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Carbon dynamics associated with different land uses in north central Alberta

by

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Abstract

Land use and land use change strongly influence the carbon (C) dynamics within ecosystems. This study quantified four aspects of land use and land use change effects: 1) ecosystem C stocks and distribution; 2) soil respiration; 3) soil C mineralization; and 4) net ecosystem productivity. Land use systems studied include agriculture (AG), 2-yr- and 9-yr-old hybrid poplar plantations (2HP and 9HP, respectively), grassland (GRA), and native forest stand (NAT). Ecosystem C stock in NAT (223 Mg C ha^{-1}) was similar to 9HP (174 Mg C ha^{-1}) and both were significantly greater than AG (122 Mg C ha^{-1}), GRA (121 Mg C ha^{-1}), and 2HP (110 Mg C ha^{-1}). Cumulative soil C loss via soil respiration averaged over two growing seasons was in the order of: NAT (7.81±0.40 Mg C ha^{-1}) > 9HP (5.51±0.31 Mg C ha^{-1}) > GRA (5.23±0.30 Mg C ha^{-1}) > AG (5.02±0.24 Mg C ha^{-1}) > 2HP (4.28±0.20 Mg C ha^{-1}). Depending on land use, seasonal heterotrophic and autotrophic respiration had respective contributions to soil respiration of up to 35 and 83%. Soil C mineralization of bulk soil across the land uses ranged between 2 to 5% of initial total organic C (C_i), with mineralization rates ranging from 0.06 to 0.12 μg C mg^{-1} C_i d^{-1} and mean residence times ranging from 30 to 51 yrs. Across particle size fractions, soil C mineralization was in the order of: AG > HPs > GRA > NAT of which the coarse fractions, representing labile C, were the main source of mineralized C (79%). Mineralization increased when NAT was converted to AG; and decreased when AG was converted to HP or GRA. Net ecosystem productivity across land uses, expressed in terms of C, ranged between -2 (AG) and 11 Mg C ha^{-1} yr^{-1} (older HP). Conversion from AG to GRA increased net ecosystem productivity three-fold. When AG was converted to HP, the plantation was a C source in the first four years and became a C sink by year five. Results obtained from this study are relevant to modeling efforts designed at determining the impact of future climate change on a variety of land uses.

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List of Abbreviations and Symbols

2HP	2-yr-old hybrid poplar plantation
9HP	9-yr-old hybrid poplar plantation
AG	Agricultural system
C	Carbon
CO_2	Carbon dioxide
C:N	Carbon to nitrogen ratio
GRA	Grassland
Mg	Megagram (10^6 g)
N	Nitrogen
NAT	Native aspen stand
NEP	Net ecosystem productivity
NPP	Net primary productivity
OHP	Older hybrid poplar plantation
ppm	Parts per million
Pg	Petagram (10^{15} g)
SOC	Soil organic carbon
SOM	Soil organic matter
YHP	Young hybrid poplar plantation
ΔB	Change in plant biomass of trees
D_a	Molecular diffusivity of CO_2 in air, $m^2\ s^{-1}$
D_{a0}	CO_2 diffusivity in air at 293.15 K and 101.3 kPa, given as 14.7 $mm^2\ s^{-1}$
D_i	Diffusivity of CO_2 in the soil, $m^2\ s^{-1}$, at depth i
F_i	Soil CO_2 flux, μmol $m^{-2}\ s^{-1}$, at depth i
P	Air pressure, kPa
Q_{10}	Relative increase in R_s for every increase of 10°C in T_s
R_a	Rate of CO_2 efflux originating from autotrophic respiration, μmol $m^{-2}\ s^{-1}$
R_h	Rate of CO_2 efflux originating from heterotrophic respiration, μmol $m^{-2}\ s^{-1}$
$R_{NAT-LFH}$	Rate of CO_2 efflux under the LFH (litter-fibric-humus) layer, μmol $m^{-2}\ s^{-1}$
R_s	Rate of total CO_2 efflux at the soil surface, μmol $m^{-2}\ s^{-1}$

R_{sub}	Rate of CO_2 efflux at 11-cm depth, μmol m^{-2} s^{-1}
T_s	Soil temperature, °C
T_a	Air temperature, K
$\partial C/\partial z$	Vertical soil CO_2 concentration gradient, μmol m^{-4}
$\delta^{13}C$	Natural abundance of ^{13}C, ‰
ε	Soil air-filled porosity, m^3 m^{-3}
θ_v	Soil volumetric water content, m^3 m^{-3}
ξ	Gas tortuosity factor
ρ_b	Soil bulk density, kg m^{-3}
ρ_m	Soil particle density, kg m^{-3}
φ	Total soil porosity, m^3 air m^{-3} soil

Chapter 1. Introduction

The biogeochemical carbon (C) cycle include the combined processes of photosynthesis and respiration, by which C cycles between its major reservoirs – the atmosphere, oceans, biota, and soil organic matter (IPCC 2001). The atmosphere contains around 807 Pg C (Bhatti et al. 2006) while the terrestrial biota contains somewhat less – around 400-600 Pg C (IPCC 2001). Carbon in soils amount to about 1500-2000 Pg C (Janzen 2004) and oceans, containing the largest reserves of C, hold about 39,000 Pg C, most of which are in deep ocean layers and are mostly not in active circulation.

Anthropogenic activities, specifically burning of fossil fuels and land use change, have increased atmospheric carbon dioxide (CO_2) concentrations. Ice cores show that during the past 1,000 years until about the year 1800, atmospheric CO_2 was fairly stable at levels between 270 and 280 ppm (Neftel et al. 1988). The current concentration of atmospheric CO_2 (385 ppm, Keeling et al. 2009) is increasing at an average of 1.65 ppm yr^{-1} (0.43 % yr^{-1}), and could rise to about 700 ppm by 2100 – more than doubling the pre-industrial levels – and could continue to rise for centuries to come (IPCC 1994). Because CO_2 traps heat, its increasing concentrations have been linked to global warming and climate change (IPCC 1996).

Ecosystem Carbon Dynamics

Carbon, as CO_2, enters the biota through the process of photosynthesis at an approximated rate of 120 Pg C yr^{-1}. About 50% (~ 60 Pg C yr^{-1}) is returned back to the atmosphere by plant respiration (Bhatti et al. 2006, Janzen 2004). Assimilated C then enters the soil system through plant litter or root exudates – acted upon by the process of decomposition by microbes thereby releasing C back to the atmosphere (heterotrophic respiration). Labile C may be present in the soil pool for short periods (days to months) whereas in other more resistant pools, such as humified residues or charcoal, the C may persist for longer periods of time, centuries or millennia (Gaudinski et al. 2000, Sauerbeck 2001). Thus, C entering the soil system may rapidly return to the atmosphere or may be distributed among soil C fractions and protected in inert forms for thousands of years. Estimated global CO_2 flux from soils range from 64 to 72 Pg C yr^{-1}, which accounts for 20-38% of annual input to the atmosphere from terrestrial and marine sources (Raich and Schlesinger 1992).

Land use and land use change influence ecosystem C stock and its dynamics. Under agriculture, for example, crop is removed from the system while residual biomass (stubble and roots) are mechanically mixed and incorporated in the surface layer of the soil each year during cultivation. Cultivation increases the active C pool which then leads to greater decomposition and soil C loss from the bulk soil. Under less frequently disturbed systems (plantations, grasslands, native forests), aboveground inputs (litterfall) are left on the ground to be decomposed and transferred into the mineral soil while belowground inputs (roots) enter the mineral soil directly. Conversion of soils from forest to agriculture therefore causes the decrease in C storage while, conversion of agricultural land to woody vegetation leads to the increase in C storage over time. On the Canadian prairies, more information on ecosystem C stocks and how its quality and distribution changes with land use change is needed. Estimating ecosystem C stocks and determining its distribution is critical in improving decision making processes in relation to managing terrestrial C sinks. Because of this, I conducted a mensurative experiment (Objective 1) to characterize C storage and distribution in ecosystem components such as above- and belowground biomass, bulk soil and its particle size fractions, and to understand the influence of land use change to the dynamics of soil C storage.

Soil Carbon Dynamics

Soil respiration

Soil respiration (R_s) is the primary pathway through which C fixed through photosynthesis returns to the atmosphere (Raich and Schlesinger 1992, Raich et al. 2002, Schlesinger and Andrews 2000). Soil respiration can account for 40-90% of total ecosystem respiration (Goulden et al. 1996, Janssens et al. 2002, Longdoz et al. 2000, Valentini et al. 2000) and is therefore one of the major components to consider in understanding ecosystem-atmosphere C exchange, particularly when determining the potential role of land use change to the function of the soil either as a source or a sink of atmospheric C.

The production of CO_2 in the soil is primarily due to autotrophic respiration (R_a) by roots and heterotrophic respiration (R_h) by microbes decomposing detritus, root exudates, and organic matter (Raich and Schlesinger 1992, Boone et al. 1998, Hanson et al. 2000). The rate of R_h ultimately controls soil C storage while R_a reflects plant activity and the allocation of organic compounds to root biomass from aboveground (Bhupinderpal-Singh et al. 2003, Binkley et al. 2006). Partitioning R_s is therefore important in understanding C cycling because R_h and R_a may respond differently to environmental controls, implying different behaviors at different time scales and under different plant communities. Recent reviews by Hanson et al. (2000), Bond-Lamberty et al. (2004) and Subke et al. (2006) have indicated that R_a generally accounts for 50% of R_s across a diversity of ecosystems and time scales. Hanson et al. (2000) reported an R_a contribution of 48 and 37% in forest and non-forest ecosystems, respectively, based on results from 50 publications, with the range between 10 and 90% across different ecosystem types. In addition, R_a has large seasonality, usually low during the dormant season and high during the active growing season (Hanson et al. 2000).

The dynamics of the two components of soil respiration, R_a and R_h, is controlled by several abiotic and biotic factors including: soil temperature (Boone et al. 1998, Rustad et al. 2001), soil water content (Davidson et al. 2000, Liu et al. 2002), soil drainage (Freeman et al. 1993, Luken and Billings 1985, Moore and Knowles 1989), soil fertility (Butnor et al. 2003), availability of C substrates for microorganisms (Seto and Yanagiya 1983), plant root densities and activities (Maier and Kress 2000), plant photosynthetic activity (Bhupinderpal-Singh et al. 2003, Högberg et al. 2001, Kuzyakov and Cheng 2001), and soil organism population size (Rai and Srivastava 1981). All of these variables are influenced by land use or land cover type (Raich and Tufekcioglu 2000).

A study on R_a and R_h across hybrid poplar plantations of varying ages (4-, 6-, 8-, and 13-year-old) in northern Alberta by Saurette et al. (2008) found that R_a and R_h had seasonal variations and that the contribution of R_h (~63%) to R_s across all plantation ages dominated over the growing season. Further information however is needed to determine how R_s, R_h, and R_a in these plantations compare with other land uses and identify biophysical factors that drive their variations. In this study, I designed a modified root exclusion experiment (Objective 2) using both discrete and continuous measurements to determine the contributions of R_h and R_a to R_s, the temporal and spatial patterns of R_h and R_a, and identified potential site factors affecting their variations.

Soil C stabilization

Soil C loss via R_h can further be elucidated by identifying the source of C within the bulk soil. Soil C is composed of a variety of fractions or pools with different decomposition rates, turnover times, and degrees of protection. Carbon associated with coarse size fractions is sometimes referred to as the *labile* C pool that has slower turnover time than that of recently shed litter but faster than that of C associated with finer particle size fractions. It responds rapidly to changes in land use, vegetation type, climate, soil type, faunal activity, and could provide early indication of effects of soil management (Camberdella and Elliot 1992, Gregorich et al. 1994). Carbon associated with finer particle size fractions, referred to as the *passive* C pool in this thesis, on the other hand, is held together by microbially processed products, polyvalent cations and other strong organic polymers (Cheshire and Mundie 1981, Christensen 2001) and is characterized as stabilized or recalcitrant. Stabilization results from C either being physically protected in microaggregates or chemically associated with silt and clay (Six et al. 2002). Recalcitrance results from the inherent chemical property of the plant material *per se* (referred to as litter quality) or from condensation and complexation of decomposition residues that cause them to be more resistant to further decomposition. The complexation process decreases microbial access to C by pore-size exclusion that leads to reduced microbial activities and mineralization rates (Six et al. 2002, Bronick and Lal 2005, Jastrow and Miller 1997) and therefore promotes C accumulation in soils. Protection of C from further microbial decomposition in these fractions is essential to significantly lengthen the residence time of C in soils (Jastrow et al. 2007).

There have been many studies investigating C stabilization patterns in different soils (e.g. Crow et al. 2006, Paré et al. 2006). However, there is still much knowledge to be gained by investigating the impact of land use and land use change on the stabilization of C from not only the bulk soil but also from its respective particle size fractions. If soils are to be used as potential C sinks, then the quantity and quality of the accumulating C in these soil systems must be known. In this regard, I designed a particle size fractionation and incubation study (Objective 3) to investigate the impact of land use and land use change on C stabilization in soils. Stabilization of soil C was measured by determining the mineralization kinetics of C in bulk soil and its fractions. I further attempted to expand the current knowledge on soil C sensitivity to temperature by incubating the bulk soil and its various fractions under three different temperatures.

Land use and Land Use Change

Conversion of forest or natural systems to agroecosystems results in the depletion the soil C stock by 20-50% (Davidson and Ackerman 1993). Depletion of soil C stock is influenced by various factors including 1) decrease in above-and belowground biomass returned to the soil; 2) change in soil microclimate (soil moisture and temperature) which affects decomposition rates of organic matter; 3) decomposability of crop residues due to litter quality differences; 4) tillage-induced disturbances that lessens the physical and biochemical protection of C, exposes protected organic matter to biological activity, and increases its decomposability (Lal 2005, Blanco-Canqui and Lal 2004).

Land that has been depleted of soil C due to land use change however may potentially be a sink for C if we can employ land use conversion (such as afforestation, reforestation, and restoration of cultivated, abandoned and marginal soils) and management practices to increase soil C storage (Ross et al. 2002, Silver et al. 2000, Trujilo et al. 1998). In this respect, the establishment of large-scale short-rotation plantations has been advocated as an effective method for sequestering CO_2 and mitigating increased atmospheric CO_2 levels (House et al. 2002), through increasing long-term C storage in woody biomass (Schimel et al. 2001) and in the soil (Garten Jr. 2002), and by providing bioenergy (Tuskan and Walsh 2001). Converting agricultural land to short-rotation woody crops has been reported to increase soil C content by 10 to 25 Mg ha^{-1} in 10 to 15 years (Grigal and Berguson 1998). Findings with respect to ecosystem C accumulation following conversion to tree plantations seem promising. However, the net C benefits of these plantations are still unclear, particularly as it is expected that those plantations would act as a C source during the initial years following establishment due to cultivation (for plantation establishment and weed control) that accelerate short-term decomposition of soil organic matter (Grigal and Berguson 1998). If short-rotation plantations are to be used as potential C sinks, the quantity of C accumulation as well as the turning point when these plantations become C sinks must be determined. In light of this knowledge gap, I designed a biometric-based experiment to determine the net ecosystem productivity of hybrid poplar plantations in relation to other land uses (Objective 4). Only one other study (Saurette et al. 2008), to my knowledge, examined the impact of hybrid poplar plantation on net ecosystem productivity after establishment. I expanded this knowledge by determining when hybrid poplar plantations act as C sources, when they become C neutral, and when they start becoming C sinks, both at the soil and ecosystem levels.

Study Objectives

The main objective of this study was to compare ecosystem C stocks and distribution, soil respiration, soil C mineralization, and net ecosystem production across different land uses (agriculture, plantation, grassland, and native forest) and determine how these aspects of the ecosystem C cycle changed with land use change (native forest to agriculture, agriculture to plantation, agriculture to grassland).

The specific objectives and hypotheses of this study were:

1. To compare ecosystem C stocks and distribution across land uses and to understand the influence of land use change on ecosystem C storage (Chapter 2).

   Hypotheses

   a. Ecosystem C stocks will be different across land uses;
   b. Soil C will be greater than biomass C for all land uses;
   c. Ecosystem C will decrease when NAT is converted to AG; and will increase when AG is converted to HP and when AG is converted to GRA.
2. To determine seasonal and diurnal changes in soil respiration rates under different land uses; partition soil respiration into its heterotrophic and autotrophic components; and examine the factors affecting variations in soil respiration and its components (Chapter 3).

   Hypotheses

   a. Soil respiration rates will be affected by soil temperature and soil moisture;

    b. Cumulative soil C loss due to soil respiration will increase with increasing ecosystem C input;

3. To determine mineralization rates and mean residence time of C in soils under different land uses; to determine the contribution of different particle size fractions to total mineralized C in the bulk soil; and to examine the effect of changes in temperature on the mineralization and turnover rates of C in bulk soils and particle size fractions from different land uses (Chapter 4).

    Hypotheses

    a. C mineralization rates and turnover time from bulk soil will be different across land uses;
    b. C mineralization and turnover time will decrease with increasing particle size;
    c. C mineralization and turnover time from bulk and particle size fractions will increase with increasing temperature.

4. To examine the land use effects on ecosystem C productivity and characterize the C source-sink dynamics in different land use systems (Chapter 5).

    Hypotheses

    a. Land conversion from NAT to AG will decrease ecosystem C productivity;
    b. Land conversion from AG to GRA or HP will increase ecosystem C productivity;
    c. HP will be a C source in the initial years after plantation establishment and become a C sink a few years thereafter.

A combination of field and laboratory experiments was conducted to test the hypotheses listed above. The first study was a mensurative experiment to determine ecosystem C stock and distribution in different land uses (Objective 1). The second experiment compared soil respiration patterns across the land uses, determined the contribution of heterotrophic and autotrophic respirations to soil respiration, and correlated soil respiration and its components to some biophysical factors (Objective 2). The third experiment was conducted in the laboratory to determine mineralization patterns and temperature sensitivity of soil C (bulk and particle size fractions) obtained from the different land uses (Objective 3). The fourth experiment synthesized all the data from the previous chapters in order to determine net ecosystem productivity and characterize C source-sink dynamics in different land uses (Objective 4).

This thesis is organized into six chapters. Following this introductory chapter, Chapter 2 presents the findings of a study comparing ecosystem C storage and distribution of various land uses (agricultural, young and older hybrid poplar plantations, grassland, and native forest stand) in north central Alberta. Chapter 3 reports results of seasonal and diurnal variations in R_s and the proportions contributed by its heterotrophic and autotrophic components in each land use, including driving factors that were found to be affecting their variations. Chapter 4 presents the results of a year-long incubation study determining the mineralization potential and temperature sensitivity of soil C in bulk soil and its particle size fractions under various land uses. Chapter 5 describes the effect of land use and land use change on ecosystem C production potential and characterizes the C source-sink relationship in different land use systems. A special portion of Chapter 5 was dedicated to examining the C source-sink dynamics along the first 11 years of hybrid poplar growth. Finally, Chapter 6 presents a summary of the previous four chapters and provides suggestions for future research.

The results obtained from this thesis will improve our understanding of C dynamics in agricultural, tree plantations, grasslands, and native forest systems, and how this process changes with land use change. The knowledge gained will provide resource managers valuable information that can be used in the determination, recommendation, and implementation of C management strategies and decision making processes to address issues pertinent to reducing C emissions by sequestering C in biomass and soils. Little information is currently available regarding the potential for C sequestration of some land uses in the boreal region on the Canadian prairies; hence this study will be invaluable as it makes a significant contribution to Canada's efforts in mitigating climate change.

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Chapter 2. Ecosystem carbon stocks and distribution under different land uses[*]

Introduction

Land use change is often associated with changes in land cover and carbon (C) stocks (Bolin and Sukumar 2000). Each soil has an equilibrium C storage potential that is determined by the nature of vegetation, climatic conditions, and physicochemical properties of the soil (Gupta and Rao 1994, Six et al. 2002). The equilibrium soil organic C (SOC) storage results from a balance between inputs and outputs of C (Fearnside and Barbosa 1998). This equilibrium can be perturbed by land use change until a new equilibrium is reached in the new ecosystem. During this process, the soil may act either as a C source or sink, depending on the interaction between land use, cropping systems, and management practices (Lal 2003, Singh and Lal 2005, West and Marland 2002). The conversion of forest lands to an agricultural land use, as an example, invariably results in the release of large quantities of CO_2 into the atmosphere and rapid decline of SOC stocks (Salinger 2007). Several reviews (Davidson and Ackerman 1993, Gregorich et al. 2005, Guo and Gifford 2002, Mann 1986, Post and Kwon 2000) estimated that loss of soil C after cultivation of native soil ranges between 20 and 50%. Davidson and Ackerman (1993) suggested that nearly all C lost from soil occurs within 20 years, and that most occurs within 5 years after initial cultivation. Much of this loss is due to (i) lower inputs of organic matter, (ii) change in soil microclimate which increase decomposition rates of organic matter, (iii) increase in decomposability of crop residues due to changed litter quality (for example, lowered C:N ratio and lignin content), and (iv) tillage-induced

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disturbances that decrease soil aggregation and physical protection of soil organic matter (Post and Kwon 2000, Lal 2005).

Changes in land use have had considerable impact on the global C cycle. It has been estimated that approximately 123 Pg of C were released to the atmosphere between 1850 and 1990 due to land use change (Houghton 1999). Currently, agricultural practices and land use change contribute about 18-20% of the total anthropogenic emissions of CO_2 each year (Baumert et al. 2004, Dumanski 2004). This accounts for approximately 60% of total emissions from the underdeveloped countries, 33% from developing countries, and up to 10% from developed countries (Baumert et al. 2004). Land that has been depleted of soil C due to land use change however may potentially be a sink for C if land use conversion (such as afforestation, reforestation, and restoration of cultivated, abandoned and marginal soils) and management practices were employed to increase SOC storage (Ross et al. 2002, Silver et al. 2000, Trujilo et al. 1998).

Establishment of large-scale short-rotation woody crop plantations has been advocated as an effective method for sequestering CO_2 and mitigating increased atmospheric CO_2 levels (Silver et al. 2000, House et al. 2002), through increasing long-term C storage in woody biomass (Schimel et al. 2000, Sedjo 1989) and in the soil (Garten 2002, Hansen 1993, Tolbert et al. 2000), and by providing an alternative source of biomass for bioenergy (Tolbert et al. 2000, Tuskan and Walsh 2001). Converting agricultural land to short-rotation woody crop plantations has been reported to increase soil C content by 10 to 25 Mg ha^{-1} in 10 to 15 years (Hansen 1993, Grigal and Berguson 1998). However, previous studies on C changes under short-rotation woody crops have been focused on C accumulation in aboveground biomass and/or bulk soil and there is still much to be learned on the impact of short-rotation woody crop plantations on ecosystem C dynamics, particularly in terms of how belowground biomass C and soil C storage changes with stand development. The impact of intense cultivation in the first a few years of plantation establishment on soil C dynamics is poorly understood.

Physical, chemical, and biological methods are often employed to separate soil organic matter (SOM) into fractions with different turnover rates. Of those methods, particle size fractionation of SOM is a commonly used physical method to separate SOM into various particle size fractions. Soil organic matter in larger particle size fractions belongs to the labile pool while the SOM in smaller particle size fractions belongs to more recalcitrant pools (Camberdella and Elliot 1992, Christensen 1992, Christensen 2001). Soil C dynamics can further be studied using information on the natural abundance of ^{13}C ($\delta^{13}C$) and ^{15}N ($\delta^{15}N$) in litter inputs, bulk soil, and soil fractions (pools). The $\delta^{13}C$ values of plant leaves reflect the integration of fractionation events during photosynthesis while $\delta^{13}C$ values of SOM reflect contributions from plant litter and the consequences of subsequent processes within the soil (Ehleringer et al. 2000). The $\delta^{13}C$ and $\delta^{15}N$ in ecosystems generally increase with increasing soil depth or decreasing organic C or N concentrations, as represented by decreasing particle size fractions, which are thought to correspond to different stages of SOM decomposition (Ehleringer et al. 2000, Natelhoffer and Fry 1988). Possible mechanisms for explaining heavy isotope enrichment include: (i) microbial discrimination against ^{13}C and ^{15}N during litter decomposition; (ii) differential preservation of organic matter; (iii) influence of atmospheric change (SOM at deeper soil depths is older, the C in the deeper soil layers should have originated at a time when atmospheric isotopic values were more positive); and (iv) soil mixing or illuviation of ^{13}C and ^{15}N enriched organic matter into lower soil layers (Ehleringer et al. 2000, Natelhoffer and Fry 1988).

The objectives of the study were: (i) to characterize trends in C storage and distribution in ecosystem components such as above- and belowground biomass and soil particle size fractions; and (ii) to understand the influence of land use changes to the dynamics of soil C storage.

Materials and Methods

Study site

This study was conducted in Linaria (54°12'10" N, 114°8'46" W) located approximately 25 km west of the Town of Westlock, in north central Alberta, Canada, from May to October 2006 (Figure 2-1). The site is 631 m.a.s.l, with a mean annual (1994-2006) temperature of 3 °C and mean annual precipitation of 1483 mm, of which 76% falls as snow (Environment Canada 2007). The site has a continental climate, with cold winters and warm summers that provide an approximate growing season of 180-185 days (Alberta Agriculture, Food, and Rural Development 2003). The soil is poorly drained and is a Dark Gray Luvisol on medium textured till on an undulating landscape with low relief (2% slope, Agriculture and Agri-Food Canada 2005).

A comparative mensurative experimental design (Hulbert 1984) was used to compare four different land uses: agricultural (AG), grassland (GRA), 2-yr-old and 9-yr-old hybrid poplar (*Populus deltoides* x *Populus x petrowskyana* cv. Walker) plantation (2HP and 9HP, respectively), and native aspen (*Populus tremuloides* Michx.) forest land use. All sites were adjacent to each other within an area of about 4 km^2 and the soils were developed on the same parent material. A basic assumption is that soils on all five land uses were similar prior to imposition of land use differences. Space-for-time substitutions (chronosequences) were used to determine ecosystem C changes resulting from land use change i.e., from NAT to AG, from AG to GRA, and from AG to 2HP to 9HP. Soil C levels in NAT may reflect the balance between C inputs and losses while that in the 2HP, 9HP, GRA, and AG indicate the changes brought about by land use changes. Three 20 x 20 m plots representing subsamples of the land uses were established for each land use type.

The agricultural field (55 ha) was cleared in the 1930s (Herb Smerychynski, 2006, personal communications). The landowners practiced conventional tillage and added fertilizers for small grains and canola production following a four-year rotation of barley (*Hordeum vulgare* L.) – barley – wheat (*Triticum aestivum* L.) – canola (*Brassica napus* L.). During the time of this study, barely was being planted in rows 15 cm apart. Fertilization was usually at 124 kg N ha^{-1} yr^{-1}, 34 kg P ha^{-1} yr^{-1}, 25 kg K ha^{-1} yr^{-1}, 11 kg S ha^{-1} yr^{-1}. The GRA (12 ha) was converted from conventional agricultural land use in 1997. The site was seeded with a mixture of *Festuca arundinacea* Schreb. (tall fescue), *Dactylis glomerata* L. (orchardgrass), and *Trifolium pratense* L. (red clover). The site was allowed to lie idle and the grasses were merely maintained by mowing three times a year. Plant litter left behind after mowing was allowed to remain on the site.

The 2HP (20 ha) was established in the spring of 2004 by the Canadian Forest Service, as part of the Forest 2020 Plantation Demonstration Assessment Initiative Program of the federal government while the 9HP (< 1 ha) was established by Millar Western Forest Products Ltd. in the spring of 1998 as part of the company's hybrid poplar (Walker) test program. Site preparation before hybrid poplar plantation establishment involved deep (25-30 cm) and shallow (10 cm) cultivation. The former was done to break up heavy soil and compact layers using a 4.3 m wide braking disk while the latter was done using a 150 hp tractor pulling a shallow field disk to further break up large soil clods. Three weeks later, a post-emergent herbicide (glyphosate, at 1.62 kg a.i. ha^{-1}) was applied to control newly emerged weeds. Thereafter, container-grown stocks (plugs) of Walker (origin: Indian Head, SK) were planted in a 2.5 x 2.5 m spacing resulting in a final plant density of 1600 plants ha^{-1}. Two weed control cultivations were conducted with a 2.4 m wide cultivator in the 2HP in the first two years after planting. In the 9HP, mowing two to three times a year was used to control weed growth between tree rows. The NAT (2.5 ha), which is more than 80 years old, has no history of cultivation, logging, or grazing by domestic animals.

Biomass carbon

Diameter at breast height (DBH) and total height of all trees taller than 1.3 m in each plot were measured in 2HP, 9HP, and NAT. Stem, branch, and coarse root biomass was calculated using published allometric estimations for hybrid poplars (Ballard et al. 2000, Dickmann et al. 2001, Fang et al. 2007, Wullschleger et al. 2005) and trembling aspen (Peterson and Peterson 1992). Coarse woody debris present in NAT was measured using the line-intercept method (Halliwell and Apps 1997). Except in the 2HP where the weed control practice was effective and no weed biomass was quantified, aboveground biomass (crop, stubs, grass, and understory) at the AG, GRA, 9HP, and NAT sites was measured in three randomly located clip plots (1 x 1 m in NAT and 0.15 x 0.15 m in AG, GRA, and 9HP) per treatment plot. All vegetation was removed from each clip plot, oven-dried at 65 °C until constant weight and weighed. Measurement values were then scaled up from g m^{-2} to Mg ha^{-1} for GRA and 9HP while also taking into consideration the 15 cm row spacing for canola in AG.

