0-30	88.1	Baah-Acheamfour <i>et al.</i> (2015)
			Herbland	Cropland		74.3	
Silvopasture	Ontario	13	Forest	Hybrid Poplar forest	0-5	0.3	Gordon and Thevathasan (2005)
			Herbland	Norway spruce forest Ryegrass		0.3 0.1	
Silvopasture	Alberta	80	Forest	Grazed aspen forest	0-10	54.8	Baah-Acheamfour <i>et al.</i> (2014)
			Herbland	Perennial grassland		46.2	
		80	Forest	Grazed aspen forest	0-30	109.1	Baah-Acheamfour <i>et al.</i> (2015)
			Herbland	Perennial grassland		91.9	
Riparian buffer	Quebec	9	Forest	Hybrid Poplar forest	0-20	39.8	Fortier <i>et al.</i> (2013)
			Herbland	Woodland buffer Herbaceous buffer		56.5 47.6	
Natural hedgerow	Alberta	80	Forest	Mixed tree forest	0-10	41.9	Baah-Acheamfour <i>et al.</i> (2014)
			Herbland	Cropland		35.3	
			Forest	Mixed tree forest	0-30	96.6	Baah-Acheamfour <i>et al.</i> (2015)
			Herbland	Cropland		81.5	

**Appendix 5-1.** Variability of soil temperature and moisture content across study sites in central Alberta, Canada. Values represent the mean from mid-May through September, during each of 2013 and 2014.

Agroforestry systems	County	Longitude, Latitude	Location	Soil Temp	Soil Moisture
				(°C)	(cm <sup>3</sup> /cm <sup>3</sup> )
Silvopasture	Athabasca	112°77'75.5"W, 54°57'10.4"N	North	12.25	0.24
Hedgerow	Athabasca	112°83'20.9"W, 54°60'92.3"N	North	12.27	0.32
Silvopasture	Smoky lake	112°30'83.5"W, 54°11'41.0"N	North	11.28	0.24
Shelterbelt	Smoky lake	112°73'49.3"W, 54°20'81.9"N	North	11.02	0.25
Hedgerow	Smoky lake	112°74'72.6"W, 54°21'08.1"N	North	10.24	0.25
Silvopasture	Smoky lake	112°73'57.3"W, 54°23'83.9"N	North	15.27	0.22
Shelterbelt	Thorhild	112°99'38.8"W, 54°05'42.5"N	North	15.79	0.32
Shelterbelt	Thorhild	112°92'67.0"W, 54°09'10.9"N	North	13.96	0.2
Hedgerow	Thorhild	113°01'57.3"W, 53°97'92.1"N	North	15.79	0.33
Shelterbelt	Thorhild	113°05'97.3"W, 54°09'30.5"N	North	8.18	0.31
Silvopasture	Beaver	112°93'77.5"W, 53°50'91.1"N	Central	12.09	0.27
Silvopasture	Beaver	111°52'61.4"W, 53°00'19.4"N	Central	13.77	0.19
Silvopasture	Parkland	114°42'04.1"W, 53°43'66.7"N	Central	12.01	0.27
Silvopasture	Parkland	114°09'61.0"W, 53°63'02.2"N	Central	13.78	0.25
Hedgerow	Sturgeon	113°63'45.2"W, 53°70'20.6"N	Central	16.31	0.21
Shelterbelt	Sturgeon	113°63'62.2"W, 53°70'95.0"N	Central	12.67	0.27
Shelterbelt	Sturgeon	113°65'83.5"W, 53°89'11.0"N	Central	12.81	0.27
Hedgerow	Sturgeon	113°65'16.5"W, 53°90'09.6"N	Central	13.81	0.27
Shelterbelt	Sturgeon	113°62'41.0"W, 53°92'02.8"N	Central	12.69	0.28
Hedgerow	Strathcona	113°17'27.0"W, 53°60'10.8"N	Central	13.45	0.28
Shelterbelt	Strathcona	112°97'25.4"W, 53°73'83.1"N	Central	10.76	0.32
Silvopasture	Strathcona	112°92'62.7"W, 53°70'56.4"N	Central	13.36	0.22
Hedgerow	Camrose	112°98'37.2"W, 52°75'56.0"N	South	11.62	0.21
Shelterbelt	Camrose	112°93'15.2"W, 52°78'57.8"N	South	15.47	0.29
Silvopasture	Camrose	112°95'16.1"W, 52°84'19.3"N	South	14.33	0.27
Hedgerow	Camrose	112°86'96.4"W, 52°89'44.5"N	South	15.13	0.31
Silvopasture	Camrose	112°92'77.7"W, 53°22'50.0"N	South	14.32	0.32
Hedgerow	Camrose	112°92'29.9"W, 53°22'54.6"N	South	16.06	0.32
Shelterbelt	Camrose	112°81'12.9"W, 52°98'28.0"N	South	16.24	0.26
Hedgerow	Camrose	112°83'49.5"W, 52°95'58.9"N	South	16.86	0.3
Hedgerow	Flagstaff	111°91'80.3"W, 52°97'45.4"N	South	13.66	0.64
Silvopasture	Flagstaff	112°14'48.5"W, 52°75'59.3"N	South	15.13	0.31
Hedgerow	Lacombe	113°65'82.9"W, 52°40'76.8"N	South	14.15	0.27
Shelterbelt	Lacombe	113°65'39.4"W, 52°40'46.3"N	South	14.87	0.29
Silvopasture	Lacombe	113°66'85.2"W, 52°40'66.1"N	South	12.75	0.32
Shelterbelt	Lacombe	113°57'88.9"W, 52°33'92.4"N	South	18.31	0.42



**Appendix 5-2.** Daily precipitation (mm) and mean air temperature ( $^{\circ}$  C) from May to September in 2013 and 2014 in central Alberta, Canada. Precipitation during the 2013 and 2014 field seasons (May 1 through Sept 30) is shown in parentheses. Daily precipitation and mean air temperature data were obtained from 15 first-order weather stations located in the study area.