Litterfall for 2HP, 9HP, and NAT was measured using three 1 x 1 m litterscreens per plot. These were installed in August 2006 and litter samples were collected in November 2006 and dried. In addition, subsamples of litter were finely ground in a ball mill (Mixer Mill MM200, Thomas Scientific, Swedesboro NJ) and analyzed for ^{13}C and ^{15}N abundances using a VG Optima continuous-flow stable isotope ratio mass spectrometer (VG Instrument, England) linked to a Carlo Erba NA 1500 CN analyzer (Milan, Italy).

In order to determine fine root biomass, three soil cores (2.5 cm diameter x 30 cm deep) per plot were collected three times (June, August, and October 2006) during the growing season. Core samples were immediately stored at 2 °C until further processing. Fine roots (< 2 mm) were separated from the soil by soaking in water and gently washing the samples over two sets of sieves (2.0 and 0.5 mm). Sieve contents were dispersed in water to separate roots from finer soil particles and organic matter. Fine roots were hand-picked and oven-dried at 65 °C for 48 h and weighed. Biomass C pools of ground vegetation, litter and fine roots were calculated by multiplying the dry mass by a standard factor of 0.5 (Pregitzer and Euskirchen 2004).

Soil microbial biomass C and N (MBC and MBN) were measured three times (June, August, and October 2006) during the growing season using the chloroform fumigation-extraction method (Voroney et al. 1993). Briefly, 20 g of fresh soil was fumigated with alcohol-free chloroform for 24 hr in an evacuated desiccator. Fumigated and unfumigated samples were extracted with 80 ml of 0.5 M K_2SO_4 by shaking on a flatbed shaker for one hour. Extracts were filtered using Fisherbrand® Q2 filter paper and filtrate was frozen until analysis. Total soluble organic C and N in extracts were analyzed using a TOC-V Total Organic Carbon Analyzer (Shimadzu Corp., Kyoto, Japan). Extractability factors of 0.25 and 0.15 were used for the calculation of MBC and MBN, respectively (Voroney et al. 1993).

Soil physical and chemical properties

Forest floor samples were collected from NAT by obtaining samples from three randomly assigned sample points (0.15 x 0.15 m) per sampling plot. The forest floor averaged 7.5-10 cm thick in this stand. Mineral soils from all land use sites were sampled from the surface (0-20 cm) and subsurface (20-50 cm) layers. In each sampling plot, three soil cores were collected per soil horizon and combined to form a composite sample. All soil samples were transported back to the lab and analyzed for total organic C and total organic N using the TOC-V Total Organic Carbon Analyzer (Shimadzu Corp., Kyoto, Japan); for soil texture (excluding forest floor), pH, available N (NH_4^+ and NO_3^-), available P (using the Bray 1 dilute acid-fluoride extraction procedure), exchangeable K (using the ammonium saturation method), exchangeable cations (Mg and Ca), and CEC following Kalra and Maynard (1991). Additionally, bulk density (BD) was determined by obtaining soil samples from each of the sampling plots and from each soil horizon using a bulk density corer (172 cm^3). The soil sample was then returned to the lab, dried at 105 °C until constant weight, and weighed. Bulk density of the soil was then computed by dividing the dry weight by soil core volume.

Soil particle size fractionation

In each treatment plot, 20 soil samples (corer size: 4.7 cm diameter x 10 cm height) were randomly collected from the 0-10 cm layer and mixed to form a composite sample. Size fractions were separated without chemical pre-treatment using a modified method derived from various techniques described in Anderson (1981), Bird (2003), Christensen (1992), Christensen (2001), Preston et al. (2000), and Six et al. (2002). Ultrasonic dispersion in water, followed by wet-sieving, were used to separate the fine earth fraction into the following: 250-2000 μm (coarse-size fractions), 53-250 μm (medium-size fractions), 2-53 μm (silt-size fractions), and < 2 μm (clay-size fractions). The complete fractionation sequence is illustrated in Figure 2-2. Briefly, air-dried soil (100 g) that has passed through a 2 mm sieve was weighed into 500 ml containers to which 150 ml of distilled water was added, and the sample shaken on a flatbed shaker for one hour to break up soil structure. The soil slurry was ultrasonically dispersed with a Fisher Sonic Dismembrator model 300 (Fisher Scientific, Pittsburgh, PA) for two minutes at 60% of maximum power. Samples were then wet sieved through a 250 μm and 53 μm mesh. Material less than 53 μm but larger than 2 μm was further separated from the < 2 μm portion by sedimentation and decantation. The remaining solution was flocculated with 1 N KCl with subsequent removal of KCl by dialysis (Fisherbrand® Nominal MWCO 12000-14000) in distilled water until washing solution was free of salt to obtain the <2 μm soil fraction. Coarse-, medium-, and silt-size fractions were dried in the oven at 40 °C (Christensen 1992) for 72 hr while the clay-size fraction was freeze-dried (FreeZone with a Bulk Tray Dryer, Labconco Corp., Kansas City MO) for one week in order to minimize shrinkage and preserve soil porosity (Lawrence et al. 1979). Average recovery of samples was 98%. The fractions other than the clay fraction were finely ground in a ball mill (Mixer Mill MM200, Thomas Scientific, Swedesboro NJ). The samples were analyzed for total organic C, total organic N, ^{13}C and ^{15}N abundance using a VG Optima continuous-flow stable isotope ratio mass spectrometer (VG Instruments, England) linked to a Carlo Erba NA 1500 CN analyzer (Milan, Italy).

Statistical analyses

The influence of land use on above- and belowground C and N stocks and natural abundance of ^{13}C and ^{15}N in litter, bulk soil, and particle size fractions were tested with a one-way analysis of variance (ANOVA) using PROC GLM in SAS v. 9.1 (SAS Institute Inc. 2004). The assumption of normality was assessed using the univariate procedure in SAS with a Kolmogorov-Smirnov and Shapiro-Wilk tests for normality (data not shown). All data conformed to a normal distribution. Tukey's (HSD) Test and the *pdiff* option in PROC GLM were used to identify and separate significant main and interaction effects at significance level $\alpha = 0.05$.

The combination of land uses that were examined in this study could not have been replicated elsewhere. Because of the nature of pseudo-replication, it should be noted that the sampling plots established in each of the land uses did not represent independent units with the same treatment (land use) but rather should be considered as sub-samples within non-replicated land use treatments. Results obtained from this study can only be applied to the research plots studied, and that they cannot be used to make wider inferences to the greater population of possible plots and treatments.

Results

Soil properties

Soil BD increased with increasing soil depth in the two plantation stands and was similar among the land use treatments in the 0-20 cm layer (Table 2-1). Mean soil pH ranged from 4.86 to 5.35 in the surface and from 4.65 to 6.59 in the subsurface layer. Surface and subsurface soils under NAT were slightly more acidic compared to other land uses (Table 2-1). Soil texture of the sites ranged from clay loam to clay. Within the surface layer, soil textural compositions were similar among the land uses. Within the subsurface layer, NAT had the highest proportion of silt and the lowest proportion of clay compared to other land uses, while AG had relatively the lowest amount of silt and the highest amount of clay. The highest nitrate (NO_3^-) levels were found in the AG and 2HP sites in both sampled layers (Table 2-2). In the surface layer available P and K contents were greater or tended to be greater in the AG, 2HP and NAT sites than in the other land uses, otherwise land use treatments caused little effects on nutrient availabilities and CEC in the surface soil layer (Table 2-2).

Carbon and nitrogen stocks

Aboveground biomass C stock in NAT (49 Mg C ha^{-1}) was significantly greater than that in all other land uses, except in the 9HP which had 40 Mg C ha^{-1} (Table 2-3). Aboveground biomass C stock in the AG site was significantly lower than 9HP and statistically similar to the 2HP and GRA sites. It should be noted that the aboveground biomass (crop) measured in the AG site represented biomass C at the peak of the growing season (July). Of the 5 Mg C ha^{-1} present during the growing season, only 20% (1 Mg C ha^{-1}) was left behind after harvest in August (data not shown). The residual biomass (stubble and roots) was then incorporated into the surface soil layer through tillage in late fall and then again in early spring of the following year. In addition, leaf biomass C significantly increased in the following order: 2HP (0.4 Mg C ha^{-1}) < 9HP (4.2 Mg C ha^{-1}) < NAT (6.9 Mg C ha^{-1}).

Belowground, coarse root biomass C in NAT was significantly higher than that in 9HP and the 2HP (Table 2-3). Fine root biomass C was greatest in NAT and least in the 2HP. Furthermore, biomass C across the land uses was generally higher aboveground than belowground (root:shoot ratios range between 1.35 and 19.4, data not shown) except in GRA (root:shoot ratio 0.87, data not shown).

The NAT had a distinctive feature of a forest floor composed of snags (12.52 Mg C ha^{-1}), coarse woody debris (14.79 Mg C ha^{-1}), and the LFH layer (11.07 Mg C ha^{-1}). The SOC content in the mineral soil ranged from 38 to 78 Mg C ha^{-1} and decreased with depth except in NAT (Table 2-3). In the surface layer, the only land use effect on SOC content was between the 9HP and the NAT. In the subsurface layer, SOC content was not different under the different land uses. Total soil organic nitrogen (N) content ranged from 3 to 7 Mg N ha^{-1} and decreased with depth except in NAT where N contents were not different between soil layers. In the surface layer, total N content was lower in NAT than in the other land uses. Soil C:N ratios were within a narrow range (10-12) and were not different among all the land uses.

Total biomass C (above- plus belowground biomass C) was in the order of NAT ≥ 9HP ≥ AG = GRA = 2HP (Figure 2-3). Total SOC (surface plus subsurface layers) ranged from 107 to 137 Mg C ha^{-1} and were not affected by land uses (Table 2-3, Figure 2-3). The total ecosystem C stock (biomass C plus SOC) to 50 cm depth (Figure 2-3) in the NAT (223 Mg C ha^{-1}) was not different from that of the 9HP (174 Mg C ha^{-1}) and both land uses were significantly greater than in the AG (122 Mg C ha^{-1}), 2HP (110 Mg C ha^{-1}), and GRA (121 Mg C ha^{-1}).

Carbon and nitrogen distribution along soil particle size fractions

All soil samples had high amounts of silt- (46 to 57%, Table 2-4) and medium-size fractions (28 to 33%); and low amounts of coarse-size fractions (6 to 12%) and clay-size fractions (7 to 9%, data not shown). To overcome the potential problem of underestimating the quantity of clay-size fraction due to incomplete separation of clay from the other fractions, the combined silt- and clay-size fractions representing the fine fraction were reported (Table 2-4).

Most of SOC and total N were generally held within the fine fraction, followed by medium fraction, then coarse fraction for all land uses, except under NAT where SOC and N were similarly distributed across particle size fractions

(Table 2-4). Soil C and N associated with the fine fraction were highest under AG, 9HP and 2HP, intermediate in GRA, and significantly lower in the NAT.

These distribution patterns were similar when SOC and N were expressed as concentrations (Figure 2-4). SOC and total N concentrations in the bulk soil were similar across the different land uses (26 g C kg^{-1}, 2 g N kg^{-1}), except under NAT where SOC and N concentrations in the bulk soil were significantly lower (21 g C kg^{-1}, 1.5 g N kg^{-1}). The majority of SOC and total N were also held within the fine fraction for all land uses with the highest in AG, 9HP and 2HP (average of 16.4 g C kg^{-1}, 1.4 g N kg^{-1}), intermediate in GRA (15.0 g C kg^{-1}, 1.3 g N kg^{-1}), and lowest in NAT (11.2 g C kg^{-1}, 0.8 g N kg^{-1} Figure 2-4). The C:N ratio decreased with decreasing size of the particle fraction (Table 2-4) for all land uses: highest in the coarse-, intermediate on the medium-size fraction, and lowest in the fine-size fractions.

Values of $\delta^{13}C$ in litter, bulk soils, and particle size fractions across the different land uses were between -29.35 to -25.57‰ (Figure 2-5). Values of $\delta^{13}C$ in bulk soils ranged from -27.01 to -26.24‰. Within soil particle size fractions, progressive (with a few exceptions) ^{13}C enrichment with decreasing particle size was evident. As compared to the AG site, two years after hybrid poplar plantation establishment, the coarse- and the fine-size fractions had become 1.5 and 1.9%, respectively, more enriched in ^{13}C, respectively, while nine years after hybrid poplar plantation establishment, the same size fractions had become 3.4 and 1.0%, respectively, more enriched in ^{13}C; whereas, when the AG was converted to GRA the same particle size fractions had become 3.6 and 0.7%, respectively, more enriched in ^{13}C after nine years. For coarse-size fractions, on the other hand, ^{13}C became depleted (^{13}C values became lower) as a consequence of converting AG to other land uses. Eighty years after AG replaced NAT, the coarse-, medium-, and fine-size fractions have been depleted with ^{13}C by 3.8 and 3.3%, respectively.

The values for $\delta^{15}N$ were all positive, ranging from 0.37 to 4.88‰ in litter; from 4.86 to 8.49‰ in bulk soil; and from 3.72 to 8.46‰ in the soil particle size fractions. Values of $\delta^{15}N$ in litter were substantially lower compared to bulk soil and particle size fractions. Like $\delta^{13}C$, soil $\delta^{15}N$ values tended to increase with decreasing particle size.

Discussion

A basic assumption for this comparative mensurative experiment is that the soil properties of the five sites (AG, 2HP, 9HP, GRA, and NAT), all were adjacent to each other, were similar prior to the imposition of different land use systems. Conversion from NAT to AG land use did not significantly change soil physical and chemical properties except increased available NO_3^- (Table 2-2). The increase of total N content in the 0-20 cm soil in the non-native aspen sites (Table 2-3) may have indicated that application of inorganic fertilizers (and the resultant increase in biomass production and incorporation of residues into the surface soil layer through cultivation) had more than offset the removal of N by crop harvest. In GRA, N input would have also have come from N-fixing red clover (*Trifolium pratense* L.) dominantly present in the site. In AG, increasing exchangeable Ca and Mg with soil depth may either be associated with inherent amounts of Ca and Mg in the parent material and/or may have been brought about by weathering and leaching. Constant mixing of the soil by plowing and disking of the soil surface release more exchangeable Ca and Mg, which are leached to subsurface layers with precipitation and infiltration (Hussain et al. 1999). This might also explain the slight increases in soil pH in the subsurface layer in this land use type. In NAT, exchangeable Ca and Mg are significantly lesser than in AG soil suggesting that exchangeable Ca and Mg may be linked to nutrient uptake by plants and active cycling in the surface layer (Dijkstra et al. 2001, Dijkstra and Smits 2002).

Although SOC content (0-50 cm) was greater than biomass C for all land uses, significant differences in SOC among the land uses were not detected (Table 2-3). Above- and belowground biomass C stocks accounted for the observed differences between the land uses. The rate of increase of biomass C in the 9HP is 6.3 Mg C ha^{-1} yr^{-1} (with above- and belowground biomass C accumulation rates of 4.5 Mg C ha^{-1} yr^{-1} and 1.7 Mg C ha^{-1} yr^{-1}, respectively). Nilsson and

Schopfhauser (1995) studying C-sequestration potential of global afforestation programs suggested that aboveground C accumulation in plantations in Canada is about 1.2 Mg C ha^{-1} yr^{-1}. This value is four times lower than what was found in this study. The lower value calculated by Nilsson and Schopfhauser (1995) was based from aboveground C accumulation potential for suitable and available lands in Canada encompassing different regions, species, and management programs whereas; the higher value in this study represents only accumulation rates for hybrid poplar, specifically for Walker, under site-specific conditions.

Changing from native forest to agricultural land use is usually followed by a decrease in SOC stocks (Guo and Gifford 2002). Previous research by Houghton (1999) estimated a SOC loss (from 1 m depth) of 51 Mg C ha^{-1} when boreal forests were converted to agricultural land use. It has also been reported that 20 to 30% of native SOC stocks are lost in the first few years due to cultivation and continue to lose SOC at a slower rate until it reaches a new equilibrium after 30-50 years (Davidson and Ackerman 1993, Mann 1986, Anderson 1995, Murty et al. 2002). From this study, it emerged that 80 years of conventional agricultural use, of a native forest soil, had only resulted in a slight non-significant decrease of SOC in the top 50 cm of soil for sites in Linaria, AB (Table 2-3).

Differences in SOC were not statistically significant between the AG, 2HP, 9HP, and GRA sites, while above- and belowground biomass C accounted for most of the variation of total ecosystem C storage among those systems. Despite intensive management of the agricultural system and hybrid poplar plantations, SOC input appears to be balanced by output resulting in similar SOC storage among the different land uses (Figure 2-3). It will be noted, however, that a non-significant decrease of SOC was observed in the 2HP and a slight increase in the 9HP (Table 2-3, Figure 2-3). Soil C loss in the early stages of plantation development may be due to organic C inputs greater than organic C losses through soil respiration brought about by cultivation during site preparation and weed management. The SOC in 9HP, on the other hand, increased by 4.6 Mg C ha^{-1} from AG site, or an increase of 51.1 g C m^{2} yr^{-1}. This is within the range of rates of C sequestration (9.4 g C m^2 yr^{-1} to 65.7 g C m^2 yr^{-1}) after converting agricultural land use to forestry in cool temperate zones reported by Post and Kwon (2000). Early soil C loss under young (<6 years old) hybrid poplar plantations followed by increased soil C gain with plantation age was similarly observed by Hansen (1993). Findings are applicable only to sites with similar conditions due to the comparative mensurative experimental design.

A large proportion (60 to 67%) of the total soil organic C and N was stored in the top 20 cm soil, except in NAT, where large amounts of C and N were in the surface residues, i.e. coarse woody debris and LFH (Table 2-3). The accumulation of litter in the humus layer and lack of mixing of the surface litter material with the mineral soil resulted in the low C and N stocks in the mineral soil of NAT.

High C and N stocks in the fine particle size fraction (Table 2-4, Figure 2-4) were related to the high recovery of silt and clay (average of 59 %) plus their high C and N concentrations. High C content in fine particle size fractions could result from a combination of factors including: lower decomposition rates of silt- and clay-associated organic matter, the transfer of stabilized decomposition products from other size separates, and the accumulation of more stable soil organic matter (Christensen 1992). Examining C and N stock changes among soil particle size fractions, it was found that C and N stocks doubled in the coarse-size fraction nine years after hybrid poplar plantation establishment as compared with the agricultural soils whilst no other changes within the other particle size fractions were observed. This suggests that after the soil disturbance and mixing by cultivation stopped in the hybrid poplar plantations there was build up of organic matter in the coarse-size fraction nine years after plantation establishment (Six et al. 2000, Six et al. 2000). Similarly, a slower SOC incorporation rate in smaller particle size fractions is expected in NAT where the ecosystem is undisturbed and where soil aggregation is expected to be high compared to the other land uses. The C:N ratio decreased with decreasing size fraction for all land uses. This trend was expected as coarser fractions generally contain undecomposed or partially decomposed SOM characterized with lower degrees of humification compared to finer-size fractions (Christensen 2001).

The ^{13}C abundance in litter were well-within the range reported (-21 to -30 ‰) in C_3-dominated land uses sites (Ehleringer et al. 2000, Balesdent et al. 1993, Bird and Pousai 1997). Increasing ^{13}C enrichment of SOC with decreasing particle size indicated different portions of young plant residues in the size fractions. The δ^{13}C values measured in the coarser-size fractions were close to that of the litter, implying that SOC contained in coarser fractions was younger and more labile as compared to SOC in finer fractions. Similar trends have been found in many other studies and such trends were mainly due to ^{13}C discrimination during microbial degradation of organic carbon (Bird and Pousai 1997, Bird et al. 2002, Bol et al. 1999). Enrichment of ^{15}N within particle size fractions may be attributed to ecosystems experiencing increasing decomposition and higher N losses whilst depletion of ^{15}N may be attributed to SOM age and to ecosystems experiencing a closed N cycle i.e. minimal N losses (Preston et al. 2000). Land conversion from AG to 2HP, 9HP, and GRA generally caused ^{13}C and ^{15}N enrichment within the particle size fractions (with a few exceptions), suggesting that organic matter had undergone much decomposition and that the remaining C has become more stable within these fractions. On the other hand, conversion of the NAT to AG, decreased ^{13}C and ^{15}N across all size fractions, and thus would suggest the presence of more labile organic matter resulting in higher soil C loss. Although conversion of agriculture to hybrid poplar plantations did not change SOC stocks, a closer examination of ^{13}C and ^{15}N within particle size fractions showed that afforestation with hybrid poplar plantations have the potential to sequester C rapidly and stabilize C in fine soil particle size fractions in the long term.

Conclusions

Land use, soil cultivation, and afforestation significantly affected total ecosystem C stocks although management-induced differences in C stocks were confined to differences in above- and belowground biomass C. Neither land conversion from the native aspen forest to agriculture 80 years ago, nor hybrid poplar plantation or grassland establishment on agricultural land nine years ago, brought significant changes in SOC stocks. The forest floor in the native aspen stand contained a sizeable fraction of the total ecosystem C stock and must be assessed in determining C budgets for land use and land cover changes. The SOC associated with the coarse-size fraction increased nine years after hybrid poplar plantation establishment. Most of the C stocks were located in the fine-size fraction of the soil and were seemingly unaffected by land use change although changes in ^{13}C and ^{15}N abundances within these particle size fractions did suggest formation of more recalcitrant organic matter in response to establishment of hybrid poplar plantations on previously cultivated agricultural soil.

Tables and Figures

Table 2-1. Physical properties (mean with standard errors in parentheses, n = 9) of 0-20 and 20-50 cm soil layers under different land use systems in Linaria, AB, Canada.

Land use	BD (g cm^{-3})	pH	Silt (%)	Clay (%)	Textural Class
			0 -20 cm		
AG	1.46 (0.05) A a	5.29 (0.24) A a	41 (2.08) A a	29 (2.33) B a	clay loam
2HP	1.30 (0.03) B a	5.12 (0.03) B a	43 (0.33) A a	27 (0.33) B a	loam
9HP	1.43 (0.08) B a	5.17 (0.04) B a	41 (1.00) A a	26 (0.88) B a	loam
GRA	1.41 (0.03) A a	5.35 (0.04) A a	38 (1.20) A a	26 (0.88) A a	loam
NAT	1.41 (0.11) A a	4.86 (0.12) A a	34 (3.67) A a	31 (5.03) A a	clay loam
			20-50 cm		
AG	1.59 (0.02) A ab	6.59 (0.53) A a	19 (1.20) B b	52 (2.62) A a	clay
2HP	1.51 (0.05) A b	6.21 (0.27) A ab	22 (2.08) B ab	50 (1.53) A ab	clay
9HP	1.73 (0.05) A a	6.33 (0.10) A a	22 (1.32) B ab	36 (1.44) A bc	clay loam
GRA	1.56 (0.05) A ab	6.01 (0.47) A ab	27 (7.33) A ab	39 (5.09) A ab	clay loam
NAT	1.52 (0.04) A b	4.65 (0.15) A b	38 (2.95) A a	32 (1.92) A c	clay loam

Values with different upper- and lowercase letters are significantly different between soil layers for each land use system and between land use systems for each soil layer, respectively, at $P < 0.05$ (Tukey’s Studentized Ranged Test $\alpha = 0.05$).

Table 2-2. Chemical properties (mean with standard errors in parentheses) of surface (0-20 cm) and subsurface (20-50 cm) soil layers under different land use systems in Linaria, AB, Canada.

Land use	NH_4^+ (g m^{-2})	NO_3^- (g m^{-2})	P (g m^{-2})	K (g m^{-2})	Ca (g m^{-2})	Mg (g m^{-2})	CEC (cmol kg^{-1})
				0-20 cm			
AG	4.33 (2.89) A a	6.36 (1.81) A a	4.56 (0.86) A a	76.3 (14.3) A ab	875 (170) B a	130 (41) B a	18.25 (2.29) B a
2HP	1.64 (0.08) A a	3.40 (0.69) B b	3.98 (0.18) A a	52.0 (0.6) A ab	839 (41) B a	83 (7) B a	16.32 (0.42) B a
9HP	2.15 (0.21) A a	0.57 (0.23) A c	1.73 (0.18) A b	46.6 (11.5) A b	914 (52) B a	116 (9) B a	19.47 (2.46) A a
GRA	2.21 (0.01) A a	0.72 (0.06) A c	1.44 (0.19) A b	42.4 (7.0) A b	1179 (84) B a	114 (12) B a	18.58 (1.40) A a
NAT	2.26 (0.39) A a	0.62 (0.18) A c	3.68 (1.93) A a	87.2 (14.2) A a	1146 (94) A a	131 (45) A a	18.93 (1.44) A a
				20-50 cm			
AG	1.84 (0.14) A a	2.28 (0.43) B b	1.66 (0.77) B a	67.8 (10.7) A b	3210 (678) A a	601 (107) A a	25.83 (1.68) A ab
2HP	2.31 (0.28) A a	9.34 (0.51) A a	1.15 (0.17) B a	62.0 (5.5) A b	2708 (285) A a	522 (21) A ab	28.16 (1.60) A a
9HP	2.72 (0.13) A a	0.45 (0.07) A c	1.39 (0.36) A a	53.6 (2.6) A b	2654 (378) A a	448 (3) A b	23.61 (1.92) A ab
GRA	3.39 (0.89) A a	0.54 (0.04) A c	1.39 (0.19) A a	59.8 (5.4) A b	2399 (185) A ab	414 (11) A b	21.10 (1.84) A bc
NAT	3.32 (0.41) A a	0.61 (0.05) A c	2.80 (0.52) A a	112.6 (30.8) A a	1694 (147) A b	253 (39) A c	18.29 (1.74) A c
				LFH			
NAT	0.66 (0.09)	0.01 (0.00)	0.16 (0.02)	6.0 (0.4)	74 (5)	6.0 (0.6)	70.38 (4.93)

NH_4^+, ammonium; NO_3^-, nitrate; P, available phosphorus; K, exchangeable potassium; Ca, exchangeable calcium; Mg, exchangeable magnesium; CEC, cation exchange capacity. Values with different upper- and lowercase letters are significantly different between soil layers (except the LFH horizon) for each land use system and between land use systems for each soil layer, respectively, at $P < 0.05$ (Tukey’s Studentized Ranged Test $\alpha = 0.05$).

Table 2-3. Total C and N (mean with standard errors in parentheses) in above- and belowground biomass, soil, and site in different land use systems in Linaria, AB, Canada. All are in Mg ha^{-1}.

Land use	Aboveground biomass C		Belowground biomass C (0-30 cm)			
	Foliage	Total aboveground biomass C	Coarse root	Fine root	Microbial biomass	Total belowground biomass C
AG	-	4.8 (0.3) b	-	0.86 (0.20) bc	1.58 (0.25) ab	2.4 (0.4) b
2HP	0.4 (0.1) c	1.6 (0.2) b	0.47 (0.05) b	0.61 (0.21) c	0.70 (0.07) b	1.8 (0.2) b
9HP	4.2 (0.5) b	39.6 (5.0) a	9.43 (1.20) b	3.50 (0.34) abc	2.68 (0.43) a	15.6 (1.0) b
GRA	-	1.8 (0.3) b	-	3.73 (1.25) ab	1.81 (0.46) ab	5.5 (0.8) b
NAT	6.9 (2.2) a	49.2 (14.8) a	31.6 (10.05) a	4.26 (0.41) a	1.50 (0.05) ab	37.4 (9.7) a

Land use	Litter		Soil C		
	LFH	Snags and CWD	0-20 cm	20-50 cm	Total soil C (0-50 cm)
AG	-	-	75.41 (2.28) A ab	39.33 (5.48) B a	114.8 (7.0) a
2HP	-	-	67.64 (2.46) A ab	39.14 (0.30) B a	106.8 (2.2) a
9HP	-	-	78.23 (8.76) A a	41.16 (8.20) B a	119.4 (14.2) a
GRA	-	-	75.59 (4.58) A ab	38.02 (3.73) B a	113.6 (8.3) a
NAT	11.1 (1.47)	23.1 (5.47)	55.28 (17.58) A b	47.29 (5.93) A a	136.8 (17.0) a

Land use	Soil N			
	LFH	0-20 cm	20-50 cm	Total soil N (0-50 cm)
AG	-	6.54 (0.31) A a	3.27 (0.26) B a	9.81 (0.57) a
2HP	-	6.02 (0.15) A a	3.93 (0.33) B a	9.95 (0.19) a
9HP	-	6.98 (0.85) A a	3.53 (0.69) B a	10.50 (1.46) a
GRA	-	6.60 (0.31) A a	3.28 (0.13) B a	9.88 (0.91) a
NAT	0.59 (0.06)	4.14 (1.05) A b	4.44 (0.49) A a	9.17 (0.50) a

Values with different upper- and lowercase letters are significantly different between soil layers (except the LFH and snags and coarse woody debris) for each land use system and between land use systems for each soil layer, respectively, at $P < 0.05$ (Tukey's Studentized Ranged Test $\alpha = 0.05$).

Table 2-4. Soil recovery in different particle size fractions, C and N contents, and C:N ratios (mean with standard errors in parentheses) in the 0-10 cm soil under different land use systems in Linaria, AB, Canada.

Land use	Recovery (%)	C (Mg ha^{-1})	N (Mg ha^{-1})	C:N
		Coarse-size fractions		
AG	5.69 (0.85)	4.30 (0.65) C b	0.37 (0.05) C b	19 (1.2) A a
2HP	7.13 (0.21)	4.71 (0.21) C ab	0.42 (0.02) C b	17 (0.5) A ab
9HP	12.23 (1.04)	9.82 (1.87) B a	0.88 (0.18) B a	15 (0.2) A ab
GRA	11.88 (1.12)	6.88 (1.08) C ab	0.60 (0.09) C ab	13 (1.4) A b
NAT	11.99 (0.64)	3.37 (1.17) A b	0.25 (0.07) B b	18 (1.1) A a
		Medium-size fractions		
AG	27.80 (1.80)	20.95 (0.69) B a	1.82 (0.09) B a	13 (0.6) B a
2HP	28.80 (1.37)	19.03 (1.28) B a	1.69 (0.09) B a	12 (0.4) B a
9HP	30.56 (1.00)	23.92 (1.92) B a	2.13 (0.19) B a	12 (0.5) B a
GRA	31.64 (1.84)	18.04 (0.83) B a	1.58 (0.06) B a	13 (0.3) A a
NAT	32.52 (1.16)	9.23 (3.20) A b	0.69 (0.19) AB b	13 (0.5) B a
		Fine-size fractions		
AG	66.16 (1.83)	50.16 (2.90) A a	4.35 (0.32) A a	9 (0.6) C a
2HP	61.44 (1.99)	40.52 (1.21) A ab	3.61 (0.09) A ab	9 (0.5) C a
9HP	56.41 (0.25)	44.49 (4.99) A ab	3.97 (0.49) A ab	9 (0.2) C a
GRA	55.46 (1.22)	31.77 (2.13) A b	2.77 (0.15) A b	9 (0.3) A a
NAT	54.73 (1.57)	15.04 (4.46) A c	1.13 (0.27) A c	9 (0.3) C a

Particle sizes: Coarse-size fractions (250-2000 μm); medium-size fractions (53-250 μm); fine-size fractions (< 53 μm). Values with different upper- and lowercase letters are significantly different among soil particle size fractions for each land use system and between land use systems for each soil particle size fraction, respectively, at $P < 0.05$ (Tukey's Studentized Ranged Test $\alpha = 0.05$).

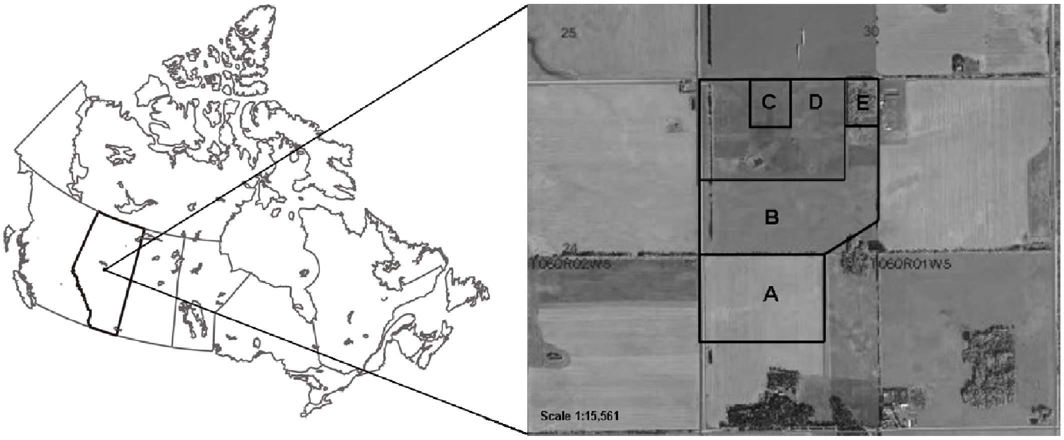

Figure 2-1. Location of land use study sites (A – agriculture; B – 2-yr-old hybrid poplar plantation, C – 9-yr-old hybrid poplar plantation; D – grassland; E – native aspen stand) in Linaria, Alberta, Canada. The source of the aerial photo is http://www.agriculture.alberta.ca. The use of the aerial photo by the author is done without any affiliation with or endorsement by the Government of Alberta. Reliance upon the author's use of this material is at the risk of the end user.

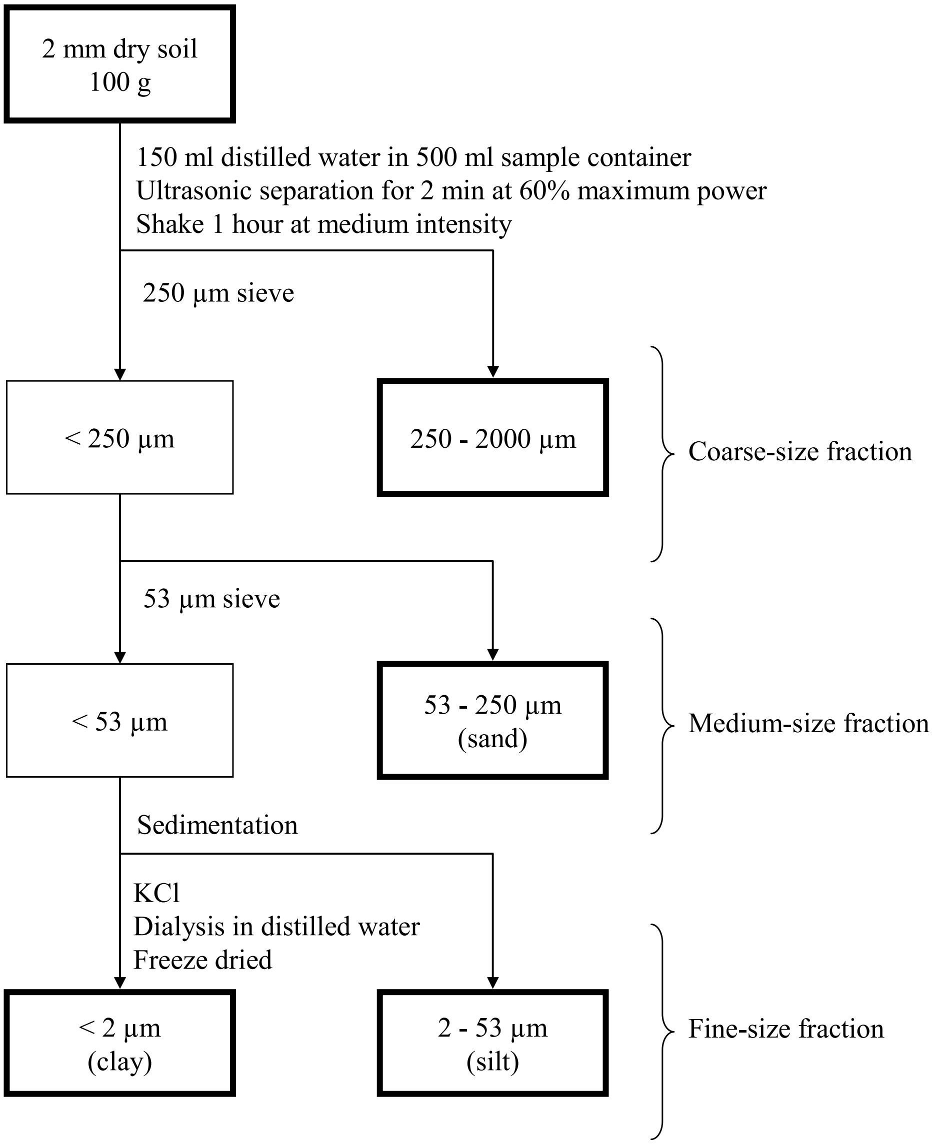

Figure 2-2. Diagram of the particle size fractionation scheme. The fractionation sequence results in the following four size sieve fractions: Coarse-size fraction, 250-2000 µm; medium-size fraction, 53-250 µm; and fine-size fraction (combined silt-, 2-53 µm, and clay-size fraction, < 2 µm).

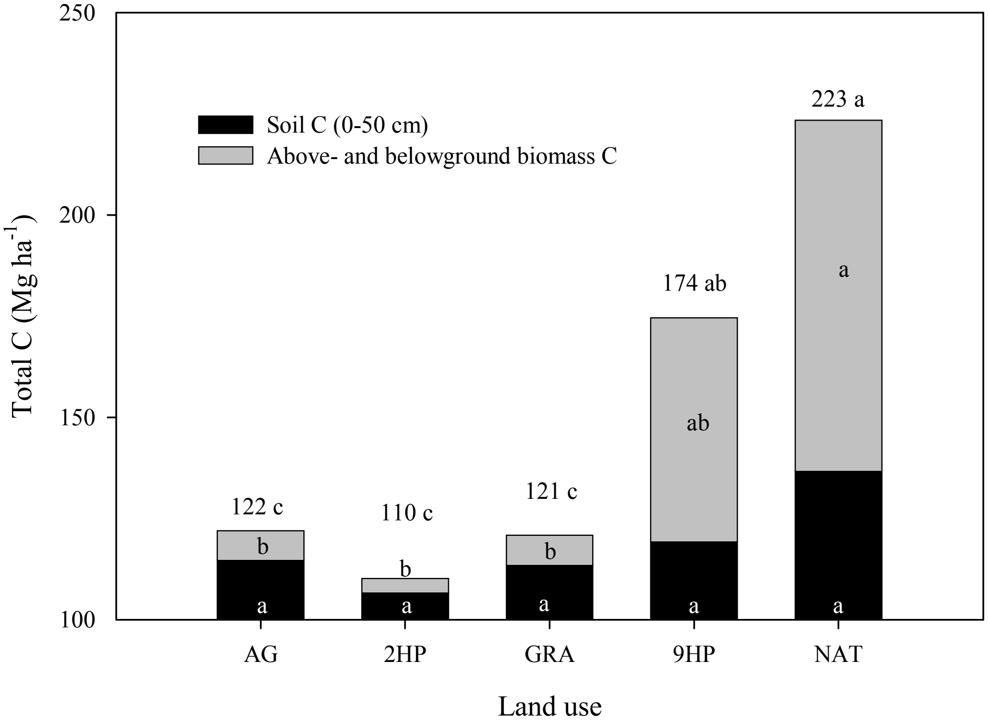

Figure 2-3. Distribution of C under different land uses (AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, GRA – grassland, 9HP – 9-yr hybrid poplar plantation, NAT – native aspen) in Linaria, AB, Canada. Values with different letters are significantly different between land use systems at $P < 0.05$ (Tukey's Studentized Ranged Test $\alpha = 0.05$).

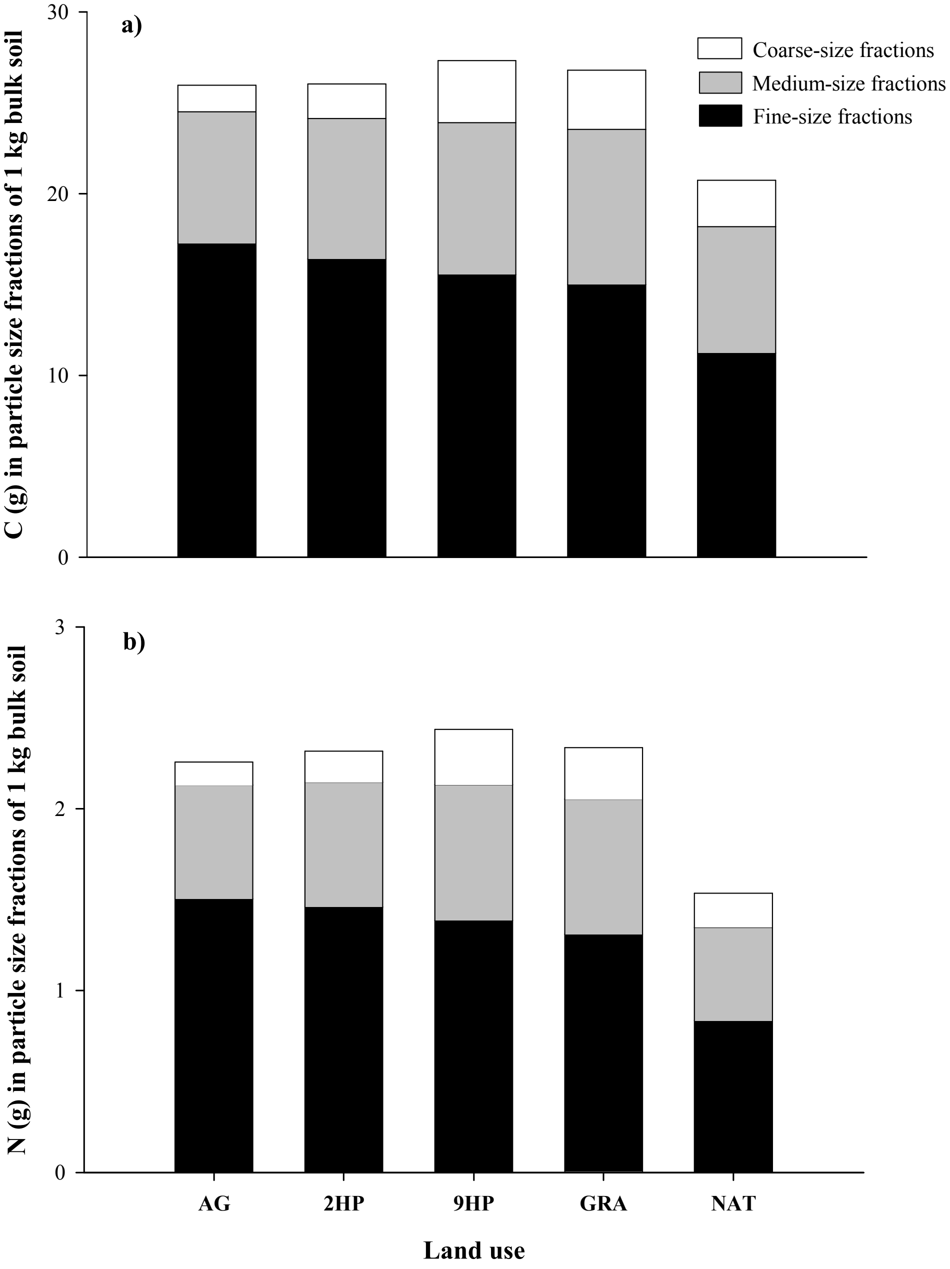

Figure 2-4. Distribution of a) soil organic C and b) total organic nitrogen mass in 1 kg soil by particle size fractions within 0-10 cm depth soils under different land uses (Ag – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP – 9-yr hybrid poplar plantation, GRA – grassland, NAT – native aspen) in Linaria, AB, Canada. No differences in C or N concentrations in particle size-fractions were determined among land uses.

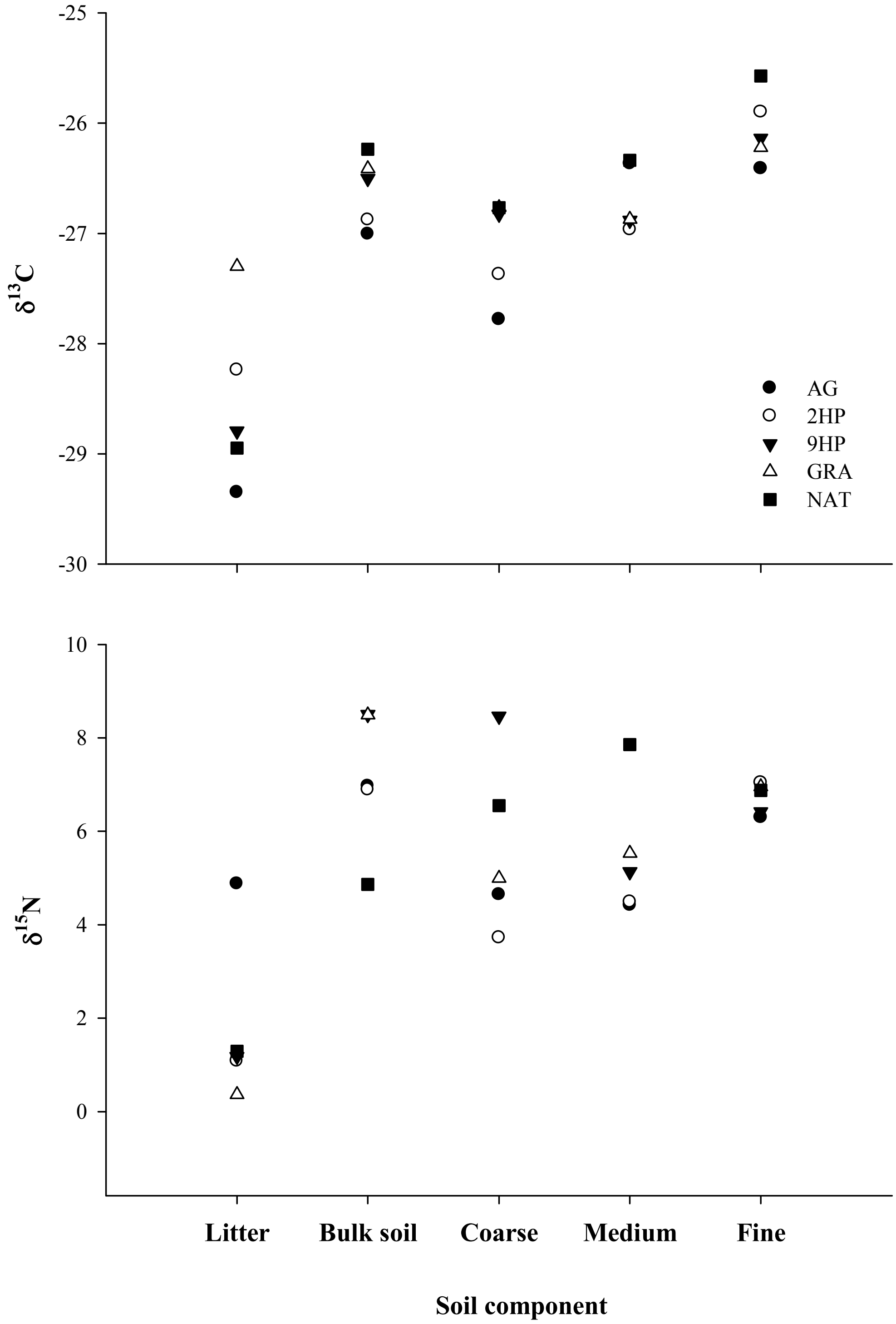

Figure 2-5. Distribution of a) $\delta^{13}C$ and b) $\delta^{15}N$ in litter, bulk soil, and particle size fractions of soils under different land uses.

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Chapter 3. Soil respiration in four different land use systems[*]

Introduction

Concerns over increasing atmospheric carbon dioxide (CO_2) concentrations and its likely effects on the global climate prompted extensive research on all aspects of the global carbon (C) cycle. Soils play an important role in the global C cycle because they both store and release large quantities of C to the atmosphere through soil respiration (R_s). In addition to fire, R_s is the primary pathway through which C fixed through photosynthesis returns to the atmosphere (Raich et al. 2002). Soil respiration can account for up to 90% of total ecosystem respiration (Hanson et al. 2000) and is therefore one of the major components to consider in understanding ecosystem-atmosphere C exchange. There is considerable interest in the effects of land use change on soil C fluxes and storage (Guo and Gifford 2002, Hibbard et al. 2005, Lal 2004, Post and Kwon 2000) specifically on R_s (Ellert and Janzen 1999, Franzluebbers 2005). Houghton (1999) estimated that globally land use change caused approximately 123 Pg C to be released to the atmosphere between 1850 and 1990, with the most important land use change being the expansion of agriculture, with croplands, forests-pasture and shifting cultivation accounting for 68, 13, and 4%, respectively, of the net ecosystem C loss to the atmosphere.

The production of CO_2 in the soil is primarily due to autotrophic respiration (R_a) by roots and heterotrophic respiration (R_h) by microbes decomposing detritus, root exudates, and organic matter (Hanson et al. 2000). The R_h ultimately controls soil C storage while R_a reflects plant activity and the allocation of organic compounds to root biomass from aboveground (Binkley et al. 2006). Partitioning R_s to R_h and R_a is therefore important in understanding C cycling because R_h and R_a may respond differently to environmental controls,

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implying different behaviors at different time scales and for different plant communities. Recent reviews by Hanson et al. (2000), Bond-Lamberty et al. (2004) and Subke et al. (2006) have indicated that R_a generally accounts for 50% of R_s across a range of ecosystems and time scales. Hanson et al. (2000), based on results in 50 publications, reported mean R_a contributions of 48 and 37% in forest and non-forest ecosystems, respectively. In addition, the contribution of R_a has large seasonality, usually low during the dormant season and high during the active growing season (2000). The dynamics of the two components, R_a and R_h, is controlled by several abiotic and biotic factors including soil temperature (Rustad et al. 2001), soil water content (Liu et al. 2002), soil drainage (Freeman et al. 1993), soil fertility (Butnor et al. 2003), availability of C substrates for microorganisms (Seto and Yanagiya 1983), plant root densities and activities (Maier and Kress 2000), plant photosynthetic activity (Bhupinderpal-Singh et al. 2003), and soil organism population size (Rai and Srivastava 1981). All of these variables are influenced by land use or land cover type (Raich and Tufekcioglu 2000).

Understanding the factors that affect R_s is essential to assess the impacts of land use change and predict outcomes under future land use scenarios. Although there have been many studies on R_s, there have been few studies that quantified the effects of land use change i.e., vegetation change on a specific site. I studied R_s in four adjacent ecosystems: native aspen forest, 2- and 9-year-old hybrid poplar plantations, grassland, and agriculture. The objectives of this study were: (i) to measure the seasonal and diurnal changes in R_s under different land uses, (ii) to partition R_s into R_h and R_a, and (iii) to examine the factors affecting variations in R_s, R_h, and R_a under different land uses.

Materials and Methods

Site description

The study sites are located near Linaria (54°12’ N, 114°8’ W), approximately 25 km west of Westlock, in north central Alberta, Canada (Figure 3-1). The study area has a continental climate, with cold winters and warm summers which provide an approximate growing season of 180-185 days between the months of May to November (AAFRD 2003). Climate normals for the study area, based on 12 years of data collected at nearby Westlock weather station (Environment Canada 2005) show that the area has a mean annual temperature of 3 °C and mean annual precipitation of 463 mm (350 mm of which falls as rain and 113 mm as water equivalent of snow). The soil is classified as a Dark Gray Luvisol in the Canadian system of soil classification on medium textured till characterized by poor drainage, and undulating, low relief landform with a slope of 2%.

Three experimental plots (20 × 20 m^2) were set up in four different land use systems in this study: (i) an agricultural field (AG) used for grains and canola production following a four-year rotation of barley (*Hordeum vulgare* L.) – barley – wheat (*Triticum aestivum* L.) – canola (*Brassica napus* L.), (ii) 2- and 9-yr-old hybrid poplar (*Populus deltoides* x *Populus x petrowskyana* cv. Walker) plantations (2HP and 9HP, respectively) with three plots in each plantation age, (iii) a grassland (GRA) seeded with a mixture of *Festuca arundinacea* Schreb. (tall fescue), *Dactylis glomerata* L. (orchard grass), and *Trifolium pratense* L. (red clover), and (iv) a native forest stand (NAT) of trembling aspen (*Populus tremuloides* Michx.). Detailed description of the four different land use sites (including soil physical and chemical properties) are given in Chapter 2.

Measurement of soil CO_2 concentrations and diffusivity

Soil CO_2 concentrations were measured at two depths every two hours, from May 22, 2007 to November 19, 2007, using Vaisala CARBOCAP solid-state infrared CO_2 sensors (Models GMM221 and GMM222; Vaisala Oyj, Helsinki, Finland). At the center of one 20 m × 20 m plot in each of the study sites, one GMM222 probe (measurement range of 0-10000 μmol CO_2 mol^{-1}) was installed horizontally at 2 cm depth while a GMM221 (range of 0-20000 μmol CO_2 mol^{-1}) was installed vertically at 20 cm depth. The sensors were separated horizontally by 5 cm. Dataloggers (Model CR-10X, Campbell Scientific Inc., Logan, UT, USA) were used to record CO_2 concentration measurements. The systems were powered by two 12 V batteries connected in series, charged by photovoltaic panels (5-15 Watts).

To determine soil CO_2 diffusivity on each of the study sites, undisturbed soil cores (11 cm inside diameter, 10 cm height) were sampled at 0-10 and 10-20 cm soil depths in the vicinity of the collars and Vaisala probes. Diffusivity in these soil cores was measured at different soil water content values. Measurements were made, under steady state conditions using CO_2 as the diffusing gas using the technique described in Jassal et al. (2005). Calculations for soil CO_2 diffusivity depend on various factors i.e. total porosity, tortuosity, and water content (White 2006). These factors change when soil temperatures decrease below 0 °C when soil water starts to freeze. Diffusivity calculated in the laboratory therefore cannot be used to represent values for the winter months in this study.

Chamber measurement of R_s and R_h

To validate R_s, calculated using soil CO_2 concentrations and diffusivity measurements, chamber measurements were made with a LI-COR 6400 portable photosynthesis system equipped with a LI-COR 6400-09 portable opaque soil chamber attachment (LI-COR Biosciences Inc., Lincoln, NE, USA). These measurements were made at 15 to 27 locations in each land use site using "collars" made from polyvinyl chloride (PVC) pipe (10 cm inside diameter and 6 cm long), beveled on one end to reduce soil compaction and disturbance during insertion into the soil. These collars were gently hammered approximately 2 cm into the soil, leaving about 4 cm above the soil surface and were left in place for the duration of the study.

As trees in the hybrid poplar plantations were planted operationally on a square spacing and such configuration is known to cause spatial variation in soil respiration (Saurette et al. 2006), a stratified random sampling design for the locations of the soil collars (distance from trees) was used to account for the spatial variability in soil properties and respiration rates. An equal number of collars were placed close to the base of a tree, mid-way between two trees, and at the center of the square formed by four trees in each plot. Each spatial arrangement of collars was replicated three times, which resulted to nine collars per plot or 27 collars per site. Collars at AG, GRA, and NAT were installed using a random sampling design using five collars per plot (or 15 collars per site). The R_s measurements on all the collars were made every four weeks during the growing seasons of 2006 (June 28, July 27, August 24, and September 26) and 2007 (May 29, June 28, July 26, August 30). All R_s measurements were carried out between 10:00 and 16:00 h (local summer time).

To determine the contribution of R_h to R_s, a modified root exclusion trenched plot technique was used following Kelting et al. (1998). This relied on severing all the roots around the perimeter of PVC collars in order to eliminate root respiration. For this purpose, nine PVC collars (10 cm inside diameter and 32.5 cm long) per study site were inserted (on May 17, 2006) 30 cm into the soil. In order to avoid plant activity, vegetation within the collars was regularly removed by clipping. Measurements of R_s (from the short collars) and R_h (from the long collars) were made at four-week intervals (June 28, July 27, August 24, and September 26 of 2006). The contribution of R_a was then determined by subtracting R_h from R_s.

Soil temperature and moisture measurements

Soil temperature (T_s) and soil volumetric water content (θ_v) were measured from May 30, 2006 to November 19, 2007 using HOBO U12 temperature sensors (Model TMC6-HD, Onset Computer Corp., Bourne, MA, USA) and porous matrix tensiometers (Watermark Sensor type WMSM, Delta-T Devices Ltd., Cambridge, UK), respectively. These sensors were also installed at the 2 and 20 cm depths but approximately 30 cm away from the Vaisala CO_2 sensors. The temperature and water content sensors were connected to a HOBO U12 external datalogger (Model U12-006, Onset Computer Corp., Bourne, MA, USA) programmed to take readings every 2 h when CO_2 concentrations were measured by the Vaisala sensors.

Soil temperature and soil volumetric water content were also measured manually at the 0-6 cm depth near (within 20-30 cm) each PVC collar. Soil temperature was measured using a soil temperature probe (LI-COR 6000-09TC, LI-COR Biosciences Inc., Lincoln, NE, USA), while θ_v was measured with a portable soil-water frequency domain reflectometry sensor (ThetaProbe ML2-X, Delta-T Devices, Cambridge, UK).

Calculation of soil respiration rates

Using measured soil CO_2 concentrations at the 2 and 20 cm depths and ambient atmospheric CO_2 concentration at the soil surface (zero depth), I calculated CO_2 flux in the soil at the 1- and 11-cm depths (midpoints of 0-2 cm and 2-20 cm layers) using Fick's law of diffusion:

$$F_i = -D\frac{\partial C}{\partial z} \tag{1}$$

where F_i is the soil CO_2 flux (μmol m^{-2} s^{-1}) at depth *i* (m), D_i is the diffusivity of CO_2 in the soil (m^{-2} s^{-1}) at depth i, and $\partial C/\partial z$ is the vertical soil CO_2 concentration gradient (μmol m^{-4}). Linear gradients were assumed between the 0 and 2 cm depths and between the 2 and 20 cm depths separately, based on the shape of soil

CO_2 profiles with concentration measurements at discrete soil depths reported in the literature (Jassal et al. 2005, Drewitt et al. 2005). My periodic chamber measurements indicated that ambient CO_2 concentrations at the soil surface generally varied between 380 and 425 ppm, except in the native aspen stand where they were somewhat higher. The F_1 was assumed to represent the total CO_2 efflux originating at the soil surface and is referred to hereafter as R_s; F_{11} represents the efflux originating from below the 11-cm depth, referred to as R_{sub}. As NAT had about 7.5-10 cm LFH (Litter-Fibric-Humus) layer, R_s was also measured in NAT after removing the LFH layer and is referred to as $R_{NAT\text{-}LFH}$.

Diffusivities of CO_2 in the soil were computed using the equation:

$$D_s = \xi D_a \tag{2}$$

where ξ is the gas tortuosity factor and D_a is the molecular diffusivity of CO_2 in air. D_a was corrected for variations in temperature and barometric pressure using the equation:

$$D_a = D_{a0}\left(\frac{T_a}{293.15}\right)^{1.75}\left(\frac{101.3}{P}\right) \tag{3}$$

where T_a is the air temperature in K, P is the air pressure in kPa, D_{a0} is the CO_2 diffusivity in air at 293.15 K and 101.3 kPa, which is given as 14.7 $mm^2\ s^{-1}$ (Jones 1992). The tortuosity factor, ξ, obtained using measured values of D_s and ε was found to be independent of soil depth and soil matrix, and was parameterized with air-filled porosity using a non-linear relationship (Curie 1965),

$$\xi = n\varepsilon^m \tag{4}$$

where ε is the soil air-filled porosity and the n and m are coefficients determined empirically from the undisturbed soil cores. Air-filled porosity was calculated based on its relationship with soil bulk density (ρ_b), particle density (ρ_m), total soil porosity (φ), and volumetric water content (θ_v):

$$\varepsilon = \phi - \theta_v = 1 - \frac{\rho_b}{\rho_m} - \theta_v \tag{5}$$

In order to parameterize R_s, regression analyses were performed on R_s (obtained by the LICOR 6400-09) and T_s or θ_v measurements. Exponential relationships between R_s and T_s were found to be the best fit:

$$R_s = a \exp^{bT_s} \tag{6}$$

where R_s is the soil surface respiration at soil temperature T_s, coefficient a is the soil respiration rate at temperature zero (i.e., basal rate), and coefficient b is the sensitivity of R_s to T_s. The regression coefficients a and b determined for each land use were used to estimate continuous 2-hourly values of R_s, which were then used to compute daily and seasonal R_s for each study site and year.

In order to determine temperature sensitivity of soil from each of the land uses, the same exponential relationship was fit and the b values in equation (6) were used to calculate Q_{10}:

$$Q_{10} = \exp^{10b} \tag{7}$$

where Q_{10} is the relative increase in R_s for every 10 °C increase in T_s. Mean values of Q_{10} were determined for each land use site using data from the two study years.

Statistical analyses

Cumulative soil CO_2 efflux through R_s for each growing season (June 1 to September 30 for 2006 and 2007) for each land use type was calculated by adding all the daily R_s values. Uncertainty associated with cumulative soil CO_2 efflux through R_s and its components were estimated by assigning a random (measurement) error of 20% to each daily value. The daily fluxes with the ±20% variation were re-sampled using bootstrap Monte Carlo method and seasonal sums calculated. This procedure was repeated 500 times and uncertainty was determined at 95% confidence level. Temporal variations in R_h, R_a, and the contribution of R_h to R_s (i.e., R_h/R_s), both among different land use types and from month to month, were analyzed using analysis of variance (ANOVA) with repeated measures coupled with Tukey's Studentized Range Test in SAS (SAS

Institute Inc. 2004). Interaction effects were elucidated using the *pdiff* option in SAS. To determine correlations of R_h and R_a and soil biophysical variables (pH, soil organic N and C, particulate organic matter, NH_4^+, NO_3^-, phosphate, MBC, MBN, and fine root biomass) that were simultaneously measured as part of Chapter 2, Pearson product-moment correlations were carried out in SAS. Significance was tested using *α-value* = 0.05, except for Pearson correlations where two levels of significance were recognized: weakly significant (*α-value* = 0.10) and significant (*α-value* = 0.05).

Results

Weather

Year 1 (June 1-September 30, 2006) was slightly warmer with the seasonal mean T_a of 15.5 °C compared to 14.6 °C in Year 2 (June 1-September 30, 2007, Figure 3-2); and close to the long-term (1994-2006) seasonal mean (15.0 °C) for the area (Environment Canada 2005). Year 1 was also slightly wetter with a total of 271 mm rainfall during the four months summer period, compared to 226 mm during the same period in Year 2, although both were within the normal seasonal range of 158-415 mm recorded for the area.

Mean daily soil temperature at the 2 and 20 cm depths followed similar seasonal trends at all land use sites (Figure 3-2), with T_s in the range 4-39 °C, peaking in mid-July. These temperatures were consistently somewhat cooler at 9HP and NAT compared to AG, 2HP, and GRA. Soil water content was highest in early June, as a result of snow melt, and remained in the range 0.13-0.45 $m^3\ m^{-3}$ throughout both growing seasons. Soil moisture status at all land use sites was within optimum conditions (field capacity = 0.40-0.50 $m^3\ m^{-3}$, permanent wilting point = 0.13-0.17 $m^3\ m^{-3}$).

Chamber- and gradient-measured soil respiration

A comparison of chamber-measured R_s values with those calculated using soil CO_2 concentration gradients and diffusivities (equations 1 and 2) showed that the two agreed very well (slope 1.008, $r^2 = 0.96$, Figure 3-3). Fitting equation 6 to chamber-measured R_s values from Year 1 and Year 2 showed that T_s (range: -2.5 to 27.0 °C) explained 88-94% of the variation in R_s (Table 3-1). On the other hand, θ_v predicted R_s poorly ($r^2 = 0.12$-0.42, data not shown). The calculated Q_{10} values differed among land use types in the order of GRA (1.88) < NAT (1.93) < 9HP (2.11) < 2HP (2.21) < AG (2.70).

Seasonal soil respiration under different land uses

The R_s ranged from a minimum of 0.12 to 0.24 μmol CO_2 m^{-2} s^{-1} to a maximum of 6.2 to 11.6 μmol CO_2 m^{-2} s^{-1} depending on land use and time of year (Figure 3-4). During both years, temporal variation in R_s corresponded closely to seasonal changes in T_s for all land uses (Figures 3-2 and 3-4), reaching maximum rates between mid-June and mid-July when soil temperatures were highest. The lowest rates were observed in late fall. Chamber-measurements made at 15-27 collars within each land use indicated lesser spatial variability in the 2HP, GRA, and AG (S.D. of 0.35, 0.38, and 0.59 μmol CO_2 m^{-2} s^{-1}, respectively), than in the 9HP and NAT (S.D. of 0.69 and 0.97 μmol CO_2 m^{-2} s^{-1}, respectively).

Cumulative seasonal (June 1 to September 30) soil C efflux due to R_s was generally higher in Year 1 than in Year 2 for all land uses (Table 3-2). Averaged across the two growing seasons, R_s was highest in NAT (781 g C m^{-2}) of which approximately 75% (586 g C m^{-2}) originated from the LFH layer and the balance from the mineral soil ($R_{NAT-LFH}$ = 194 g C m^{-2}). Other land uses released smaller amounts of C, which, averaged over the two growing seasons, ranked in the order: 9HP (551 g C m^{-2}) > GRA (523 g C m^{-2}) > AG (502 g C m^{-2}) > 2HP (428 g C m^{-2}). The seasonal soil CO_2 efflux originating from R_{sub}, varied from a minimum of 136 g C m^{-2} in NAT, followed by 2HP (210 g C m^{-2}), 9HP (216 g C m^{-2}), GRA

(243 g C m^{-2}), to a maximum of 358 g C m^{-2} in AG. I found that R_s was highly related to aboveground biomass C ($r = 0.66$, *p-value* = 0.008), belowground biomass C ($r = 0.77$, *p-value* = 0.001), total ecosystem biomass C ($r = 0.80$, *p-value* = < 0.001), and marginally related to soil C ($r = 0.54$, *p-value* = 0.036).

Diurnal variation in soil respiration under different land uses

Like seasonal variations in R_s, the two-hourly measurements of R_s corresponded closely to diurnal temperature variations (Figure 3-5). The R_s originating from the mineral soil (excluding the LFH layer in NAT) increased after 08:00 h and peaked at around 15:00-17:00 h for all land uses during the early, middle, and late parts of the growing season, typically lagging T_s by 1-2 h. Diurnal variations in θ_v (at both measurement depths) did not change much over the growing season and showed no effect on diurnal variation in R_s. Mean daily R_s across different land uses were 3.9, 6.2, and 2.8 μmol CO_2 m^{-2} s^{-1} during the early, middle, and late parts of the growing season, respectively. In the middle of the growing season, the 9HP had the highest R_s (10.0 μmol CO_2 m^{-2} s^{-1}) followed by GRA (6.9 μmol CO_2 m^{-2} s^{-1}), AG (6.6 μmol CO_2 m^{-2} s^{-1}), 2HP (4.0 μmol CO_2 m^{-2} s^{-1}), and NAT (2.0 μmol CO_2 m^{-2} s^{-1}).

Heterotrophic and autotrophic respiration

Cumulative C released due to R_h (decomposition) was highest in NAT followed by AG > 9HP > 2HP > GRA whereas cumulative C released in R_a (autotrophic respiration) was highest in NAT followed by GRA > 9HP > AG > 2HP (Table 3-2). Mean R_h/R_s, across all sampling dates, was found to be highest at AG and NAT (R_h/R_s = 0.35 and 0.33, respectively), intermediate at 2HP and 9HP (R_h/R_s = 0.26 and 0.25, respectively), and lowest at GRA (R_h/R_s = 0.17). The average contribution of R_h to R_s (R_h/R_s) for all land uses over the growing period was 0.27.

Mean monthly R_h was not much different between NAT and AG (1.92 and 1.89 μmol CO_2 m^{-2} s^{-1}, respectively, Figure 3-6) but was higher than in the 9HP, GRA, and 2HP (1.22, 0.96, and 0.86 μmol CO_2 m^{-2} s^{-1}, respectively). Within each land use, R_h was significantly different between sampling dates (AG: *p-value* = <0.001, 2HP: *p-value* = 0.001, 9HP: *p-value* = <0.001, GRA: *p-value* = 0.003, and NAT: *p-value* = < 0.001). The general pattern for all land uses showed R_h to be higher only in June, although at the native aspen stand R_h was found to increase from 1.95 μmol CO_2 m^{-2} s^{-1} in June to 3.00 μmol CO_2 m^{-2} s^{-1} in July and then decline to a stable rate of 1.36 μmol CO_2 m^{-2} s^{-1} for the remainder of the study period. R_h was positively correlated with T_s (r = 0.79, *p-value* = 0.100, see Appendix 1).

At the AG site, R_a did not differ appreciably among the four sampling dates (*p-value* = 0.997) and hence appeared stable throughout the growing season whereas at the other land use sites, R_a varied significantly between sampling dates (2HP: *p-value* = < 0.001, 9HP: *p-value* = < 0.001, GRA: *p-value* = < 0.001, NAT: *p-value* = <0.001). At 2HP, 9HP and GRA, R_a was highest in July and lowest in September, whereas at NAT, R_a was statistically similar in June, July, and August (averaging 6.31 μmol CO_2 m^{-2} s^{-1}) but decreased significantly during September to 1.65 μmol CO_2 m^{-2} s^{-1} (*p-value* = <0.001). Averaged over all four sampling dates, R_a was significantly higher for NAT, GRA, and 9HP (4.50, 4.36, and 3.83 μmol CO_2 m^{-2} s^{-1}, respectively) than for 2HP and AG (3.06 and 2.69 μmol CO_2 m^{-2} s^{-1}). R_a was positively correlated with the fine root biomass (r = 0.44); but negatively correlated with θ_v (r = -0.81) and available NO_3^- (r = -0.92) and P (r = -0.94, see Appendix 1).

Discussion

Seasonal soil respiration under different land uses

Cumulative R_s for NAT, 2HP, and 9HP, during the growing seasons compared well with values of 513-802 g C m^{-2} reported by Tang et al. (2009) for recently disturbed, young, intermediate, and mature aspen stands in Wisconsin and Michigan. Differences in cumulative R_s between the land uses were also related to the differences in above-, belowground, and total ecosystem C stocks observed in this study. Several studies (Bond-Lamberty et al. 2004, Raich and Tufekcioglu 2000) have suggested that R_s tend to be greater in sites with greater detritus, which provide substrates for heterotrophic activity. I found that above- and belowground biomass C stocks (and presumably detritus production) increased nine years after hybrid poplar plantations were established on agricultural land, and resulted in a 10% increase in cumulative R_s (Table 3-2). In NAT, the presence of snags, coarse woody debris, and the LFH layer represents greater surface C accumulation. Even if the productivity of the agricultural field was relatively high (17 Mg C ha^{-1} in total plant biomass), only a small portion of the biomass was left in the field after harvest (3 Mg C ha^{-1} in the form of fine roots and stubble). The higher amount of detritus accumulated on the soil surface in NAT was reflected in a greater cumulative R_s, which was 55%, 82%, 42%, and 49% higher relative to the AG, 2HP, and 9HP, respectively.

At the NAT stand, I evaluated the relative contributions of the LFH layer and found that 75% of the efflux originated from this layer (Table 3-2). This agrees closely with Khomik et al. (2006) who used litter layer removal experiments to show that 62-85% of the R_s originated from the 12 to 20 cm thick LFH layer in a 74-year-old boreal mixedwood forest in central Ontario, Canada; and with Uchida et al. (1998) who examined the vertical distribution of R_s and found that 50-60% of R_s originated in the FH layer in an 80-year-old boreal forest dominated by black spruce with a few aspen trees in Saskatchewan, Canada.

Cumulative seasonal soil C efflux via R_s in AG (502 g C m^{-2}, average for 2006 and 2007), was approximately 2.5 times higher than in NAT without the LFH layer ($R_{NAT\text{-}LFH}$ = 194 g C m^{-2}) with a similar ratio for R_{sub} values (Table 3-2). The higher emissions of soil C under agriculture may be related to the land management because soil aeration and moisture become favorable for decomposition during cultivation. In addition, mixing of the soil also disrupts soil aggregates to further expose physically-protected C to decomposition (Six et al.

1998). As much as 20-50% of initial soil organic C stocks from native soils have been reported to be lost due to cultivation (Guo and Gifford 2002).

The proportion of respiration originating from below 11 cm depth, R_{sub}, was highest (72%) in the AG field, intermediate in 2HP (49%), GRA (47%), and 9HP (39%) sites, and lowest (17%) in NAT. Higher R_{sub} in AG and 2HP may be due to multiple tillage operations that occurred on the site for crop or plantation establishment and weed control which likely stimulated greater deeper root growth and turnover (Ellert and Janzen 1999). Site preparation in the 2HP involved deep (25-30 cm) and shallow (10 cm) tillage the previous year while weed control involved two passes of shallow (10 cm) tillage at the beginning and end of each growing season. Lower R_{sub} in NAT, on the other hand, may be attributed to unfavorable soil environment, viz., lower T_s (Figure 3-2), higher soil bulk density, and lower NO_3^- content (Chapter 2), as each of these factors would tend to reduce root and microbial activity and respiration and thus decrease R_{sub} (Burton et al. 1997) compared to systems with soil cultivation.

Vegetation cover can alter soil temperature and soil moisture conditions and thereby significantly influence soil respiration rates (Raich and Tufekcioglu 2000). I found that T_s in NAT was consistently lower compared to the AG. This is likely due to the shading of trees and the presence of the LFH layer. The LFH layer acts as an insulating mulch to reduce heat gain (from solar radiation) during daytime and loss during nighttime and hence moderate temperature conditions. My data clearly showed that R_s was highly correlated with T_s for both NAT and AG, both diurnally and seasonally (Table 3-2).

Diurnal soil respiration under different land uses

R_s originating from the mineral soil (excluding the LFH layer in NAT) showed similar diurnal variations for all land use types with early morning (generally before 09:00 h) minima and late afternoon (generally after 15:00 h) maxima. This pattern persisted throughout the early, middle and late parts of the growing season. The diurnal minimum and peak R_s not only corresponded with variations in T_s but also were related to different growth stages of vegetation present at each site. For example, these values in general were highest in the middle of the growing season. Fine root growth, especially for hybrid poplar plantations, has been shown to begin early in the spring, increasing exponentially until the middle of the growing season, and decreasing in the fall and winter (Coleman et al. 2000). It was also shown that the majority (65%) of the hybrid poplar fine roots appeared during the growing season and while half of fine roots survived after 149 days only 6% remained after 700 days (Coleman et al. 2000). In Chapter 2, I found that fine root biomass generally increased from June to October in the AG and 2HP sites and more than doubled by the end of the growing season, increased more than five-fold in 9HP but increased by only 39% in GRA. This growth pattern for fine roots may partially explain the increase in R_s from the early to middle parts of the growing season. In the fall, R_s decreased, presumably resulting from decreasing root biomass production and lower T_s. Comparing the two hybrid poplar plantations in mid-July (Figure 3-5), R_s was highest in 9HP compared to 2HP. This may be attributed to higher fine root biomass in the 9HP (614 g C m^{-2}) as compared to 2HP (230 g C m^{-2}).

Heterotrophic soil respiration

On average, R_h accounted for approximately 25% of R_s across all land uses during the growing season. Although not statistically different across land uses, AG had the highest R_h/R_s ratio (0.35), followed by NAT (0.33) > 2HP and 9HP (0.26 and 0.25, respectively) > GRA (0.17). According to literature reviews (Hanson et al. 2000, Subke et al. 2006), R_h and R_a generally account for approximately one half of R_s each. However, these averages mask considerable variation due to the vast diversity of ecosystems and potential biases of different partitioning techniques and time scales. The partitioning method (root exclusion) I used involved severing (but not removing) all the roots around the perimeter of collars in order to eliminate root respiration and inhibit future root growth. One limitation of this technique is that cessation of root activity within the collars would have eliminated transpiration losses resulting to possibly higher soil moisture, which may in turn, affect decomposition (Hanson et al. 2000) and potentially overestimate R_h (or underestimate R_a) in my calculations. Another limitation of this technique is that I took first measurements one month after the long collars were established. This may not have been enough time for R_h from dying roots to disappear and may cause R_h to be overestimated. Another complicating factor of this technique is the presence of roots below the root-exclusion zone i.e., below 30 cm depth; and the possibility of roots growing into the root-exclusion zone from the bottom (Hanson et al. 2000). I determined fine root distribution in the 0-30 cm soil profile during one of the sampling periods (data not shown) and found that fine root biomass decreased with depth (0-10 cm: 55%; 10-20 cm: 32%; 20-30 cm: 13%) in all land uses. However, I was not able to quantify whether and to what extent possible artifacts related to root biomass below the root-exclusion zone affected R_s. Therefore a more detailed investigation into the changes in soil conditions following root-exclusion is necessary to estimate proportions of R_h and R_a more accurately.

The quality of soil organic C (as determined by the natural abundance of ^{13}C) under different land uses could be related to its decomposability (Ehleringer et al. 2000) and its sensitivity to T_s (as determined by its Q_{10} values). A more enriched (less negative) ^{13}C value indicates a more stabilized (less labile) form of C. The more stable the C, the lower its lability and temperature sensitivity (Q_{10} value). In Chapter 2, I found decreasing ^{13}C abundance in the 0-10 cm mineral soil layer in the order of: NAT (δ^{13}C values of -26.24 ‰) > GRA (-26.77 ‰) > 9HP (-26.83 ‰) > 2HP (-27.37 ‰) > AG (-27.78 ‰). These results suggest that soil organic C in NAT had the highest proportion of the humified form compared to the other sites. Conversely, AG had the highest proportion of the labile C, with GRA, 2HP, and 9HP sites having intermediate levels of humified and labile C forms. In addition, I found that Q_{10} values were in the order of: AG > 2HP > 9HP > NAT > GRA (Table 3-1), which means that soil organic matter in the AG field is most sensitive to changes in T_s, intermediate under HPs, and least under the NAT and GRA systems. A comparison of cumulative C efflux in NAT and other land uses (Table 3-2) indicates that the majority of R_s in the former originated from labile C in the LFH layer. A more enriched form of ^{13}C and somewhat lower Q_{10} for mineral soil in NAT is indicative of stabilized C translocated from the upper layer. On the other hand, lower values of ^{13}C and higher values of Q_{10} under AG and 2HP suggest the presence of more labile C forms in these systems. This may be responsible for greater R_s in these ecosystems as fresh litter was mixed up in the upper 30 cm layer during tillage operations.

Autotrophic respiration

My estimates of R_a/R_s (65-83%) were slightly higher in comparison to other published studies involving trenched plots, viz., 37% in Saurette et al. (2008), 35-70% in Lavigne et al. (2003), and 27-71% in Lee et al. (2003). Lower estimates of R_a could have been due to the contribution from decomposition of severed roots (Ohashi et al. 2000), and lateral diffusion of CO_2 from the control to the root exclusion plot at its base (Jassal and Black 2006). In a review involving 54 sites, Bond-Lamberty et al. (2004) proposed a global relationship between R_s and R_a ($R_a^{0.5} = -7.57 + 0.93R_s^{0.5}$; $r^2 = 0.87$). My cumulative R_a was higher than the global relationship proposed by Bond-Lamberty et al. (2004), but fell within the 95% confidence interval of this global relationship.

The proportion of R_a to R_s decreased in the order of: NAT > GRA > 9HP > AG > 2HP (Table 3-2) and was related to fine root biomass ($r = 0.79$). Aside from fine root biomass, R_a is also related to nutrient availability, specifically NO_3^- ($r = -0.92$) and P ($r = -0.94$), and soil moisture content ($r = -0.81$). Based on the inverse relationship, it seems reasonable to hypothesize that fine root biomass was inversely related to NO_3^- and P availability on these sites. With limited supply of nutrients, plants tend to allocate more biomass to fine roots which in turn increases R_a. Availability of limiting nutrients has been suggested to be the major factor governing biomass allocation patterns in plants (Vogt et al. 1985).

Conclusions

Changes in land use led to differences in annual input of plant litter to the soil, and above- and belowground biomass. In addition, the change in land use altered the micro-environmental conditions of the soil, which is known to affect litter decomposition rates and ultimately the quality and quantity of soil organic matter. The conversion of a native forest to agricultural land resulted in the loss of the surface organic layer, which is the primary location of R_s in natural forest ecosystems in boreal regions. The results highlighted the importance of discerning R_s as a product of the production and consumption of organic matter that are inextricably linked to and influenced by a variety of controlling factors. Soil surface CO_2 flux was lower in the 2-yr-old hybrid poplar plantation resulting from lower fine root biomass (and thus lower R_a), although this was somewhat offset by slightly higher R_h due to higher T_s. Higher soil surface CO_2 flux in the 9-yr-old hybrid poplar plantation and grassland was due to greater fine root biomass that increased R_a as well as higher rates of litter inputs to support heterotrophs. It is thus important to consider the alterations in the relationship between R_s and environmental factors, brought about by land use change, when estimating variations in the C cycle and its response to future climate change.

Tables and Figures

Table 3-1. Analysis of the dependence of measured soil surface respiration (R_s, μmol CO_2 m^{-2} s^{-1}) on soil temperature (T_s, °C) using the equation $R_s = a \exp^{bT_s}$; and Q_{10} (unitless, temperature sensitivity parameter, relative change in R_s for a 10 °C change in T_s) across various land uses in Linaria, AB.

Land use	R_s		T_s		a	b	r^2	Q_{10}
	Min	Max	Min	Max				
AG	0.1	11.6	-0.4	19.3	0.80	0.10	0.89	2.70
2HP	0.2	6.2	-2.2	21.6	0.93	0.08	0.94	2.21
9HP	0.1	9.9	-2.5	24.2	1.29	0.07	0.89	2.11
GRA	0.2	7.3	-1.6	27.0	1.53	0.06	0.93	1.88
NAT	0.2	9.5	-0.7	25.8	2.24	0.07	0.88	1.93

Table 3-2. Cumulative seasonal (June 1-September 30) soil C efflux from the soil surface (R_s) and subsurface (R_{sub}) over two study years and the corresponding proportions originating from hetero- (R_h and R_h/R_s) and autotrophic (R_a) respirations in the first study year from various land uses in Linaria, AB. Values are cumulative estimates ± uncertainty at 95% confidence. $R_{NAT\text{-}LFH}$: respiration originating from the soil mineral under the forest floor in the NAT stand. n.d.: not determined.

Land use	Components of soil respiration (g C m^{-2})				
	R_s	R_{sub}	R_h	R_h / R_s	R_a
			2006		
AG	530 ± 26	359 ± 24	186 ± 35	0.35	344 ± 65
2HP	449 ± 20	228 ± 14	116 ± 22	0.26	333 ± 63
9HP	598 ± 33	232 ± 23	148 ± 22	0.25	450 ± 68
GRA	569 ± 32	257 ± 11	97 ± 10	0.17	473 ± 47
NAT	833 ± 41	149 ± 25	272 ± 44	0.33	560 ± 90
$R_{NAT\text{-}LFH}$	203 ± 32				
			2007		
AG	474 ± 22	357 ± 27		*n.d.*	
2HP	408 ± 19	192 ± 35		*n.d.*	
9HP	504 ± 28	201 ± 34		*n.d.*	
GRA	478 ± 22	229 ± 20		*n.d.*	
NAT	728 ± 38	123 ± 28		*n.d.*	
$R_{NAT\text{-}LFH}$	185 ± 42				

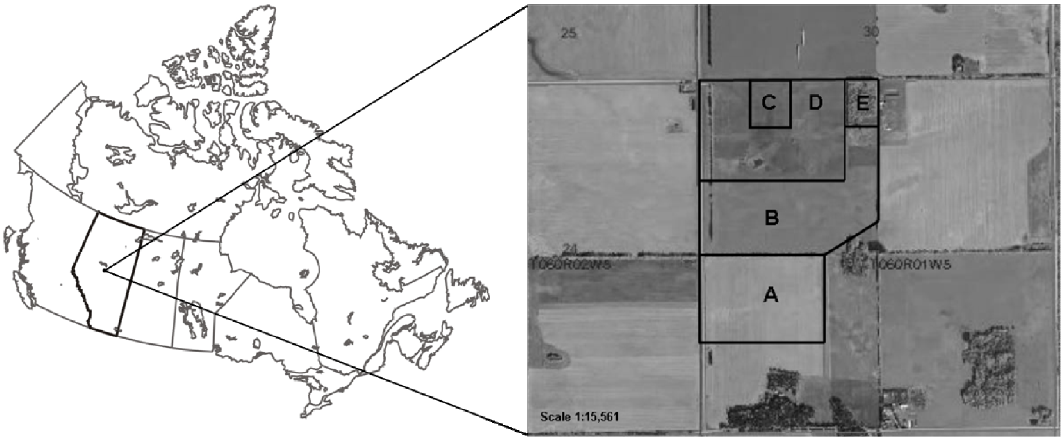

Figure 3-1. Location of land use study sites (A – agriculture; B – 2-yr-old hybrid poplar plantation, C – 9-yr-old hybrid poplar plantation; D – grassland; E – native aspen stand) in Linaria, Alberta, Canada. The source of the aerial photo is http://www.agriculture.alberta.ca. The use of the aerial photo by the author is done without any affiliation with or endorsement by the Government of Alberta. Reliance upon the author’s use of this material is at the risk of the end user.

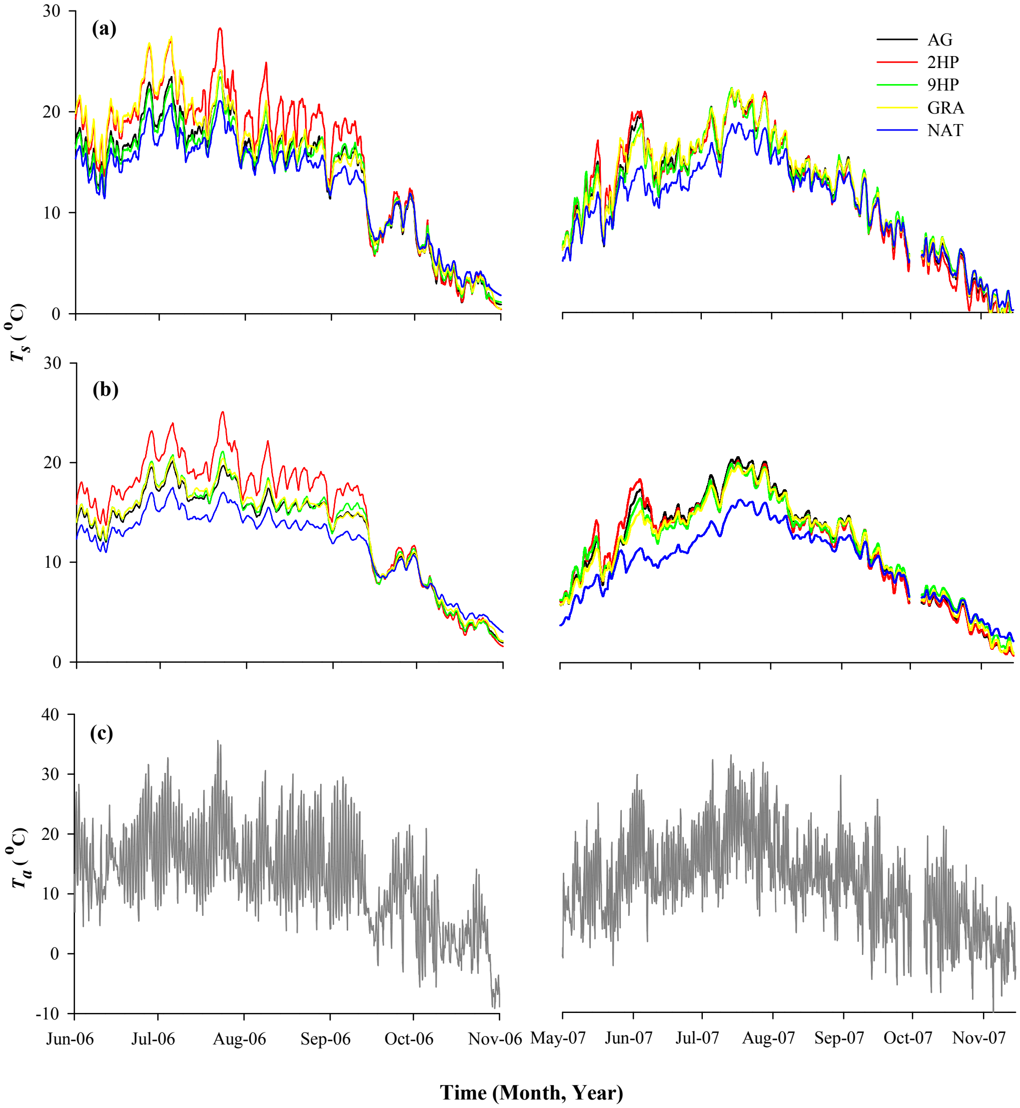

Figure 3- 2. Seasonal patterns of soil temperature (T_s) at (a) 2 cm and (b) 20 cm depths in various land uses; and (c) air temperature (T_a) during the two-year study period in Linaria, AB.

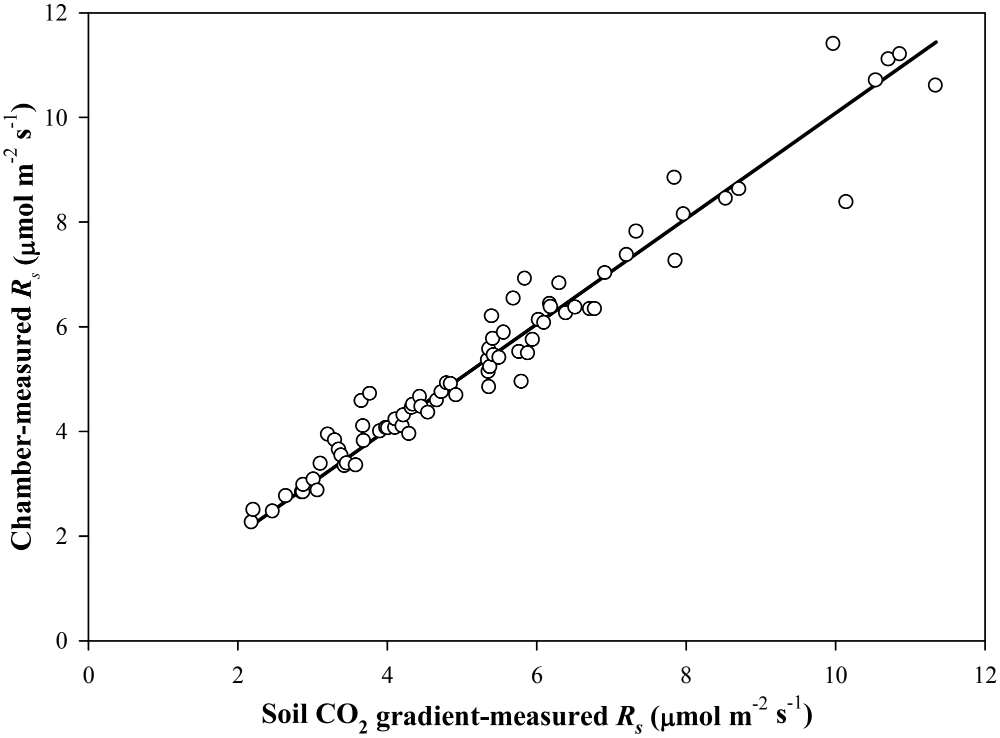

Figure 3- 3. Comparison of soil CO_2 effluxes (R_s) measured using the chamber and the soil CO_2 gradient techniques.

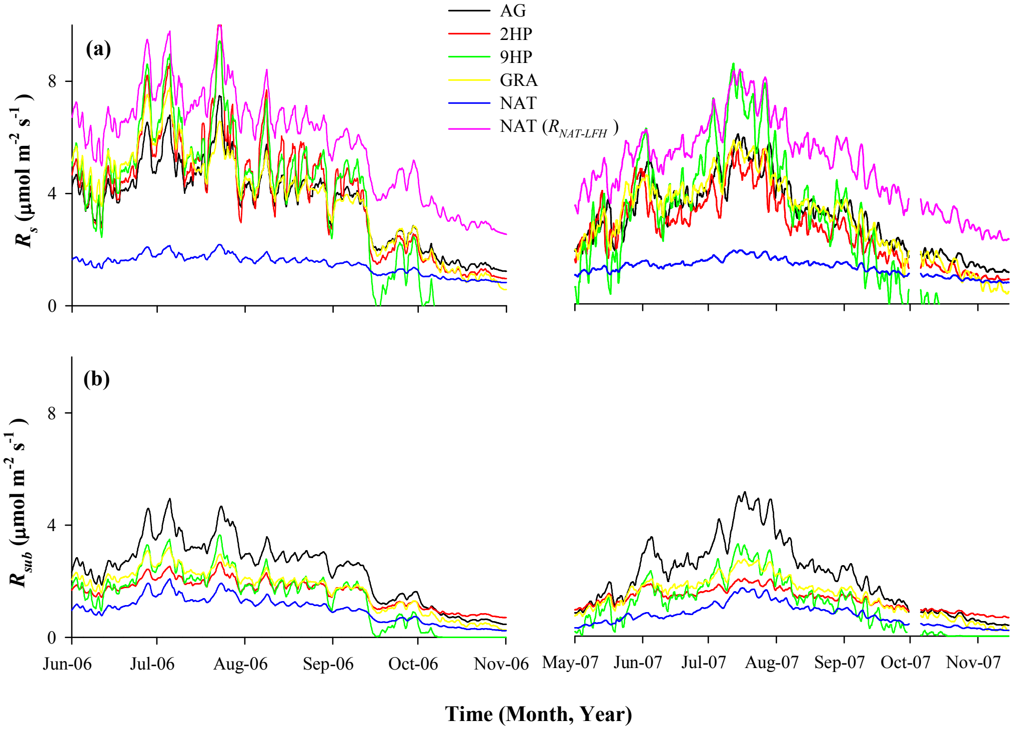

Figure 3-4. Seasonal patterns of soil respiration at the soil surface (R_s) and at 11 cm depth (R_{sub}) in various land uses; and at LFH layer in the native aspen stand ($R_{NAT-LFH}$) during the two-year study period in Linaria, AB.

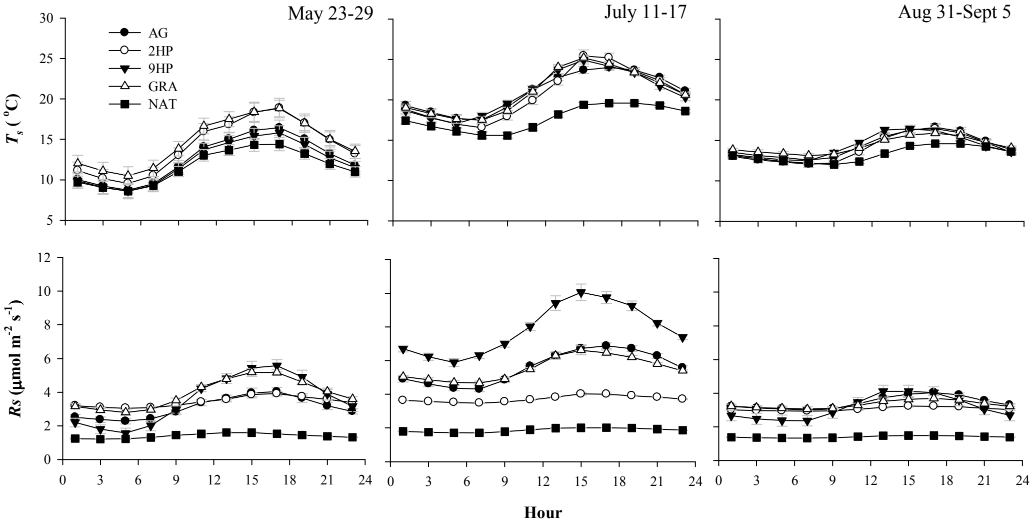

Figure 3-5. Mean diurnal patterns and their standard errors of soil temperature at the 2 cm depth and soil respiration during the early (May 23-29), middle (July 11-17), and late (August 31-September 5) parts of the growing season of 2007 across various land uses in Linaria, AB.

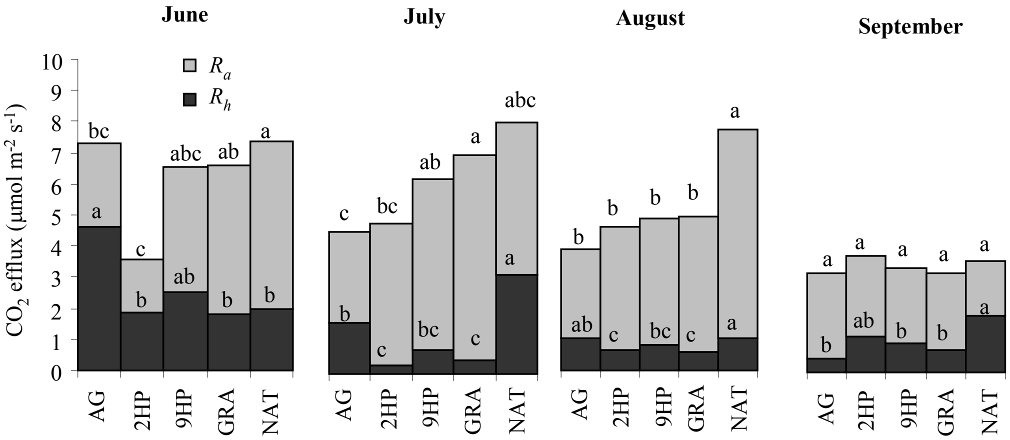

Figure 3-6. Mean monthly autotrophic (R_a) and heterotrophic (R_h) respiration in the 2006 growing season in various land uses in Linaria, AB. Different letters among land uses are significantly different at p-value < 0.05.

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Chapter 4. Mineralization potential and temperature sensitivity of soil organic carbon under different land uses

Introduction

Land use change alters the soil environment, the rates of biomass production, soil microbial activities and soil aggregate formation, and thus in turn affects carbon (C) cycling and storage in the soil. Changes in land use, along with the associated management practices, influence the quantity and quality of plant litter returning to the soil system and the biophysical properties of the soil. In the Prairie Provinces of Canada, forests of the southern boreal plains of Saskatchewan declined from 1.8 Mha in 1966 to 1.35 Mha in 1994, through an overall conversion of 24% of the boreal transition zone to agriculture (Hobbs and Theobald 2001). Fitzsimmons (2002) estimated that forest land had been converted to agriculture, industrial and urban uses at a rate of 1215 ha yr^{-1} along the southern boreal zone of Canada. Conversion of forest land to agricultural use reduces plant biomass and increases soil disturbance, decomposition rates and erosion. It is well documented (Balesdent et al. 1988, Guggenberger et al. 1995, Martin et al. 1990, Vitorello et al. 1989) that native soils (forested or grassland) lose C when converted to agricultural use. Globally, conversion from forests to pastures, permanent croplands and shifting cultivation has been estimated to cause the release of 123 Pg of C to the atmosphere since 1850 (Houghton 1995).

Agricultural practices such as cultivation disrupt soil aggregates and expose physically protected C. Depending on the nature and degree of mineral-organic complexes formed, C associated with different particle size fractions behaves differently in soil (Bonde et al. 1992, Christensen 1987, Tiessen and Stewart 1983). In a study by Balesdent et al. (1988), native prairie soils lost greater quantities of C in larger size fractions (>25 μm) than in fine size fractions (2-25 μm) during 27 years of annual cultivation. On the other hand, C associated with finer size fractions (<2 μm) was persistent under both cultivation and native prairie, suggesting that land use change mainly affected C associated with larger size fractions (>2μm). Higher mineralization rates of organic C in coarse fractions have been ascribed to physical disintegration of organic materials (Tiessen and Stewart 1983), presence of macroaggregates, high microbial activity (Christensen 1987, Christensen 2001) and a larger pool of mineralizable C (Gregorich et al. 1989). Carbon in this size fraction is described as uncomplexed or unprotected organic matter (OM) that is neither present at readily recognizable litter components (>2 mm) nor incorporated into primary mineral-organic complexes. It is a transitory pool between litter and mineral-associated C (Christensen 2001) and consists mainly of particulate OM and partially decomposed plant litter and animal residues (Guggenberger et al. 1995, Cheshire and Mundie 1981) and can include fungal hyphae, root fragments, spores and seeds (Gregorich and Janzen 1996).

Carbon associated with coarse size fractions is referred to as the *labile* C pool (Figure 4-1) that has slower turnover time than that of recently shed litter but faster than that of C associated with finer particle size fractions. It responds rapidly to changes in land use, vegetation type, climate, soil type, faunal activity, and could provide early indication of effects of soil management (Camberdella and Elliot 1992, Gregorich et al. 1994). Carbon associated with finer particle size fractions, referred to as the *passive* C pool (Figure 4-1), on the other hand, is held together by microbially processed products, root exudates, polyvalent cations and other strong organic polymers (Christensen 2001, Cheshire and Mundie 1981) and is characterized as stabilized or recalcitrant. Stabilization results from C either being physically protected in microaggregates or chemically associated with silt and clay (Six et al. 2002). Recalcitrance results from the inherent chemical property of the plant material *per se* (referred to as litter quality) or from condensation and complexation of decomposition residues that cause them to be more resistant to further decomposition. The complexation process decreases microbial access to C by pore-size exclusion that leads to reduced microbial activities and mineralization rates (Six et al. 2002, Bronick and Lal 2005, Jastrow and Miller 1997) and therefore promotes C accumulation in soils. Protection of C from further microbial decomposition in these fractions is essential to significantly lengthen the residence time of C in soils (Jastrow et al. 2007).

Another important factor affecting C decomposition and ultimately the potential capacity of soils to store C is temperature. Temperature, a major determinant of microbial processes, plays an important role in the release of C by soil heterotrophic respiration. It is well known that different plant species produce litter of varying decomposability. While decomposition rate of soil OM is highly dependent on temperature, this relationship may also be dependent on the type of land use (Raich and Tufekcioglu 2000). It has been shown that temperature sensitivity of C decomposition increases with increasing substrate complexity (Bosatta and Ågren 1999). However, others have reported contradictory results regarding temperature sensitivity of C (Fang et al. 2005, Giardina and Ryan 2000, Kirschbaum 1995). There have been many studies investigating temperature responses of C mineralization rates in different soils in both field and laboratory conditions (Anderson 1973, Edwards 1975, Moore 1986, Nadelhoffer et al. 1991, Pöhhacker and Zech 1995). Additional information can be gained by using soil fractions, provided that the obtained fractions can be related to their functional entities in soil (Christensen 1987). Organic C in different fractions because of differences in their physical and chemical properties will likely have different sensitivities to changes in temperature and thus affect the loss of C from the bulk soil.

In this paper, soils originated from the same parent material (Chapter 2) that were under a number of different land uses were collected: 1) to determine mineralization rates and mean residence time of C in soils under different land uses; 2) to determine the contribution of different particle size fractions to total mineralizable C from the bulk soil; and 3) to examine the effect of changes in temperature on the mineralization and turnover rates of C in bulk soils and particle size fractions from different land uses.

Materials and Methods

Soil sampling and particle size fractionation

Soils were collected from sites under four different land uses near Linaria (54°12’ N, 114°8’ W), approximately 25 km west of Westlock, in north central Alberta, Canada (Figure 4-2). Samples were obtained from (i) an agricultural field (AG), (ii) a 2-yr-old (in 2006) hybrid poplar (*Populus deltoides* x *Populus x petrowskyana* cv. Walker) plantation (2HP), (iii) a 9-yr-old (in 2006) hybrid poplar (Walker) plantation (9HP), (iv) a grassland (GRA), and (v) a native forest stand of trembling aspen (*Populus tremuloides* Michx., NAT). Details of the four different land use sites can be found in Chapter 2.

Mineral soil samples to 10 cm were collected on June 15, 2006 using a bulb planter (surface area of 27.3 cm^2). Twenty samples were randomly collected from each sampling plot, mixed, and composited in the field to yield approximately 2 kg of sample per plot. Samples were immediately transported to the laboratory, where they were air-dried, sieved to remove roots and material >2 mm, and stored in airtight containers at room temperature until further analysis.

Fractionation methods (physical or chemical or their combination) are employed to separate C into multiple pools representing different stages of decomposition that have distinct functional roles in stabilization mechanisms (Six et al. 2002, Cambardella and Elliot 1994, Golchin et al. 1994, Jastrow 1996). Physical fractionation of soil according to particle sizes was chosen in this study because it is considered less destructive and the results are anticipated to relate more to the structure and function of C *in situ* (Christensen 1992, Elliott and Cambardella 1991, Oades 1993). In this study, soil samples were fractionated without chemical pre-treatment using a wet- sieving technique (Christensen 2001, Six et al. 2002, Christensen 1992, Anderson et al. 1981, Bird et al. 2003, Preston et al. 2000) to separate the soil sequentially into four fractions: 250-2000 μm (coarse sand-size fraction), 53-250 μm (fine sand-size fraction), 2-53 μm (silt-size fraction), and <2 μm (clay-size fraction) (Figure 4-3). Briefly, air-dried soil (100

g) that has passed through a 2 mm sieve was weighed into 500 mL containers to which 150 mL distilled water was added and the sample was shaken on a flatbed shaker for one hour on medium speed to break up soil structure. The soil slurry was ultrasonically dispersed with a Fisher Sonic Dismembrator (model 300, Fisher Scientific, Pittsburgh PA) for two minutes at 60% of maximum power. Samples were then wet sieved through a 250 μm and 53 μm mesh. Material less than 53 μm, but larger than 2 μm, was further separated from the <2 μm portion by sedimentation (8 hr for every 10 cm of suspension) and decantation. The remaining solution was flocculated with 1 N KCl, with subsequent removal of KCl by dialysis (Fisherbrand® Nominal MWCO 12000-14000, Fisher Scientific, Pittsburgh PA) in distilled water until washing solution was free of salt to obtain the <2 μm soil fraction. Coarse sand-, fine sand-, and silt-size fractions were dried in a forced-air oven at 40 °C (Christensen 1992) for 72 hr while the clay-size fraction was freeze-dried (FreeZone 6 Liter with a Bulk Tray Dryer, Labconco Corp., Kansas City MO) for one week in order to minimize shrinkage and preserve soil porosity (Lawrence et al. 1979). All fractions were then re-weighed to calculate recovery and relative distribution.

Average recovery of bulk soil in fractionated samples was 98% across the different land uses with the relative distribution ranges of 6 to 13%, 28 to 33%, 46 to 57%, and 7 to 9% for the coarse sand, fine sand, silt, and clay size fractions, respectively. Particle size analysis previously performed on the same bulk soil samples (Table 4-1), however, determined that the soil samples contained more of the clay size fraction (26-31%) than was separated by fractionation. Thus, to account for the quantity of the clay size fraction, the silt- and clay-size fractions were combined to represent the *fine-size fraction* (<53 μm), and designated the sand-size fraction (53-250 μm) as the *medium-size fraction* and 250-2000 μm as the *coarse-size fraction*. Average recovery of samples from particle size fractionation was 99%. The fine size fraction accounted for 59% of the recovered mass, followed by the medium (30%) and coarse-size (10%) fractions. A second particle size analysis was carried out for each of the coarse- and medium-size fractions and fine-size fractions and it was determined that the coarse- and medium-size fractions contained an average of 66% sand-, 24% silt- and 10% clay-size particles while the fine-size fraction contained an average of 75% silt- and 25% clay-size particles (Table 4-1). This showed that during the fractionation procedure, ultrasonic dispersion and subsequent wet sieving of the soil samples did not fully separate mineral particles from aggregates.

The bulk soil and particle size fractions were finely ground in a ball mill (Mixer Mill MM200, Thomas Scientific, Swedesboro NJ) and analyzed for total organic C, total organic N, ^{13}C and ^{15}N abundance using a VG Optima continuous-flow stable isotope ratio mass spectrometer (VG Instruments, England) linked to a CN analyzer (NA-1500 series, Carlo Erba, Milan, Italy).

Incubation of bulk soils and particle size fractions

Bulk soil and particle size fractions from each plot were incubated for one year. Approximately 10 g of bulk soil or 2 g of each of the particle size fractions was placed into a 177 mL specimen container. To each of the particle size fractions, 8 g of purified quartz sand (40-100 mesh, Fisher Scientific, Pittsburgh PA) was added to improve aeration in these samples during incubation (Swanston et al. 2002). At the start of the incubation, the sand and soil mixture was adjusted to 60% field capacity (Paul et al. 2001, Robertson et al. 1999) by adding 4 mL of inoculum solution prepared by shaking 100 g of fresh soil from the respective plot with 1 L of distilled water (1:10 w:v soil:water) for 10 min and leaving the suspension to rest for 24 hr (Christensen 1987). Each soil sample contained in specimen containers was placed into a wide-mouth 1 L canning jar (Mason type) containing 20 mL of water to maintain humidity throughout the incubation (Paul et al. 2001). The lid of the canning jar was fitted with a rubber septum and sealed with silicon. A total of 180 such microcosms were prepared: 5 sites × 3 replications × 4 soil samples (bulk soil and 3 particle size fractions) × 3 temperatures to study C mineralization as described below.

Carbon mineralization

The microcosms were incubated in growth chambers at three temperatures: 7, 14, and 21 °C. The CO_2 concentrations in the canning jars were measured on days 0, 2, 4, 6, 9, 12, 21, 29, 35, 42, 56, 73, 92, 105, 119, 147, 175, 198, 231, 286, and 370. A 10 mL gas-tight syringe (SGE Gas Tight Plunger Syringe with PTFE Luer Lock, Fisher Scientific, Pittsburgh PA) was used to mix the headspace gas several times before 2 mL of air sample was drawn from the closed jar and immediately injected into a buffer volume of CO_2-free air flowing into an infrared gas analyzer (LI-6262 CO_2/H_2O Analyzer, LI-COR Biosciences, Lincoln NE). After each sampling, headspace was flushed with compressed air to allow the replenishment of O_2 and a uniform starting condition in air composition. Also, soil moisture content in the samples was adjusted to its initial value. After closing the lids tightly, immediate measurements were to record initial CO_2 concentration for the next incubation period.

Data analysis

The CO_2 concentration (ppm) in the headspace of the canning jars was calculated using calibration curves established between measured vs. actual (standard gases were 460 and 1200 ppm, respectively) CO_2 concentrations. Calibration was performed at the beginning and end of each series of 15 samplings. Total CO_2 evolved (μg CO_2) was calculated as:

$$([Final\ CO_2] - [Initial\ CO_2]) \times n \qquad (1)$$

where n is the quantity of gas in μmol in the incubation jar and is calculated using the ideal gas law:

$$n = \frac{PV}{RT} \qquad (2)$$

where P is pressure and is considered fixed at 101.3 kPa, V is head space volume calculated as the total volume of the incubation jar minus soil, water, inoculums, and specimen container volume, R is the universal gas constant (0.08205 L atm K^-

^{1} mol^{-1}), and T is the incubation temperature, in K (Robertson et al. 1999, Paré et al. 2006).

Various kinetic models (linear, first-order, two-pool first-order, and three-pool first-order) were tested to find out the type of linear or non-linear function that best describe the data. It was intended that a common equation for all the soil and temperature treatments be obtained, so as to enable comparison of curve fitting parameters while achieving the smallest possible residuals. A simple linear regression model was finally chosen because it allowed convergence of all parameters and gave the least sum of squares of residuals for the greatest majority of temperature and soil treatments. In order to exclude the initial flush of CO_2 caused by disturbance from setting up the experiment, measurements up to day 30 were not considered in the regression analysis using:

$$C_m = a + kt \tag{3}$$

where C_m is the cumulative mineralized C (mg C mg^{-1} of initial total organic C, C_i), t is the incubation period (d), k is the mineralization rate constant (mg C mg^{-1} C_i d^{-1}), and a is a constant. The turnover of C was quantified (Six and Jastrow 2002) by its mean residence time (MRT), calculated as:

$$MRT = \frac{1}{k} \tag{4}$$

Further, regression analysis was performed to determine the exponential relationships between cumulative mineralized C (C_m) and incubation temperatures, T_i:

$$C_m = a \exp^{bT_i} \tag{5}$$

where coefficient a is the C mineralization rate at temperature zero (i.e., basal rate), and coefficient b is the sensitivity of C_m to temperature (T_i). For bulk soil, temperature sensitivity, Q_{10}, was calculated from:

$$Q_{10} = \exp^{10b} \tag{6}$$

where Q_{10} is the relative increase in C_m for every 10 °C increase in soil temperature, T_s. For the particle size fractions, temperature sensitivity, γk, was calculated from:

$$\gamma k = \frac{k_{T1}}{k_{T2}} \qquad (7)$$

Where γk is the relative increase in C_m when soil temperature increased from T_1 to T_2. In this case, T_1 was equal to 7°C while T_2 was either 14 or 21°C.

Statistical analyses were performed using univariate, general linear model (GLM), linear (REG) and non-linear (NLIN) regression procedures in SAS version 9.1 (SAS Institute Inc., 2004). The assumption of normality was assessed with a Kolmogorov-Smirnov test (data not shown). Analysis of variance (ANOVA) coupled with Tukey's Studentized Range Test was performed to evaluate the main effects of land use, temperature, particle size fractions and their interactions on the parameters analyzed. Interaction effects were elucidated using the *pdiff* option in SAS. In all comparisons, $\propto = 0.05$ was used as the significance level.

Results

Carbon, nitrogen contents and isotopic compositions of bulk soil and particle size fractions

Soil organic C and N contents in the bulk soil ranged from 28 to 78 and 2 to 7 Mg ha^{-1}, respectively, with most of them held within the fine fraction (Table 4-1). Soil C and N in the coarse fraction increased by more than an order of magnitude nine years after hybrid poplar plantations were established on agricultural soil. Within the medium fraction, C and N were similar across land uses except that they were significantly lower in NAT. In the fine fraction, C and N were high under AG, 9HP and 2HP, intermediate in GRA, and low in NAT. The C:N ratio decreased with decreasing size of the particle fraction for all land uses. The only significant effect of land use on soil organic C content was between the 9HP and the NAT, while total N content was lower in NAT than in the other land uses.

The $\delta^{13}C$ in the bulk soil varied between -27.0 and -26.2‰, and in the particle size fractions between -27.8 and -25.6‰ (Table 4-1). Within particle size fractions, ^{13}C enrichment progressively increased with decreasing particle size (with a few exceptions). Eighty years after agriculture replaced the native aspen stand, the coarse and fine size fractions have been depleted with ^{13}C by 3.8 and 3.3%, respectively. In contrast, two years after agriculture was converted to hybrid poplar plantation, the coarse and fine fractions became 1.5 and 1.9%, respectively, more enriched in ^{13}C. Seven years later (from 2HP), the same size fractions became 3.4 and 1.0%, respectively, more enriched in ^{13}C. Likewise, nine years after the agricultural land was converted to grassland the same particle size fractions became 3.6 and 0.7%, respectively, more enriched in ^{13}C. The $\delta^{15}N$ were all positive, with $\delta^{15}N$ in litter substantially lower than in bulk soil and particle size fractions. Soil $\delta^{15}N$ values also tended to increase with decreasing particle size.

Effects of land use and temperature on bulk soil C mineralization

Mineralization rates of bulk soils during the year-long incubation period averaged across all land uses were in the following ranges: 0.01-0.30, 0.02-0.63 and 0.04-1.42 μg C mg C_i d^{-1} for incubations at 7, 14 and 21°C, respectively (Figure 4-4). The peak of mineralization (maximum CO_2 evolution) occurred between days 6 and 9 at 7°C, between days 2 and 6 at 14 °C and between days 2 and 4 at 21 °C, after which, mineralization gradually decreased to nearly the same rates for the rest of the incubation. A second peak of CO_2 flush was observed between days 29 and 42 for the 7 °C and between days 9 and 21 for the 14 °C incubation, before finally deceasing to more stable rates. No second CO_2 peak was observed for the 21°C incubation. Across the land uses, rates of mineralization increased with incubation temperature. The AG, 2HP, 9HP and GRA treatments had similar mineralization rates while NAT had lower rates especially at incubation temperatures of 14 and 21 °C.

The simple linear regression model (Equation 3) had R^2 values ranging between 0.86 and 0.95, 0.94 and 0.96 and 0.93 and 0.97 for incubation temperatures of 7, 14 and 21 °C, respectively. Cumulative mineralized C, normalized by the initial amount of C present (mg C mg^{-1} C_i), presented a larger initial release of CO_2 followed by a smaller linear increase throughout the remaining 370-day incubation period (Figure 4-5).

With respect to initial C present in the bulk soil (286 to 815 mg C_i) across land uses, only 2 to 5% (12 to 19 mg C mg^{-1} C_i) was mineralized (C_m) over a year-long incubation period at different temperatures. Proportions of mineralized C (C_m/C_i) were statistically similar (average of 4.4%) between AG, 2HP, 9HP and GRA while it was significantly lower in NAT (2.3%). Similarly, mineralization rates (k) of the bulk soil under NAT ($k = 0.06$) was significantly slower than under the other four land uses (average $k = 0.12$). Consequently, MRT of C across four of the land uses had average residence time of 30 yrs that was shorter than NAT with a residence time of 51 yrs.

Regardless of land use, C_m and C_m/C_i increased with increasing incubation temperatures (Table 4-2). Proportions of mineralized C generally doubled with the increase of temperature from 7 to 14 °C and increased by 1.5 times from 14 to 21 °C; which translated to an average Q_{10} value of 2.2 across all sites. The C_m in AG was most sensitive to temperature increase with Q_{10} value of 2.6, followed by 2HP (2.4) > 9HP (2.2) > GRA (2.2) > NAT (1.7). Rates of mineralization (k values) also increased 2 and 1.5 times when incubation temperature increased from 7 to 14 °C and from 14 to 21°C, respectively (Table 4-2). The MRTs decreased with increasing temperatures, with average decrease of 26 yrs when temperature increased from 7 to 14 °C and of 9 yrs when temperature increased from 14 to 21 °C.

Effects of land use and temperature on C mineralization in particle size fractions

The C_m/C_i were significantly affected by land use, particle size and temperature ($p = 0.028$, Table 4-4). The C_m/C_i averaged across land uses and temperatures decreased in order of coarse (34%) > medium (5%) > fine (3%) size fractions. As a specific example, in AG C_m/C_i averaged across temperatures was 67, 6 and 3% in coarse, medium and fine size fractions, respectively; while in NAT it was 12, 2 and 1% for the same size fractions, respectively. Further, across all land uses and particle size fractions, C_m/C_i increased from 7 to 14 to 21% with increasing incubation temperatures of 7, 14 and 21 °C, respectively (Tables 4-3 and 4-4).

Mineralization rates generally decreased with decreasing degree of disturbance (Figure 4-8), i.e., in the order of: AG (the range across particle size fractions was 0.23-1.33 $\mu g\ C\ mg^{-1}\ C_i\ d^{-1}$) > 2HP (0.16-0.52) > 9HP (0.06-0.33) ≈ GRA (0.07-0.34) > NAT (0.03-0.18). For all land uses, mineralization rates increased with increasing temperature. Average mineralization rate at 7 °C was 0.24 $\mu g\ C\ mg^{-1}\ C_i\ d^{-1}$; which then doubled at 14 °C (0.47 $\mu g\ C\ mg^{-1}\ C_i\ d^{-1}$) and more than tripled at 21 °C (0.79 $\mu g\ C\ mg^{-1}\ C_i\ d^{-1}$). The peak of mineralization in the particle size fractions for all land uses occurred between days 2 and 12 (at 7 °C), between days 2 and 6 (at 14 °C) and between days 2 and 4 (at 21 °C), after which, mineralization gradually decreased to nearly the same rates and remained so for the rest of the incubation. In some fractions, a second peak of CO_2 flush was observed before finally deceased to more stable rates.

Mineralization rates averaged across particle size fractions and incubation temperatures decreased with decreasing levels of disturbance (Table 4-5), i.e., AG ($k = 0.77$) > 2HP ($k = 0.52$) > 9HP ($k = 0.28$) ≈ GRA ($k = 0.26$) > NAT ($k = 0.14$). Averaged across land uses and incubation temperatures, k values decreased with decreasing particle size in the order of coarse ($k = 0.97$) > medium ($k = 0.14$) > fine ($k = 0.07$). Averaged across land uses and particle size, k values increased with increasing temperature i.e., 0.18 at 7 to 0.39 at 14 and 0.61 at 21 °C.

Temperature sensitivity (γk) increased within the coarse size fraction when AG was changed to 9HP or GRA. For the coarse fraction in AG, k increased from 1.16 to 1.98 (a 1.71 fold increase or $\gamma k = 1.71$) and from 1.16 to 3.04 ($\gamma k = 2.62$) as temperature increased from 7 to 14 and to 21 °C, respectively. Under 9HP or GRA, although values of k within the coarse fraction were four times lower compared to AG, the temperature sensitivity was greater with $\gamma k = 2.35$ and 3.77 in 9HP or $\gamma k = 1.96$ and 3.72 in GRA when temperatures increased from 7 to 14 °C and from 7 to 21 °C, respectively. In the medium size fraction, temperature sensitivities across land uses were found to be similar except under 9HP with $\gamma k = 7$ when temperatures increased from 7 to 21 °C.

Mean residence time was significantly affected by land use and particle size fraction ($p = 0.0035$, Table 4-4). Averaged across particle size fractions and incubation temperatures, MRT was shorter in 2HP (26), GRA (29) and AG (30 yrs), intermediate in 9HP (33 yrs) and longest in NAT (74 yrs). Across land uses and temperatures, MRT was shortest in the coarse size fraction (6 yrs), intermediate in the medium size fraction (39 yrs) and longest in the fine size fraction (69 yrs). Although not statistically significant (Table 4-4), MRT of C generally decreased with increasing temperature and followed more or less an exponential form of decay (Figure 4-9). Residence time of C averaged 72 yrs under 7 °C, 26 yrs under 14 °C and 17 yrs under 21 °C.

Comparison of C mineralization in bulk soil vs. particle size fractions

Mineralization rates of the bulk soil and the sum of soil fractions were similar only from day 2-4 of the year-long incubation period (Figure 4-4). By day 4 (under 7 °C) or day 2 (under 14 and 21 °C), however, the difference between the bulk and sum of fractions started to increase i.e., mineralized C from the bulk soil samples were lower than the sum of fractions for all cases; and the differences between them increased with increasing temperature. By the end of the incubation period, bulk soil mineralized half the amount mineralized in the fractions (0.02

versus 0.04 mg C mg^{-1} C_i) at 7 °C, three times lower at both 14 °C (0.03 versus 0.10 mg C mg^{-1} C_i) and 21 °C (0.05 versus 0.14 mg C mg^{-1} C_i).

Discussion

Carbon mineralization in bulk soil

The proportions of mineralized C from the bulk soils under different land uses (2 to 5%) was comparable to mineralized C (3-13%) in a 282-day incubation of mineral soil samples from various forest stands and soil types in the northeastern boreal region (Côté et al. 2000). Lower proportions of mineralized C and mineralization rates with higher MRT under NAT may be due to the exclusion of the LFH layer, where recent inputs of C undergo much of the microbial processes before they translocate into the mineral soil. Once C reaches the mineral soil and forms mineral-organic complexes, it becomes more stable and less prone to further decomposition. This is reflected by the lower decomposability (indicated by $\delta^{13}C$ and $\delta^{15}N$ values) of substrates in NAT compared to other land uses (Table 4-1). The natural abundances of ^{13}C and ^{15}N correspond with different stages of organic matter oxidation due to various mechanisms including microbial discrimination against ^{13}C and ^{15}N during litter decomposition (Bird et al. 2003, Bird et al. 2002, Bol et al. 1999, Ehleringer et al. 2000, Nadelhoffer and Fry 1988). Enrichment of ^{13}C indicates increasing degree of decomposition of mineral-associated organic matter during the stabilization of C (Baisden et al. 2002). Lower proportions of mineralized C, slower mineralization rates, and higher MRT under NAT therefore indicates that soil organic C under NAT is more stable and resistant to mineralization and decomposition compared to soil organic C under AG, 2HP, 9HP and GRA.

When NAT was converted to AG, proportion of mineralized C, mineralization rate and turnover of C significantly increased. This may be due to litter from crop residues and roots annually added and incorporated into the system by cultivation. The $\delta^{13}C$ value in the bulk soil in AG was more negative

(-27.0‰) than in NAT (-26.2‰, Table 4-1) and reflected values that were closer to plant litter (-29.35‰, Chapter 2) implying that soil organic C in AG contained more labile organic matter compared to NAT. Mixing by cultivation causes the breakdown of soil structure that enhances oxidation of physically and chemically protected C inside aggregates (Six et al. 2000, Six et al. 2000). The combination of constant additions and mechanical disintegration of new organic matter every year resulted in higher C losses from AG. Conversion from agriculture to either hybrid poplars or grassland, on the other hand, did not lower the proportion of mineralized C, mineralization rate or increase the residence time (Table 4-2). However, there was some indication that litter decomposability in 2HP and 9HP decreased, as evidenced by the enrichment of ^{13}C and ^{15}N in the bulk soil (Table 4-1). Conversion of AG to HP or GRA started reversing the effect of cultivation on organic C pool. Although these changes maybe slow to occur, lower rates of mineralization are of significance for soil management.

The Q_{10} values (1.7-2.6) found in the bulk soil under laboratory conditions were consistent with the range of values (1.9-2.7) obtained under field conditions (Chapter 3). These values were also similar to values reported at the ecosystem level (Janssens and Pilegaard 2003, Raich and Schlesinger 1992, Rey and Jarvis 2006) and previous studies on soil decomposition (Fang et al. 2005, Kirschbaum 1995, Dalias et al. 2001). The proportions of mineralized C, mineralization rates and turnover of C in the bulk soil, for all land uses, responded positively to soil temperature, reflecting that temperature sensitivity of soil organic C decreases as temperature increases (Janssens and Pilegaard 2003, Lloyd and Taylor 1994, Price and Sowers 2004). Davidson et al. (2006), MacDonald et al. (1995), Monson et al. (2006) and Zogg et al. (1997) suggested that this effect may be due to one or a combination of factors: 1) a change in microbial community composition, 2) change in biochemical composition of the C fraction being mineralized, 3) change in the transport processes (diffusivity), or 4) lower microbial efficiency at higher temperatures.

Carbon mineralization in particle size fractions

Organic C was largely held by the fine size fraction, followed by the medium size fraction, and least by the coarse size fraction across land uses, which was consistent with patterns reported by Christensen (1987, 1985), Christensen and Sorensen (1985), Amelung et al. (1998), Bonde et al. (1992) and Gregorich et al. (1988). In contrast, the proportions of mineralized C, mineralization rates and turnover of C were highest in the coarse size fraction followed by the medium and the fine size fractions (Tables 4-3 and 4-4), again consistent with the literature (Gregorich et al. 1989). Tiessen and Stewart (1983) compared particle size fractionated soils originating from grassland soils that have been cultivated between 4 and 90 years under native prairie in Blain Lake, SK and found that C associated with the sand size fraction (>50 μm) depleted rapidly within 4 years of cultivation (43% of the initial C lost). The higher proportion of mineralized C observed from the coarse size fraction indicates the higher mineralizability of the organic C associated with this fraction (Hassink 1995, Parfitt and Salt 2001) as this fraction compose mostly of particulate OM and partially decomposed plant litter (Guggenberger et al. 1995, Cheshire and Mundie 1981).

It was found that the fine size fraction had the lowest proportions of mineralized C, had the lowest mineralization rates and had the longest mean residence times (Tables 4-3 and 4-5), similar to Hassink and Whitmore (1987) and Christensen (1987). It has been found in other studies that silt-associated C was often found to be more stable and resistant to microbial attack than that of clay (Gregorich et al. 1989, Parfitt and Salt 2001). Parfitt and Salt (2001) proposed that substrates adsorbed on the external surfaces of the clay-size fraction are readily accessible to microbial attack, whereas, in the silt-size fraction, substrates may be trapped in small pores and become physically protected from microbes.

Of the particle size fractions studied, it was the coarse size fraction that was more sensitive to land conversion compared to the medium and fine size fractions (Tables 4-3, 4-4 and 4-5). The decrease in mineralized C due to the establishment of hybrid poplars or grassland on previously cultivated land corresponded not only to the increase in stocks and MRT of C within the coarse size fraction but also to the enrichment of ^{13}C (Table 4-1). This suggests that establishment of plantations over agricultural soil wherein soil disturbance or mixing by cultivation is stopped, leads to a faster build-up of organic matter within the coarse size fraction and enhancement of more stable forms of organic matter.

The temperature sensitivity (γk) of soil organic C in this study was found to be higher in the coarse and medium size fractions when hybrid poplar plantations were established on agricultural land. With plantation establishment, litter returned to the system becomes rich in lignin. According to Christensen et al. (2000), lignin is composed of aromatic polymers with higher molecular weight. Soil OM rich in lignin is more resistant to decomposition (White 2006) and has been recently found to be more responsive to temperature change compared to labile OM. For example, Fierrer et al. (2005), in a 53-day incubation study found that more recalcitrant litter types, litter in advanced stages of decomposition, or specific C compounds of lower lability were most sensitive to increased temperature. Similarly, Leifeld and Fuhrer (2005) physically separated soil OM components (bulk soil, size fractions and hydrolysis residues) and incubated them separately under 5, 15, 25 and 35 °C. They found that the fractions representing the more stabilized C pool and that contained older C produced more CO_2 and were more responsive to increased temperature. These results support the hypothesis proposed by Bosatta and Ågren (1999) that the enzymatic reactions required to metabolize structurally complex C substrates involve higher activation energies than reactions metabolizing simpler or more labile C substrates. With increase in temperature, there is higher activation energy available resulting in increased rate of decomposition of these complex C. While the data suggest that the change in C substrates with plantation establishment on agricultural land results in lower degree of mineralization and turnover of C, it was also found that with land conversion, soil C mineralization also became more responsive to change in temperature. This may have important implications in the future ecosystem models that, even with small increases in temperature, it may prompt large releases of C as soils from these types of land use respond to global change.

Carbon mineralized from bulk soil vs. particle size fractions

At the end of the year-long incubation period, the sum mineralized C from the fractions were 2 to 3 times higher than the bulk soil (Figure 4-6). Similar results have been obtained by Zhang (2005) and Mueller and Koegel-Knabner (2007). Mueller and Koegel-Knabner (2007) also found that the main source of C (70%) was the clay size fraction. In this study, however, the main source of C (79%) in the recombined fractions was originating from the coarse size fractions (Table 4-3). In the coarse size fractions, the proportion of mineralized C was significantly greater in AG, 2HP, 9HP and GRA than in NAT (Table 4-3) which is explained by the quality of litter inputs in these systems. Higher proportions of mineralized C observed from the recombined fractions may have been caused by the disruption of aggregates during ultrasonic dispersion and physical size fractionation that reduced physical protection to organic C especially in the coarse size fraction (Christensen 1987, Christensen and Sorensen 1985). The position of C within aggregates provides longer-term storage for C because organic C within soil aggregates has lower rates of decomposition than those located outside. The disruption of aggregates exposes organic C and makes them readily available for microbial decomposition. The difference between the magnitude of C loss from the recombined fractions and bulk soils may indicate the amount of C released during fractionation.

Influence of land use and land use change on long term soil C storage

There are many factors and processes that influence the direction and rate of soil C storage with land use and land use change, including: 1) rates of C input into the soil, 2) the degree of soil disturbance, 3) physical protection of C either through aggregation or organo-mineral complexes, 4) change in litter quality, and

5) translocation rates of new organic matter from surface layer to mineral soil. Under AG, grain harvest is removed from the system while stubble residues and majority of belowground biomass (roots) are mechanically mixed and incorporated in the surface layer of the soil each year during cultivation, which increases the active C pool leading to greater mineralizability of bulk soil. Under HP, GRA and NAT, aboveground inputs (litterfall) are left on the ground to be decomposed and transferred into the mineral soil while belowground inputs (roots) enter the mineral soil directly. Thus conversion of native soils to agriculture causes the decrease in C storage while, conversion of agriculture land to woody vegetation leads to the increase in C storage over time.

Land use determines litter composition that largely controls soil organic C quality and its decomposability. In the native aspen stand, litter is woody in nature that may be composed of aromatic polymers with high molecular weight. The molecular structure of wood makes it resistant to microbial decay resulting in lower degree of mineralization and turnover of C in soil. Conversion of native forest into agriculture changes the quality of plant litter, which is not woody in nature, and thus results in a higher degree of mineralization, with least amount of C storage. Conversion of agriculture to grass or hybrid poplar plantations, on the other hand, produce roots and plant litter which are woody in nature that could significantly decrease the mineralization potential of soil organic C leading to the build-up and stability of C in soil.

In this study, soil organic C was separated into three size fractions with varying degree of mineralizability. The coarse size fraction constituted the labile C pool and mineralized quickly in soil. The labile pool was found to be the most affected by land use change as observed in the change in mineralization patterns. The mineralization pattern of this pool is in large part due to the dynamic nature of this pool that has an MRT of a few years. The dynamics of this pool was strongly influenced when the native aspen stand was converted to agriculture or when agriculture was converted into hybrid poplar plantations or grassland. However, majority of C was found in the medium and fine size fractions and were determined to be more stable and less sensitive to land use change than C found in the coarse size fraction, and MRT was in the order of decades.

Conclusions

The sensitivity of soil organic C to temperature change is critical to the global C balance and will determine whether soils under different land uses will respond with positive feedback to climate change. Litter quality and its temperature sensitivity play major roles in mineralization of soil C. At the ecosystem level, higher woody litter means lower degrees of mineralization, turnover, and cycling of C. Higher degrees of C cycling would have to be accompanied with higher biomass production to maintain ecosystem C balance. Establishment of hybrid poplar plantations over agriculture land have enhanced the overall cycling of C over nine years. Greater biomass productivity and less soil disturbance will likely lead to higher quality and faster build-up of organic C into the soil over the life cycle of these plantations.

Tables and Figures

Table 4-1. Particle size distribution, carbon and nitrogen contents, C:N ratios and isotopic values in the bulk soil and its particle size fractions originating from 0-10 cm depth under different land use systems in Linaria, AB, Canada.

Soil	Land use	Sand (%)	Silt (%)	Clay (%)	C (Mg ha^{-1})	N (Mg ha^{-1})	C:N	δ^{13}C (‰)	δ^{15}N (‰)
Bulk	AG	30	41	29	75.4 (2.3) ab	6.5 (0.3) a	12 (0.4) a	-27.0	7.0
	2HP	30	43	27	64.3 (2.3) ab	5.7 (0.1) a	11 (0.1) a	-26.9	7.0
	9HP	33	41	26	78.2 (8.8) a	7.0 (0.9) a	11 (0.1) a	-26.5	8.5
	GRA	36	38	26	56.7 (3.4) ab	5.0 (0.2) a	11 (0.2) a	-26.4	8.5
	NAT	35	34	31	27.6 (8.8) b	2.1 (0.5) b	13 (1.1) a	-26.2	4.9
Coarse	AG	81	8	12	4.3 (0.6) C b	0.4 (0.05) C b	19 (1.2) A a	-27.8	4.6
	2HP	73	15	12	4.7 (0.2) C ab	0.4 (0.02) C b	17 (0.5) A ab	-27.4	3.7
	9HP	63	23	15	9.8 (1.9) C a	0.9 (0.18) C a	15 (0.2) A ab	-26.8	8.5
	GRA	70	20	10	6.9 (1.1) C ab	0.6 (0.09) C ab	13 (1.4) A b	-26.8	5.0
	NAT	63	32	5	3.4 (1.2) A b	0.2 (0.07) AB b	19 (1.1) A a	-26.8	6.6
Medium	AG	62	31	7	21.0 (0.7) B a	1.8 (0.1) B a	13 (0.6) B a	-26.4	4.4
	2HP	63	29	8	19.0 (1.3) B a	1.7 (0.1) B a	12 (0.4) B a	-27.0	4.5
	9HP	61	32	7	24.0 (1.9) B a	2.1 (0.2) B a	12 (0.5) B a	-26.9	5.1
	GRA	61	31	8	18.0 (0.8) B a	1.6 (0.16) B a	13 (0.3) A a	-26.9	5.5
	NAT	61	20	18	9.2 (3.2) A b	0.7 (0.2) AB b	13 (0.5) B a	-26.3	7.9
Fine	AG		74	26	50.2 (2.9) A a	4.4 (0.3) A a	9 (0.6) C a	-26.4	6.3
	2HP		76	24	40.5 (1.2) A ab	3.6 (0.1) A ab	9 (0.5) C a	-25.9	7.0
	9HP		76	24	44.5 (5.0) A ab	4.0 (0.5) A ab	9 (0.2) C a	-26.1	6.4
	GRA		75	25	31.8 (2.1) A b	2.8 (0.2) A b	9 (0.3) A a	-26.2	7.0
	NAT		73	27	15.0 (4.5) A c	1.1 (0.3) A c	9 (0.3) C a	-25.6	6.9

Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP- 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand. Values (±SE) with different upper- and lowercase letters are significantly different between particle size fractions for each land use and between land use systems for each particle size fraction, respectively, at $p < 0.05$.

Table 4-2. Initial C (C_i), mineralized C (amounts, C_m, and proportions, C_m/C_i), mineralization rates (k) and mean residence times (MRT) of bulk soil under various land uses incubated at different temperatures over a 370-day period.

Land use	Temperature (°C)	C_i (mg)	C_m (mg)	C_m/C_i (%)	k (µg C mg^{-1} C$_i$ d^{-1})	MRT (years)
AG	7	370.1 (7.6) B a	7.0 (1.1) CD c	1.9 (0.3) AB c	0.05 (0.01) AB c	56 (8) AB a
	14	365.4 (8.0) B a	16.4 (1.3) A b	4.5 (0.3) A b	0.12 (0.01) A b	23 (2) B b
	21	368.1 (10.3) B a	25.8 (4.2) A a	7.1 (1.4) A a	0.19 (0.04) A a	15 (3) B b
2HP	7	287.4 (17.3) C a	5.6 (1.1) D c	1.9 (0.3) AB c	0.05 (0.01) AB c	55 (7) AB a
	14	288.6 (16.8) C a	13.5 (1.4) A b	4.7 (0.6) A b	0.12 (0.01) A b	22 (3) B b
	21	283.4 (13.2) C a	17.8 (1.1) A a	6.3 (0.6) A a	0.17 (0.01) A a	16 (2) B b
9HP	7	391.6 (17.3) B a	8.9 (0.3) B c	2.3 (0.1) A c	0.06 (0.00) A c	45 (2) B a
	14	390.2 (18.2) B a	19.0 (3.0) A b	4.8 (0.6) A b	0.13 (0.02) A b	22 (3) B b
	21	389.1 (19.3) B a	26.6 (3.0) A a	6.8 (0.5) A a	0.18 (0.01) A a	15 (1) B c
GRA	7	380.3 (59.8) B a	7.9 (0.5) BC c	2.2 (0.4) A c	0.06 (0.01) A c	50 (10) B a
	14	377.4 (58.0) B a	15.4 (2.0) A b	4.1 (0.1) A b	0.11 (0.00) A b	25 (1) B b
	21	377.4 (60.8) B a	24.3 (2.0) A a	6.6 (0.8) A a	0.18 (0.02) A a	16 (2) B b
NAT	7	810.9 (16.1) A a	12.6 (0.5) A b	1.6 (0.0) B b	0.04 0.00 B b	65 (2) A a
	14	816.1 (22.6) A a	17.3 (3.8) A b	2.1 (0.4) B b	0.06 (0.01) B b	52 (8) A b
	21	818.8 (25.2) A a	26.7 (6.6) A a	3.2 (0.7) B a	0.09 (0.02) B a	36 (11) A c

Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP- 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand. Values (±SE) with different upper- and lowercase letters are significantly different between land uses for each temperature and between incubation temperatures for each land use, respectively at $p < 0.05$.

Table 4-3. The distribution of soil organic C in particle size fractions relative to bulk soil, initial C, and mineralized C (amounts, C_m, and proportions, C_m/C_i) in various land uses incubated at different temperatures over a 370-day period.

Land use	Size fraction	Relative C distribution (%)	C_i (mg)	C_m (mg)			C_m/C_i (%)		
				7 °C	14 °C	21 °C	7 °C	14 °C	21 °C
AG	Coarse	6 (0)	21.0 (1.5)	8.3 (1.1)	13.7 (3.2)	19.7 (0.9)	43 (12)	64 (19)	94 (4)
	Medium	28 (1)	102.9 (4.4)	2.1 (0.2)	9.2 (2.1)	8.0 (1.5)	2 (0)	9 (1)	8 (2)
	Fine	67 (1)	244.0 (2.9)	2.1 (0.2)	7.3 (2.1)	10.0 (1.9)	1 (0)	3 (1)	4 (1)
2HP	Coarse	7 (0)	21.0 (0.6)	3.9 (0.4)	9.8 (2.0)	16.0 (0.6)	19 (2)	46 (8)	77 (2)
	Medium	29 (0)	85.0 (3.7)	2.1 (0.5)	8.1 (1.7)	9.3 (0.5)	2 (0)	9 (1)	11 (1)
	Fine	63 (1)	180.5 (4.1)	1.9 (0.2)	5.5 (0.4)	7.8 (0.8)	1 (0)	3 (0)	5 (0)
9HP	Coarse	12 (0)	48.4 (3.0)	4.6 (0.6)	11.3 (2.7)	17.9 (4.0)	10 (2)	23 (3)	37 (5)
	Medium	31 (1)	120.0 (1.8)	1.6 (0.0)	5.4 (1.2)	12.1 (3.2)	1 (0)	5 (1)	10 (3)
	Fine	57 0	221.9 (5.2)	2.6 (0.2)	5.7 (0.6)	7.6 (2.0)	1 (0)	3 (0)	3 (1)
GRA	Coarse	13 (1)	49.9 (7.5)	3.7 (0.4)	8.2 (1.2)	14.0 (2.4)	9 (3)	18 (4)	34 (13)
	Medium	32 (1)	120.0 (7.0)	1.9 (0.2)	7.0 (1.7)	7.3 (0.9)	2 (0)	6 (1)	6 (1)
	Fine	55 (1)	208.5 (15.6)	2.8 (0.2)	5.7 (0.3)	13.7 (1.8)	2 (0)	3 (1)	7 (1)
NAT	Coarse	12 (0)	98.7 (3.6)	6.4 (0.2)	11.9 (2.0)	17.6 (2.3)	6 (0)	12 (3)	18 (1)
	Medium	33 (1)	267.6 (8.3)	2.0 (0.5)	6.1 (1.3)	9.4 (2.1)	1 (0)	2 (1)	3 (0)
	Fine	55 (1)	449.0 (2.6)	1.8 (0.2)	4.8 (0.1)	10.0 (1.5)	0 (0)	1 (0)	2 (0)

Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP- 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand. Values in parenthesis are ± standard errors.

Table 4-4. Summary of ANOVA results (*p-values*) for the effects of land use, particle size fraction and temperature on the distribution of C relative to the bulk soil, initial C (C_i), mineralized C (amounts, C_m, and proportions, C_m/C_i), mineralization rates (k) and mean residence times (MRT) within 0-10 cm soil depth from Linaria, Alberta.

Land use	Relative C distribution	C_i	C_m	C_m/C_i	k	MRT
Source						
Land use (L)	<0.001	*n.s.*	0.043	<0.001	<0.001	*n.s.*
Temperature (T)	*n.s.*	*n.s.*	<0.001	<0.001	<0.001	*n.s.*
L × T	*n.s.*	*n.s.*	*n.s.*	0.004	0.001	*n.s.*
Particle size (P)	<0.001	<0.001	<.001	<0.001	<0.001	0.029
L × P	<0.001	0.002	*n.s.*	<0.001	<0.001	0.004
P × T	*n.s.*	*n.s.*	0.049	<0.001	<0.001	*n.s.*
L × P × T	*n.s.*	*n.s.*	*n.s.*	0.028	0.014	*n.s.*

Table 4-5. The mineralization rates (*k*) and mean residence times (MRT) of mineralized C from various land uses and particle size fractions incubated under different temperatures over 370 days.

Land use	Size fraction	*k* (μg C mg^{-1} C$_i$ d^{-1})			MRT (years)		
		7 °C	14 °C	21 °C	7 °C	14 °C	21 °C
AG	Coarse	1.16 (0.34)	1.98 (0.76)	3.04 (0.60)	3 (1)	2 (1)	1 0
	Medium	0.06 (0.01)	0.24 (0.03)	0.22 (0.06)	52 (9)	12 (1)	14 (3)
	Fine	0.02 (0.00)	0.08 (0.02)	0.11 (0.02)	119 (9)	38 (8)	27 (4)
2HP	Coarse	0.51 (0.06)	1.23 (0.21)	2.09 (0.06)	6 (1)	2 (0)	1 0
	Medium	0.07 (0.01)	0.25 (0.04)	0.30 (0.01)	44 (7)	11 (2)	9 (0)
	Fine	0.03 (0.00)	0.08 (0.01)	0.12 (0.01)	98 (10)	34 (3)	24 (3)
9HP	Coarse	0.26 (0.04)	0.61 (0.06)	0.98 (0.15)	11 (1)	5 (0)	3 (1)
	Medium	0.04 (0.00)	0.12 (0.03)	0.28 (0.08)	77 (3)	25 (7)	12 (5)
	Fine	0.03 (0.00)	0.07 (0.01)	0.09 (0.02)	87 (11)	41 (6)	35 (11)
GRA	Coarse	0.25 (0.08)	0.49 (0.10)	0.93 (0.35)	15 (6)	6 (1)	4 (1)
	Medium	0.04 (0.01)	0.16 (0.04)	0.17 (0.03)	67 (14)	20 (6)	17 (4)
	Fine	0.04 (0.01)	0.08 (0.02)	0.19 (0.03)	80 (20)	37 (8)	15 (2)
NAT	Coarse	0.18 (0.01)	0.33 (0.08)	0.48 (0.04)	16 (1)	9 (2)	6 (1)
	Medium	0.02 (0.01)	0.06 (0.01)	0.09 (0.01)	150 (36)	50 (13)	31 (5)
	Fine	0.01 (0.00)	0.03 (0.00)	0.06 (0.01)	260 (26)	94 (1)	48 (6)

Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP- 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand. Values in parenthesis are standard errors.

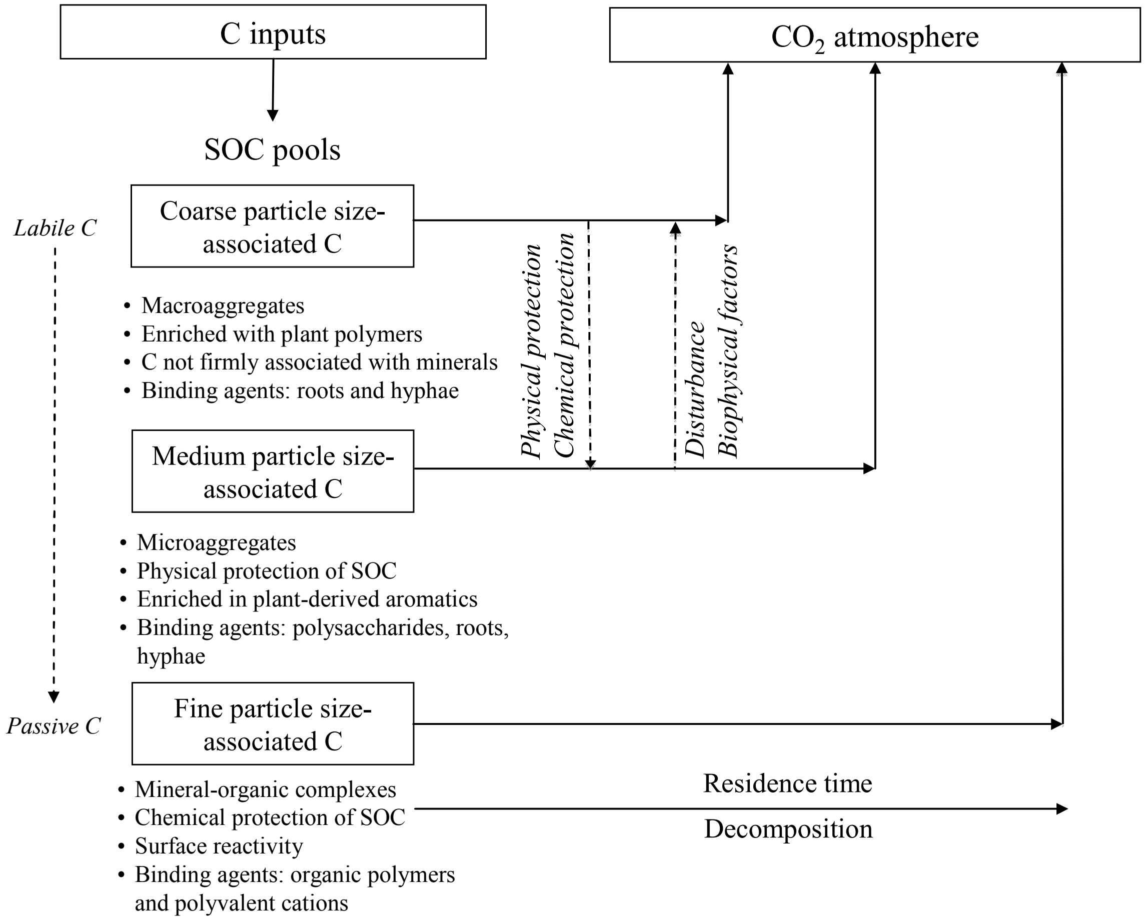

Figure 4-1. Soil carbon pool flow chart.

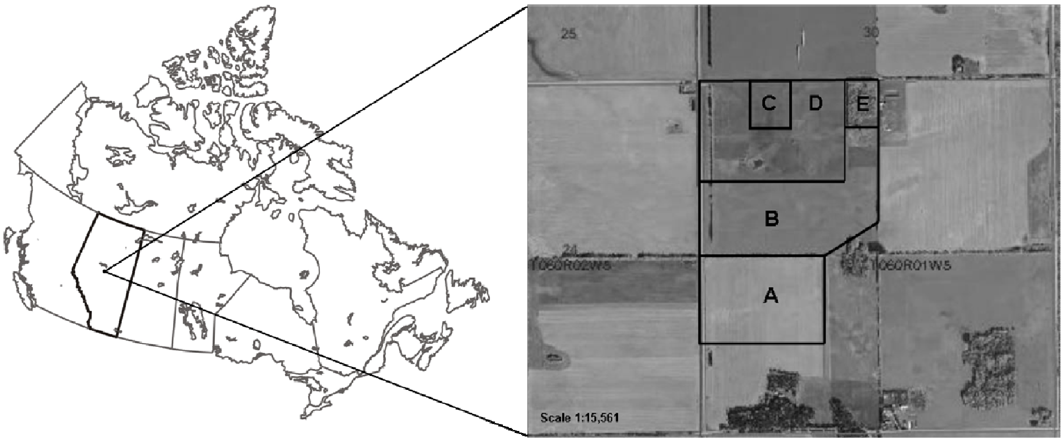

Figure 4-2. Location of land use study sites (A – agriculture; B – 2-yr-old hybrid poplar plantation, C – 9-yr-old hybrid poplar plantation; D – grassland; E – native aspen stand) in Linaria, Alberta, Canada. The source of the aerial photo is http://www.agriculture.alberta.ca. The use of the aerial photo by the author is done without any affiliation with or endorsement by the Government of Alberta. Reliance upon the author's use of this material is at the risk of the end user.

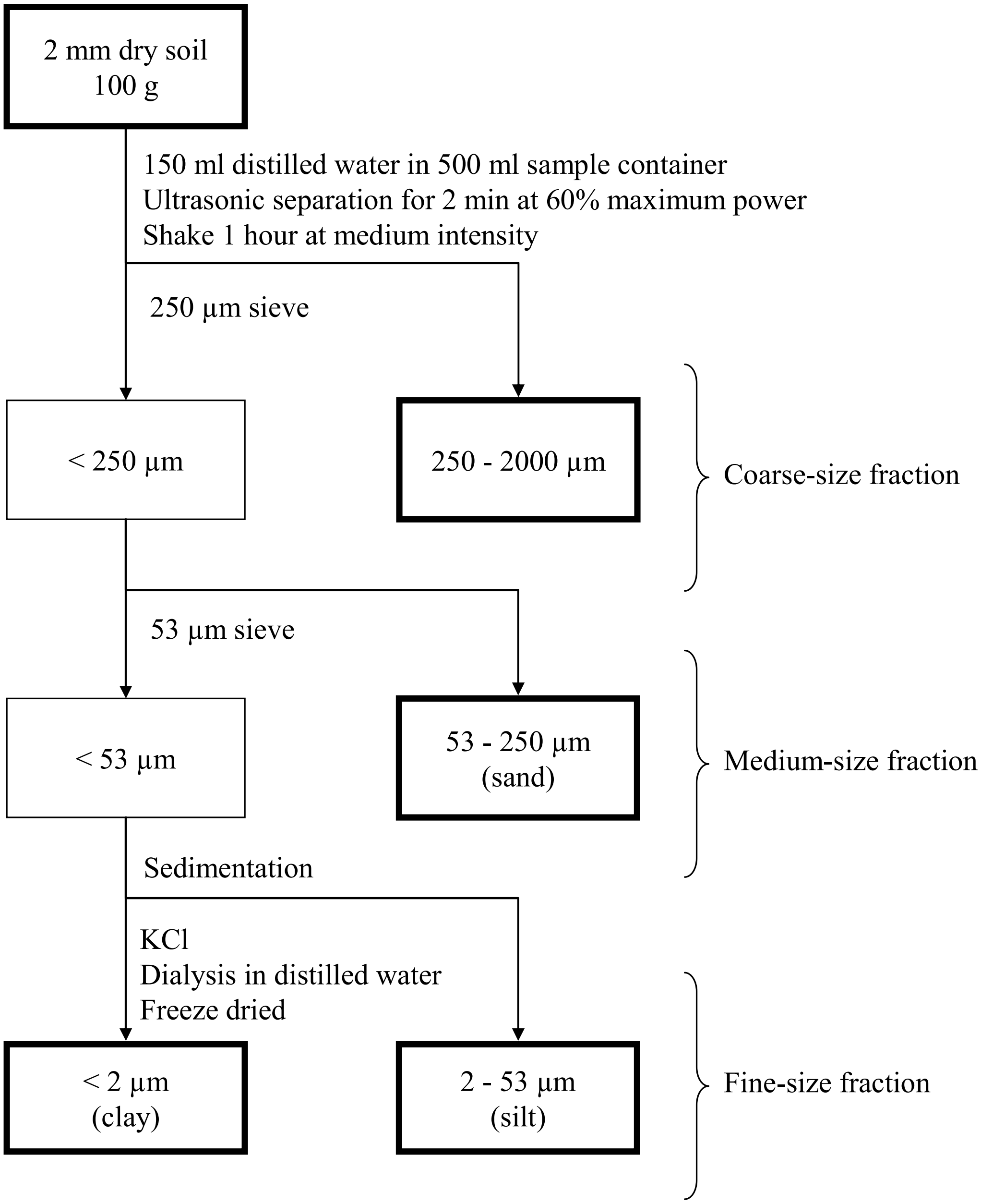

Figure 4-3. Diagram of the particle size fractionation scheme used in the study. The fractionation sequence results in the following three size sieve fractions: coarse size fraction, 250-2000 µm; medium size fraction, 53-250 µm; and fine size fraction (combined silt-, 2-53 µm, and clay-size fractions, <2 µm).

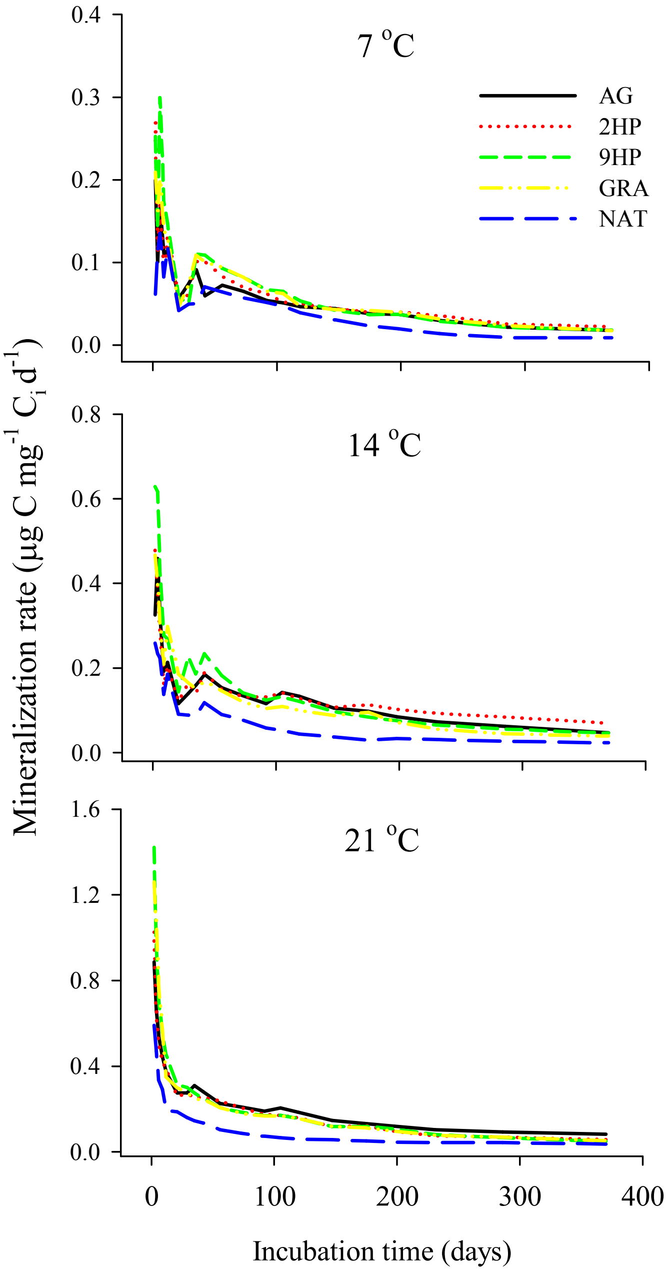

Figure 4-4. Mineralization rates of bulk soils incubated at different temperatures over a 370-day period. Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP – 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand. Note scales increase with increasing temperature.

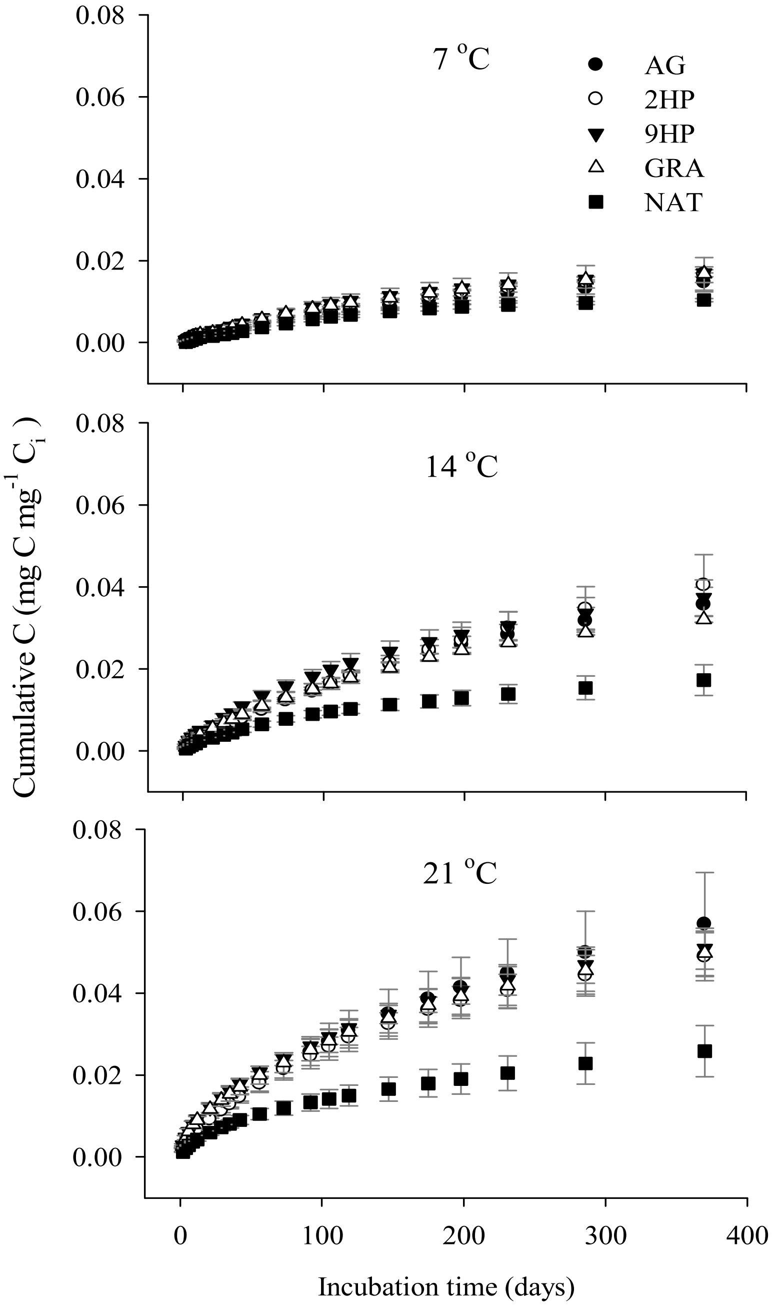

Figure 4-5. Cumulative mineralized C from bulk soil taken from various land uses incubated over a 370-day period at three different temperatures. Bars are standard errors. Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP – 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand.

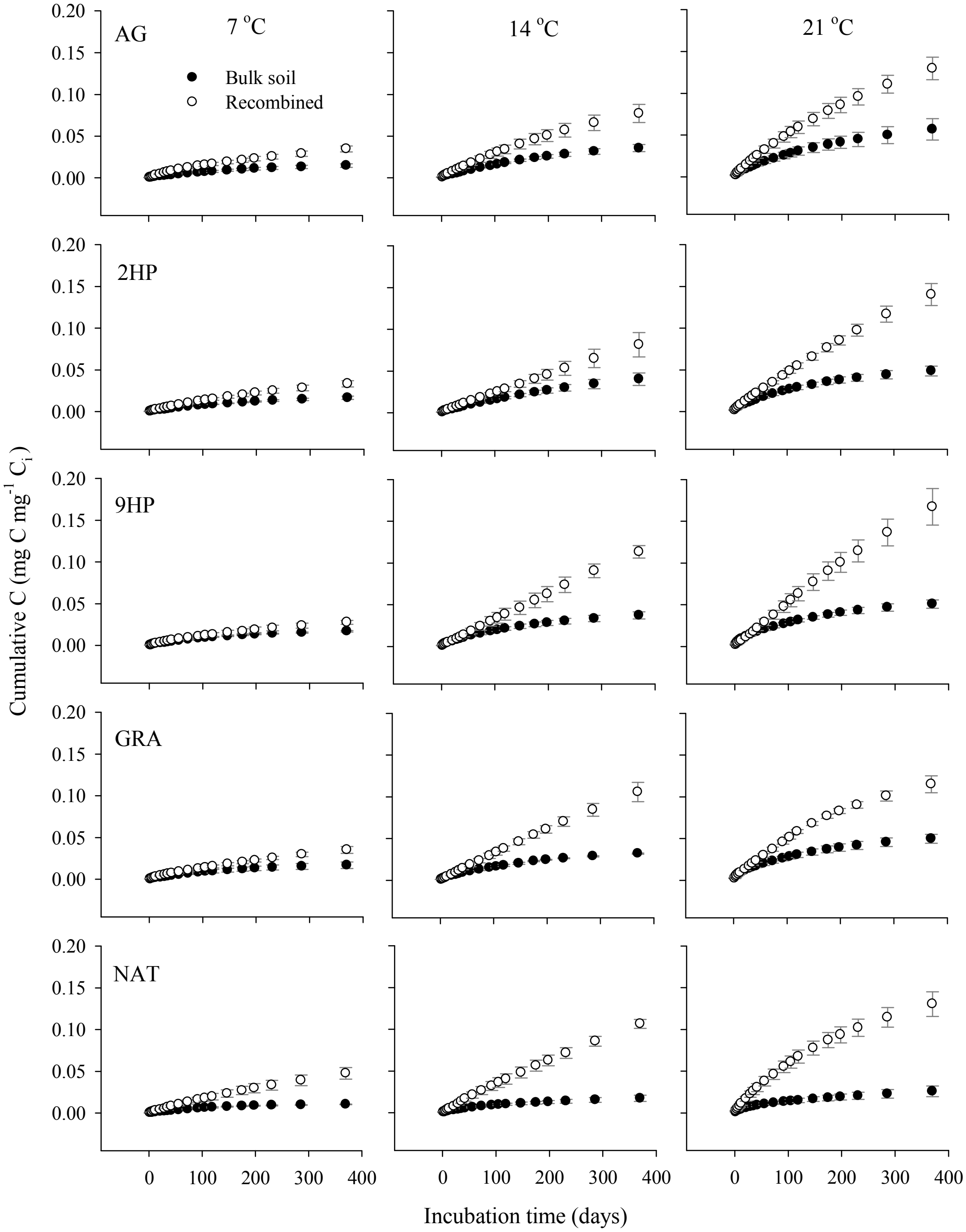

Figure 4-6. Cumulative CO_2 evolution from bulk and mathematically recombined (summed) particle size-fractions taken from various land uses and incubated under different temperatures over a 370-day period. Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP – 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand.

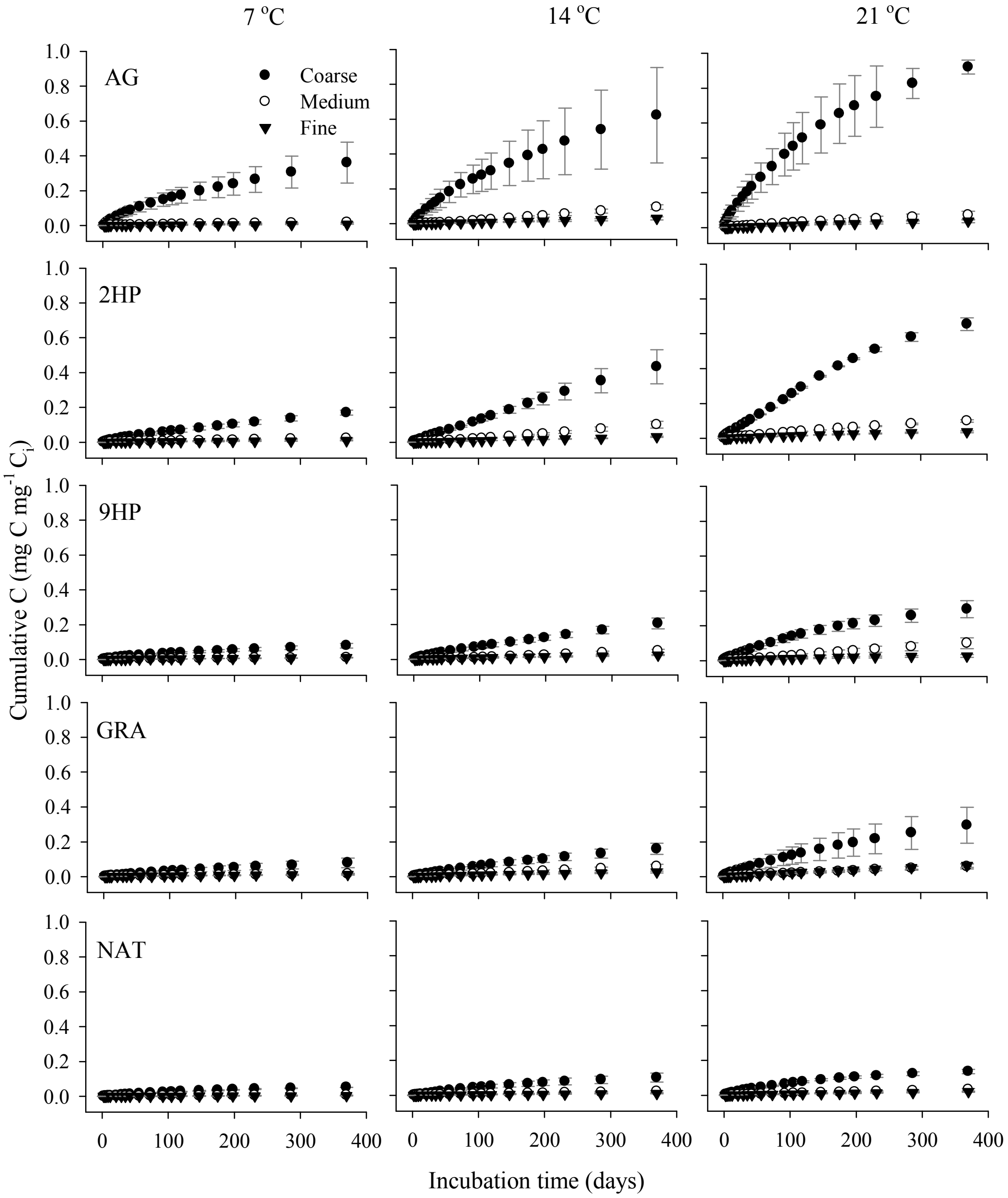

Figure 4- 7. Cumulative CO_2 evolution from particle size-fractions isolated from bulk soils taken from various land uses and incubated under different temperatures over a 370-day period. Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP – 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand.

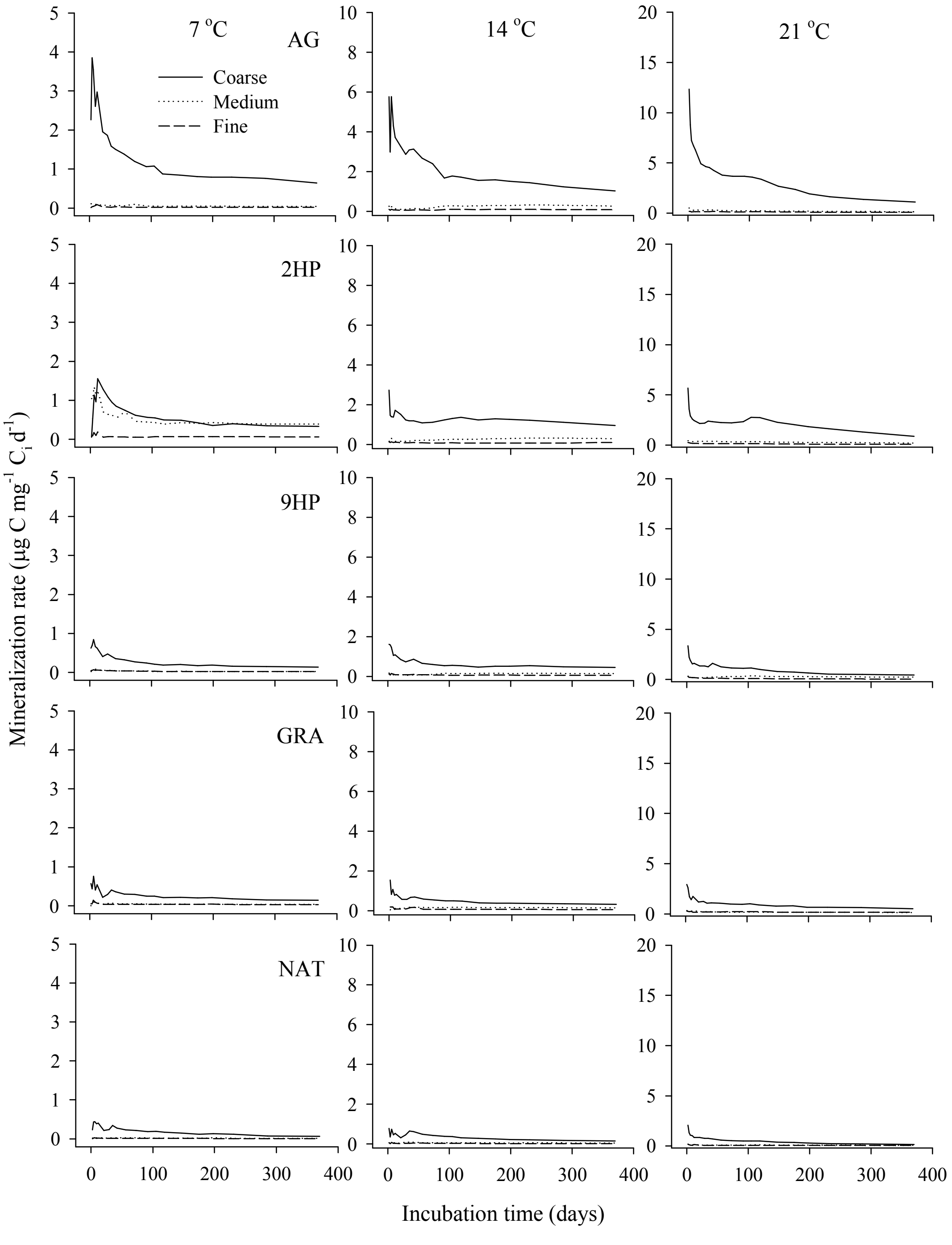

Figure 4-8. Mineralization rates of particle size fractions incubated at different temperatures over a 370-day period. Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP – 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand. Note scales increase with increasing temperature.

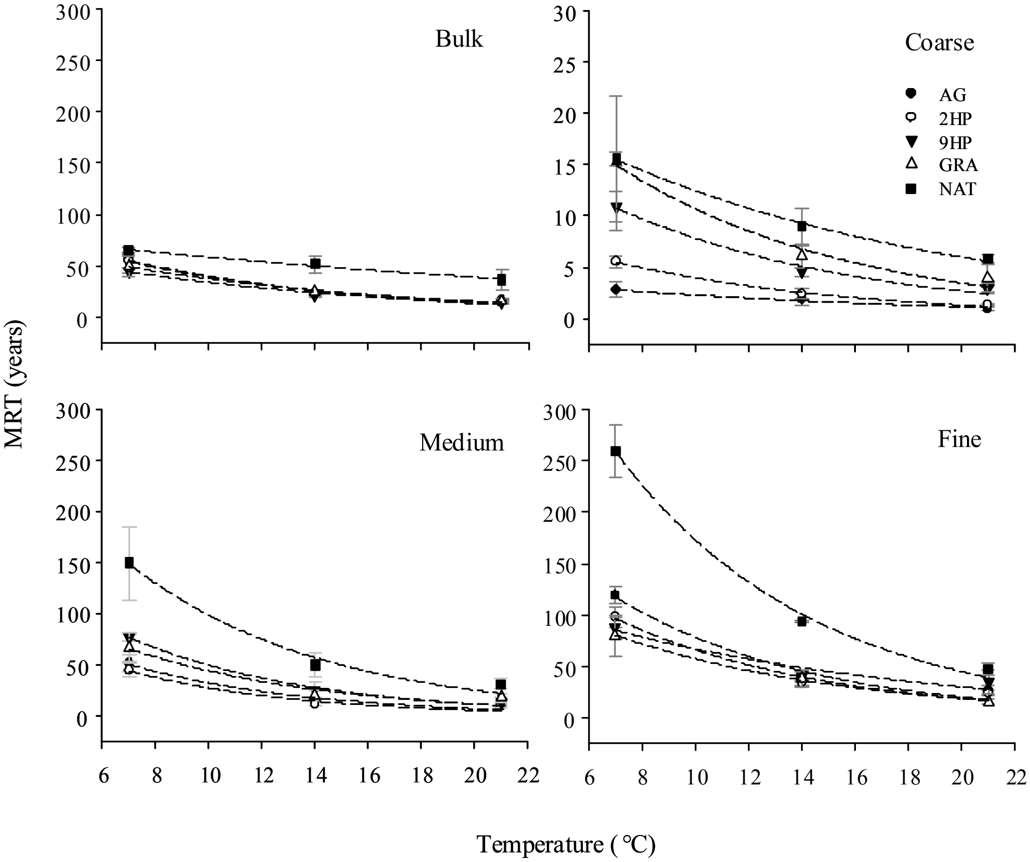

Figure 4-9. Relationship between mean residence times (MRT) of bulk and particle size fractions and incubation temperatures. Land uses: AG – agriculture, 2HP – 2-yr-old hybrid poplar plantation, 9HP – 9-yr-old hybrid poplar plantation, GRA – grassland, NAT – native aspen stand. Note different scale for coarse-size fractions.

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Chapter 5. Carbon balance under different land uses

Introduction

Changes in land use affect the cycling of carbon (C) and consequently the storage of C in ecosystems. The magnitude of change in C storage depends on how physical, chemical, or biological processes are changed under different land uses over time. For example, the conversion of natural to agricultural systems results in C loss due to removal of natural vegetation cover, and loss of C associated with mineralization, erosion, and leaching (Lal 2003). Such a conversion also reduces biomass C inputs and causes changes in soil temperature and moisture conditions that could accelerate organic matter mineralization. In North America, approximately 30-50% of soil C that amounts to 30-40 Mg C ha^{-1} has been lost to the atmosphere and soil organic C (SOC) stocks rapidly declined following conversion from natural to agricultural ecosystems (Lal 2006). Globally, it has been estimated that changes in land use (from forests to pastures or permanent croplands, and shifting cultivation) released 123 Pg C over the period from 1850 to 1990 (Houghton 1999), contributing about 18-20% of the total anthropogenic emissions of CO_2 each year (Dumanski 2004).

On the other hand, afforestation, reforestation, restoration of cultivated, abandoned and marginal soils can potentially reverse the process of C loss and increase ecosystem C storage (Ross et al. 2002). In this respect, the establishment of large-scale short-rotation plantations has been advocated as an effective method for sequestering CO_2 and mitigating increased atmospheric CO_2 levels (House et al. 2002), through increasing long-term C storage in woody biomass (Schimel et al. 2001) and in the soil (Garten Jr. 2002), and by providing bioenergy (Tuskan and Walsh 2001). Converting agricultural land to short-rotation woody crops has been reported to increase SOC content by 10 to 25 Mg ha^{-1} in 10 to 15 years (Grigal and Berguson 1998). However, the net C benefit of these plantations is still unclear, particularly as it is expected that those plantations would act as a C source during the initial years following establishment due to cultivation (for plantation establishment and weed control) that accelerate short-term mineralization of soil organic matter (Grigal and Berguson 1998).

If short-rotation plantations are to be used as potential C sinks, the quantity of C accumulation as well as the turning point when these plantations become C sinks must be determined. Net ecosystem productivity (NEP) conceptually reflects the net balance of C of different processes (Randerson et al. 2002). Estimation of not only the total C budget, but also of the sequestration potential of each C pool is important to understanding the dynamics of terrestrial C sinks. The objectives of this study were: (i) to examine the effect of land use and land use change on ecosystem C production and C sequestration potential, and (ii) to characterize the C source-sink dynamics in different land use systems.

Materials and Methods

Study area

The study area is located near Linaria (54°12' N, 114°8' W), approximately 25 km west of Westlock, in north central Alberta, Canada (Figure 5-1). The site has a continental climate, an approximate growing season of 180-185 days (AAFRD, 2003), and has a mean annual temperature of 3 °C and mean annual precipitation of 463 mm (Environment Canada 2005). The soil is classified as a Dark Gray Luvisol on medium textured till characterized by poor drainage, and a landform with low relief with a slope of less than 2%.

Three experimental plots (20×20 m^2) were set up in four different land use systems in this study: (i) an agricultural field (AG), (ii) a young and older hybrid poplar (*Populus deltoides* x *Populus x petrowskyana* cv. Walker) plantation (YHP and OHP, respectively), (iii) a grassland (GRA), and (iv) a native forest stand of trembling aspen (*Populus tremuloides* Michx., NAT). Details (including soil physical and chemical properties) of the four different land use sites are given in Chapter 2.

Net Primary Production

Net primary production (NPP), which is a direct measure of the amount of C that has been brought into an ecosystem, is expressed as the amount of plant material produced in an area per year. Net primary productivity is usually represented as biomass per unit area per time. However, in this study, I opted to express my estimates in terms of C per unit area per unit time. In YHP, OHP, and NAT systems, NPP was calculated using the following equation (Waring and Schlesinger 1985):

$$NPP = \Delta B + D + C + M \tag{1}$$

where,

ΔB - the change in plant biomass (except for foliage and fine root production) of trees i.e., aboveground woody biomass and coarse roots over a period of a year,

D - detritus (in the form of foliage, coarse woody debris, understory and fine root litter) produced during the year,

C - consumption of biomass by animals during the year; this component is often assumed to be negligible in healthy forest stands (Landsberg and Gower 1997) and was not considered in this study, and

M – mortality of woody biomass.

Due to the annual nature of biomass production for AG and GRA, ΔB was in these land uses were not calculated. Instead, annual biomass production was averaged between the two measurement years.

Tree diameter at breast height (DBH) and total height of all trees taller than 1.3 m in each experimental plot were measured in the YHP, OHP and NAT in 2006. Above- (stem, branches) and belowground (coarse roots) biomass was calculated using published allometric estimations for hybrid poplars (see Chapter 2). All plots were resurveyed in 2008. Then, ΔB was the difference in biomass between the two measurements for each land use.

Detritus production (D) was quantified by:

$$D = \text{foliage} + \text{coarse litter} + \text{fine roots} \tag{2}$$

Foliage production in YHP, OHP and NAT was computed using allometric equations whose independent variables were DBH and tree height (Fang et al. 2007, Wullschleger et al. 2005, Peterson and Peterson 1992). Because of the deciduous nature of these stands, foliage production for each measurement year was assumed to have returned to the soil each year as litterfall. Coarse woody debris (CWD) was only present in NAT and was measured using the line-intercept method (Halliwell and Apps 1997). Annual accumulation of CWD was calculated by taking the average difference between measurements obtained across two years (2006 and 2008) divided by two. Except in the YHP where the weed control practice was effective and no weed biomass was quantified, aboveground biomass of weeds at the OHP, crop and stubble at the AG, grass at GRA, and herbs and shrubs at NAT were measured using three randomly located clip plots (1×1 m) per experimental plot. All vegetation was removed from each clip plot, oven-dried at 65 °C until constant weight and weighed. Fine root production was measured using the maximum-minimum soil core method (Vogt et al. 1998). Three 6 cm^2 soil cores were taken to 30 cm depth in each experimental plot three times during the two growing seasons (June, August, and October of 2006 and 2008). Core samples were immediately stored at 2 °C until further processing. Fine roots (< 2 mm) were separated from the soil by soaking in water and gently washing the samples over two sets of sieves (2.0 and 0.5 mm). Fine roots were hand-picked and oven-dried at 65 °C for 48 h and weighed. Fine root biomass production for each year was then calculated by taking the difference between the maximum and minimum fine root biomass values. Biomass C of ground vegetation, litter and fine roots were calculated by multiplying the dry mass by a standard factor of 0.5 (Pregitzer and Euskirchen 2004).

Changes in soil C stock

Forest floor samples were collected from NAT in 2006 from three randomly assigned sample points (0.15 x 0.15 m) per experimental plot. The forest floor averaged 7.5-10 cm thick in this stand. Mineral soils from all land use sites were sampled from the surface (0-20 cm) and subsurface (20-50 cm) layers. In each experimental plot, three soil cores were collected and combined to form a composite sample for each depth. All soil samples were transported back to the lab and analyzed for total organic C using the TOC-V Total Organic Carbon Analyzer (Shimadzu Corp., Kyoto, Japan). Additionally, bulk density was determined for each soil depth in each experimental plot using a bulk density corer (172 cm^3). Soils in all land uses were resampled in 2008 to allow changes in soil C content from 2006 to be computed.

Soil surface CO_2 efflux and heterotrophic respiration

Soil surface CO_2 efflux (soil respiration, R_s) includes autotrophic respiration (R_a) from roots and heterotrophic respiration (R_h) resulting from soil organic matter decomposition by microbes and soil fauna. Soil surface R_s were measured at 15 to 27 locations per land use. To do that, short "collars" made from polyvinyl chloride (PVC) pipe (10 cm inside diameter and 6 cm long), beveled on one end, were gently hammered approximately 2 cm into the soil, leaving about 4 cm above the soil surface and were left in place for the duration of the study. Surface R_s within the collars was measured using a LI-COR 6400 portable photosynthesis system equipped with a LI-COR 6400-09 portable opaque soil chamber attachment (LI-COR Biosciences Inc., Lincoln, NE, USA). Measurements were made every four weeks during the growing seasons of 2006 (June 28, July 27, August 24, and September 26) and 2007 (May 29, June 28, July 26, August 30). All R_s measurements were carried out between 10:00 and 16:00 hours.

To determine the contribution of R_h to R_s, a modified root exclusion trenched plot technique was used following Zhou et al. (2007). This relied on severing all the roots around the perimeter of PVC collars in order to eliminate root respiration. For this purpose, nine long PVC collars (10 cm inside diameter and 32.5 cm long) per land use were inserted (on May 17, 2006) 30 cm into the soil. In order to avoid interference of respiration from plants, vegetation within the collars was regularly removed by clipping. Measurements of R_s (from the short collars) and R_h (from the long collars) were made at four-week intervals (June 28, July 27, August 24, and September 26 of 2006). The R_a was then calculated by subtracting R_h from R_s.

Annual R_s was extrapolated from bi-hourly measurements of CO_2 efflux at 2 cm below the soil surface, soil temperature and moisture measurements during the study period. Continuous soil CO_2 efflux measurements at 2 cm below the soil surface were obtained using solid-state infrared CO_2 sensors (Vaisala CARBOCAP® GMM series, Helsinki, Finland) connected to a datalogger (Model CR-10X, Campbell Scientific Inc., Logan, UT, USA) following methods described in Jassal et al. (2005). Continuous soil temperature and water content measurements were made using HOBO temperature sensors (Model TMC6-HD, Onset Computer Corp., Bourne, MA, USA) and porous matrix tensiometers (Watermark Sensor type WMSM, Delta-T Devices Ltd., Cambridge, UK), respectively. These sensors were connected to a HOBO external datalogger (Model U12-006, Onset Computer Corp., Bourne, MA, USA) programmed to take readings every 2 h when CO_2 concentrations were being measured by the Vaisala sensors.

To further quantify the contribution of particle size fractions to R_h, soil samples from each land use were collected and fractionated without chemical pre-treatment using a wet-sieving technique (Christensen 2001). Bulk soil and its different particle size fractions (coarse, 250-2000 μm; medium, 53-250 μm; fine, <53 μm) were incubated following Swanston et al. (2002) at three different temperatures (7, 14 and 21°C), at a constant soil moisture. Carbon loss was monitored for 370 days to determine the contribution of each fraction to R_h.

Net ecosystem production and C storage

Carbon pools (Mg C ha^{-1} yr^{-1}) in each of the land use systems were compartmentalized into: plant biomass (above- and belowground) and soil organic

C (in the upper 50 cm depth). Heterotrophic respiration was further subdivided to fluxes originating from coarse, medium and fine particle size fractions. Net ecosystem productivity (NEP), expressed in terms of C, was calculated as the difference between NPP and R_h (Curtis et al. 2002):

$$\mathrm{NEP} = \mathrm{NPP} - R_h \qquad (3)$$

The potential NEP ranges from negative (indicating a loss of stored C) to positive (indicating a gain in C). It should be noted, however, that R_h in equation (3) refers to ecosystem respiration i.e. not just soil R_h. I did not have the capacity, however, to measure total ecosystem respiration in this study. Therefore, it must be kept in mind that R_h measured from this study may have underestimated total ecosystem R_h. Also, it was assumed that C losses due to herbivory, volatile organic compounds or other processes were negligible to estimate annual ecosystem C storage:

$$\Delta C = \Delta B + \Delta S \qquad (4)$$

where ΔB is the annual increment in wood and ΔS is annual increment of C in soil:

$$\Delta S = D - R_h \qquad (5)$$

Results

Net primary production

Annual biomass production in AG averaged 9.72 Mg ha^{-1} yr^{-1} (dry weight), or 4.86 Mg C ha^{-1} yr^{-1} of which 3.79 Mg C ha^{-1} yr^{-1} was harvested and removed from the system (Table 5-1, Figure 5-3). Because of annual cultivation, residual biomass in the form of stubble and roots (1.50 Mg C ha^{-1} yr^{-1}) was incorporated into the soil resulting in $\Delta B = 0$ and $D = 1.50$ Mg C ha^{-1} yr^{-1}. NPP of AG, expressed in C terms, was thus equal to 1.50 Mg C ha^{-1} yr^{-1} (Table 5-2).

In YHP, woody biomass above- (stems and branches) and belowground (coarse roots) almost doubled two years after plantation establishment (Table 5-1). Annual change in woody biomass, ΔB, in YHP averaged 1.88 Mg ha^{-1} yr^{-1} (dry weight), or 0.94 Mg C ha^{-1} yr^{-1}, while litterfall and fine root production averaged 0.61 and 0.46 Mg C ha^{-1} yr^{-1}, respectively. NPP of YHP was thus equal to 2.02 Mg C ha^{-1} yr^{-1} which was 35% higher than NPP in AG (Table 5-2).

In OHP, above- and belowground biomass C increased by 30% within the two years of measurements although litterfall in OHP (4.68 Mg C ha^{-1} yr^{-1}) was 7 times and fine root production (0.83 Mg C ha^{-1} yr^{-1}) was 80% more than in YHP. Annual NPP in OHP was equal to 12.66 Mg C ha^{-1} (Table 5-2, Figure 5-3c), which was approximately 6 times greater than YHP and eight times greater than AG.

Net primary productivity in GRA was 3.84 Mg C ha^{-1} yr^{-1} (Table 5-2). Because GRA was regularly mowed, aboveground biomass C was maintained at 2.09 Mg C ha^{-1} yr^{-1} (Table 5-1). Belowground biomass production was 1.75 Mg C ha^{-1} yr^{-1}, which was four times greater than that in AG. It was assumed that all new biomass produced for each year died back and was returned to the system as litter.

In NAT, biomass of trees decreased during the two years from 56 Mg C ha^{-1} to 38 Mg C ha^{-1} aboveground and from 33 Mg C ha^{-1} to 23 Mg C ha^{-1} belowground (Table 5-1). Due to annual mortality, ΔB (expressed in C) in NAT decreased (-12.65 Mg C ha^{-1} yr^{-1}) while aboveground mortality and coarse litter increased (4.47 Mg C ha^{-1} yr^{-1}, Figure 5-4). Annual litterfall and fine root production was estimated at 6.75 and 1.47 Mg C ha^{-1} yr^{-1}, respectively, while the understory was 0.62 Mg C ha^{-1} yr^{-1}. The NPP (Table 5-2) was thus equal to 0.66 Mg C ha^{-1} yr^{-1}.

Net ecosystem production

Annual C losses through R_s, R_h, R_a, and proportions of R_h originating from soil particle size fractions (coarse, medium, and fine size fractions) across the various land-uses are presented in Figures 5-3 and 5-4. In AG, NEP (expressed in C) over two years was -1.65 Mg C ha^{-1} yr^{-1} (Table 5-2). Because no woody biomass was produced under AG, the change in ecosystem C storage was the same as the change in the soil C pool.

When AG was converted to hybrid poplar plantation, NEP increased from -1.65 to -0.75 Mg C ha^{-1} yr^{-1} (YHP, Table 5-2), indicating that the plantation, although it was acting as a C source, was able to sequester approximately 0.90 Mg C ha^{-1} yr^{-1} two years following plantation establishment. Although the trees contributed 1.07 Mg C ha^{-1} yr^{-1} to the soil as foliage and fine root litter, there was more C lost (1.69 Mg C ha^{-1} yr^{-1}) from the system through R_h. In the same period, 0.94 Mg C ha^{-1} yr^{-1} was gained as tree biomass, however, it was not enough to compensate for the loss from the soil. Nine years after plantation establishment (OHP) NEP (10.81 Mg C ha^{-1} yr^{-1}, Table 5-2) was 11 times greater than in YHP. Of the total C sequestered in OHP, 34% (3.66 Mg C ha^{-1} yr^{-1}) was stored in the soil system while 66% (7.15 Mg C ha^{-1} yr^{-1}) was stored in plant biomass (Figure 5-3c).

The NEP in GRA over the two years was 2.30 Mg C ha^{-1} yr^{-1} (Table 5-3) which was 2.8 times greater than that in AG. Assuming that all new biomass was returned to the system each year, 33% (0.76 Mg C ha^{-1} yr^{-1}) was stored in the soil C pool while the rest (1.54 Mg C ha^{-1} yr^{-1}) was lost from the system through R_h (Figure 5-3d). When comparing GRA with OHP, NEP in OHP was 5 times greater than that in GRA. The greater NEP in OHP was attributed to the accretion of C in plant biomass and the soil system i.e., OHP had 7 times more C in woody tissue and 60% more C in the soil system compared to GRA.

In NAT, plant biomass C decreased by 12.65 Mg C ha^{-1} yr^{-1} while detritus production increased by 13.32 Mg C ha^{-1} yr^{-1} in the two years of the study. With NPP at 0.66 Mg C ha^{-1} yr^{-1} and R_h at 4.98 Mg C ha^{-1} yr^{-1}, NEP was calculated to equal -4.32 Mg C ha^{-1} yr^{-1} which meant that the NAT stand was losing more C through respiration than it was gaining though photosynthesis (Table 2, Figure 5-4). In spite of the negative NEP, the stand was producing large amounts of litter that was being transferred to the soil system, which was accumulating 8.33 Mg C ha^{-1} yr^{-1}.

Discussion

Net primary production

Although the sites in this study differed in many ways from those reported in the literature, including local climate, soil characteristics, species composition, land use history, and measurement periods, the NPP (expressed in C values) were comparable to other studies. For AG, aboveground biomass C (which was under canola production during this study) was 4.85 Mg C ha^{-1} (which included straw and grain). This was comparable to average annual canola plant (straw plus grain) production (4.93 Mg C ha^{-1} for canola) estimated for Canada after assuming a harvest index (straw:grain ratio) of 50% and C content of oil seed crops of 50% (Wood and Layzell 2003). Of total crop NPP, 78% was removed during harvest and the remaining NPP was returned into the soil system in the form of crop residues and stubble (1.5 Mg C ha^{1} yr^{-1}). Crop residue in AG (22% of total aboveground biomass C produced) was within the range of relative proportions (21-78%) of returned to soil for Canadian agroecosystems (Bolinder et al. 2007). In GRA, aboveground biomass was not harvested thus 100% of the NPP was returned to the soil system. In the hybrid poplar plantations, ΔB (0.94 and 7.15 Mg C ha^{-1} yr^{-1} for YHP and OHP, respectively) accounted for 46 and 56% of NPP values (2.02 and 12.66 Mg C ha^{-1} yr^{-1} for YHP and OHP, respectively). Changes in woody biomass C were comparable to estimates reported for short-rotation woody crops by Dickmann (2006). He estimated that mean annual biomass increments for willows and poplars generally fall between 5-20 Mg ha^{-1} yr^{-1} which is translated to approximately 2.5-10 Mg C ha^{-1} yr^{-1}. In Canada, mean annual biomass increments for 14-20 year-old hybrid poplar and clonal aspen trees are estimated to range between 4.6-6.8 Mg C ha^{-1} yr^{-1} (Keddy et al. 2009). For native forest stands, Luyssaert et al. (2007) estimated NPP for boreal semi-arid region to range from 3.34 Mg C ha^{-1} yr^{-1} for evergreens to 5.39 Mg C ha^{-1} yr^{-1} for deciduous forests. Unique in this study, however, was that the increase in mortality in NAT resulted in a near zero NPP (0.66 Mg C ha^{-1} yr^{-1}), a result most likely affected by recent climate change and the mature status of the native aspen stand.

Forest ecosystem C fluxes may be significantly affected by climatic perturbations. The prairie provinces of western Canada, within which this study is located, experienced a severe regional drought in 2001-2002 (Bonsal and Wheaton 2005). The Climate Impacts of Productivity and Health of Aspen (CIPHA) study monitored aspen health and productivity in 30 areas across the boreal forest and adjacent aspen parkland of western and central Canada. They reported that mean precipitation in the region declined 27% below the long-term normal and that net mean biomass increment in the region decreased from 2.2 Mg ha^{-1} yr^{-1} (or 1.1 Mg C ha^{-1} yr^{-1}) in 2000-2002 to near zero following the drought (Hogg et al. 2008). They attributed the decrease in aspen productivity to defoliation and stem damage that lead to massive aspen dieback and mortality. The native aspen forest in this study manifested these same symptoms of deterioration which may be attributed to the effect of drought similarly experienced by other native aspen forests in the region. Mortality in older aspen stands is also typically large as aspens are short-lived trees reaching maximum development at about 50-60 years, and on poor sites at 30-50 years (Martin and Lorimer 1997).

Net ecosystem production

The estimated NEP of NAT (-4.32 Mg ha^{-1} yr^{-1}) in this study showed that it was a C source during the time of the study. This resulted from the decrease in biomass productivity and the increase in mortality (Table 5-2). As mentioned above, the site experienced a severe drought a few years earlier which caused defoliation and stem damage that lead to high dieback and mortality, similar to what's reported in Hogg et al. (2008). In a study that determined NEP at the onset of, during, and following the region-wide drought (2002-2003) in a mature boreal aspen stand in central Saskatchewan, Canada, Krishnan et al. (2006) reported that NEP for their site decreased to near zero (0.04 Mg C ha^{-1} yr^{-1}) following the drought, and that the decrease in NEP was 97% lower than the 11-year normal (pre-drought) NEP values that were recorded from the site (1.33 Mg C ha^{-1} yr^{-1}).

Conversion from native forest to agricultural land use results in C losses through both the initial depletion, associated with the removal of natural vegetation, and the subsequent losses from soil, through mineralization, erosion, and leaching (Lal 2003). From this study, it emerged that 80 years of conventional agricultural use, of a native forest soil, had resulted to an 18-fold decrease in plant biomass C and a 27% decrease of SOC in the top 50 cm soil. Although the NEP for NAT was negative, C was being transferred to pools of long-term storage (4.47 and 8.33 Mg C ha^{-1} yr^{-1} in coarse woody debris and soil, respectively) while in AG, most of the C produced each year was either harvested or lost back to the atmosphere (no C was being transferred to long-term storage). Considering the ecosystem as a whole, conversion from NAT to AG therefore considerably decreased the potential capacity of the site to accumulate C.

In AG, the NEP data indicated that the site was a source of C (-1.65 Mg C ha^{-1} yr^{-1}). Although AG produced 9.72 Mg ha^{-1} yr^{-1} as dry matter or 4.86 Mg C ha^{-1} yr^{-1}, a large portion (78%) of aboveground crop biomass C was harvested and removed from the site. A net source of C by agriculture was also observed by Saurette (2006) who reported that barely plots he studied in Boyle, Alberta had an average NEP value of -1.3 Mg C ha^{-1} yr^{-1}. Some may argue that the removed biomass should be included in the calculation of net primary productivity that is then used for calculating NEP. Understandably, if the removed portion is included, then the NEP of my agricultural site would become positive (2.14 Mg C ha^{-1} yr^{-1}). Saurette et al. (2008) included the harvested portion into their calculations, and determined their barely plots to be a small net C sink (0.03 Mg C ha^{-1} yr^{-1}).

Over nine years from the time AG was converted to GRA, NEP increased 2.8 times with an estimated net C gain of 2.30 Mg C ha^{-1} yr^{-1}. The NEP value was within the range of annual NEP values (between 1 and 6 Mg C ha^{-1} yr^{-1}) reported by IPCC (1996), depending on the radiation, temperature, water regimes, nutrient status, and age of the sward. During the growing season, the grass in this study site was not exported out of the system but rather, merely mowed and residues left in place to decompose. Due to the annual nature of grass, it was assumed that all the plant biomass (3.84 Mg C ha^{-1} yr^{-1}) was transferred to the soil system the same year it was produced. Close to half the amount of plant biomass C was lost to the atmosphere through R_h (1.54 Mg C ha^{-1} yr^{-1}).

When AG was converted to hybrid poplar plantation, NEP increased (it became less negative, -0.75 Mg C ha^{-1} yr^{-1}) two to four years after land use conversion, making it still a source of C relative to AG. Although the NEP of YHP was not determined during the first growing season in my study, NEP of the poplar plantation after land conversion was most likely more negative. For example, Saurette et al. (2008) reported NEP of hybrid poplar (Walker) plots in Boyle, Alberta was -3.35 Mg C ha^{-1} yr^{-1} in the first year after conversion from agricultural land use. Similarly, Price et al. (2009) measured net ecosystem exchange (NEE) from hybrid poplar plantation of the same clone (Walker) in central Alberta using the eddy covariance method and reported that the plantation was mainly a C source releasing more than 2.1 and 1.2 Mg C ha^{-1} yr^{-1} in the first and second year of growth, respectively; and that the plantation became C neutral (0.2 Mg C ha^{-1} yr^{-1}) in the third year and a small net C sink by the fourth year (-0.6 Mg C ha^{-1} yr^{-1}). Many studies have found that during plantation establishment, there occurs a net loss of C to the atmosphere, i.e., losses outweigh gains of C (Grigal and Berguson 1998, Hansen 1993, Paul et al. 2002, Wang et al. 2006) which may be caused by lower input of C early growth with increased rate of decomposition due to changes in micro-environmental conditions (Vesterdal et al. 2002). However, the magnitude and duration of when a plantation either acts as a C source or sink, is based on the type of species or clone used for afforestation, site location, climate, length of growing season, and the level of productivity and intensity of cultural practices.

Between nine to eleven years after plantation establishment, NEP in the hybrid poplar plantation increased significantly (10.81 in OHP vs. -1.65 Mg C ha^{-1} yr^{-1} in AG). The NEP of OHP lies within the range of NEP values (3.2 to 19.6 Mg C ha^{-1} yr^{-1}) reported for different forests types in Canada (Birdsey et al. 2007).

Variations in NEP values may be due to the differences in 1) management practices (irrigation, fertilization, planting at lower tree densities); and in 2) physical and climatic environments i.e., planting on old forest sites (therefore would have less labile organic C to potentially oxidize upon cultivation) and under a relatively milder climate. These factors allow for higher above- and belowground NPP, greater litter production and faster C stabilization in soil resulting in greater NEP (Waring and Schlesinger 1985).

C sequestration potential of hybrid poplar plantations

After the agricultural site was converted to hybrid poplar, in this study, soil C under the plantation decreased during the first two years. Shortly thereafter, soil C gradually increased and reached pre-plantation levels (114 Mg C ha^{-1}) i.e., soil C neutrality, at year 7 (Figure 5-2b). The rate of soil C loss in the plantation of this study (4 Mg C ha^{-1} yr^{-1}) was similar with that observed by Grigal and Berguson (1998) who collected data from five hybrid poplar plantations in Minnesota, ages 6-15 years, and compared soil C changes after they were established on cultivated land. They reported that the rate of soil C loss associated with their hybrid poplar plantations was equal to 3.5 Mg C ha^{-1} yr^{-1} in the first year, which declined to 2.6 Mg C ha^{-1} yr^{-1}, by year 5. The rate of increase in soil C that was found in this study (2 Mg C ha^{-1} yr^{-1}) was comparable to the accretion rate reported by Hansen (1993) which was 1.6 Mg C ha^{-1}yr^{-1} based on 10-20 year-old plantations that were studied in north central US.

The total amount of soil C lost from AG after hybrid poplar plantations were established, in this study, was 31 C Mg ha^{-1}, which was 27% of pre-plantation soil C levels (Figure 5-2b). This was within the range of values (20-50%) that several reviews (Davidson and Ackerman 1993, Gregorich et al. 2005, Guo and Gifford 2002, Mann 1986, Murty et al. 2002, Post and Kwon 2000) have estimated for soil C loss following land use change. Much of soil C loss is due to the low inputs from aboveground and belowground biomass (litter input from AG was 1.5 Mg C ha^{-1} yr^{-1}, Table 5-2); and tillage-induced disturbances that decrease soil aggregation and physical protection of soil organic C as observed in the increase in mineralized C in the medium and fine particle size fractions (Figures 5-3a and 5-3b).

In north central US, Hansen (1993) determined that soil C under hybrid poplar plantations reach pre-plantation levels between 8-10 years; while in Minnesota, Grigal and Berguson (1998) calculated that soil C under their plantations reached pre-plantation levels by year 15. In this study, soil C is estimated to reach pre-plantation levels by year 7, which is sooner than the two other studies mentioned. It was also calculated that soils under the plantations in this study will fully recover the total C lost from the initial 7 years and start functioning as net C sinks by year 11. The disparity between the results of this study and others may be attributable to the difference in climate. According to Paul et al. (2002), the most important factors affecting change in soil C following afforestation are: previous land use, forest type, and climate. However, given that the type of plantation in question compose of trees that are genetically similar and are managed more or less the same way, it can be argued that the difference from when these plantations convert from being C sources to C sinks lie on the differences in climate. The previous studies were all located in north central US where the plantations were under milder climate with longer growing seasons compared to study sites in northern Alberta. Milder climate and longer growing seasons promote higher litter production and faster decomposition (Paul et al. 2002), which consequently lead to greater soil C loss via soil respiration. Whereas, in colder climates, growing seasons are shorter and decomposition is slower. This leads to lower soil C loss via respiration that consequently increases soil C retention. It follows that there may be a greater capacity for long-term recovery of soil C following afforestation of cooler sites compared to warm temperate sites.

If trees were included when viewing C sequestration potential of hybrid poplar plantations at an ecosystem level, then the C source-sink dynamics of the system would be slightly different. In the first two years, the plantation acted as a C source due to small contribution of plant biomass and litter hybrid poplars relative to soil C loss via heterotrophic respiration (Table 5-2). After the third year however, ecosystem C started to increase, accreting ecosystem C at a rate of 11 Mg C ha^{-1} yr^{-1} (above-, belowground biomass, and soil accretion rates are 7, 2, and 2 Mg C ha^{-1} yr^{-1}, respectively).

Ecosystem C in hybrid poplar plantations reached neutrality on year 4, which translated to 10.81 Mg C ha^{-1} of total C loss from the ecosystem. In order to compensate the total C loss, it merely took the ecosystem one more year to start acting as an ecosystem C sink which is attributable to the increase in plant biomass both above and belowground. As mentioned above, hybrid poplar plantations in this study (based on biometric measurements) reached C neutrality and acted as net C sinks one year later (year 5) than plantations studied by Price et. al. (2009, year 4) that was based on eddy-covariance methods. The probable cause of the difference of these estimates may be due to the differing temporal estimation of C dynamics of both methods (Curtis et al. 2002). Nevertheless, on an ecosystem level, data show that growing hybrid poplars on rotations less than 4 years would result in C loss but would otherwise sequester significant quantities of C if the rotation is longer than 5 years. This characteristic will indeed contribute to the attractiveness of growing short-rotation tree plantations as a strategy for aiding the sequestration of atmospheric CO_2 especially under the pressing need to mitigate the effects of future climate change.

Tables and Figures

Table 5-1. Carbon storage in 2006 and 2008 for various land uses in Linaria, AB (AG – agriculture; YHP – young hybrid poplar plantation, OHP – older hybrid poplar plantation). Values are means and SE, in Mg C ha^{-1}.

	AG		YHP		OHP		GRA		NAT	
	2006[*]	2008	2006	2008	2006	2008	2006	2008	2006	2008
Aboveground biomass C	1.0[†] (0.0)	1.1 (0.2)	2.0 (0.2)	3.8 (0.7)	43.9 (5.6)	55.9 (4.6)	1.8 (0.3)	2.3 (0.4)	56.5 (17.2)	37.9 (8.4)
Crop biomass	4.8 (0.3)	4.9 (0.6)								
Harvested biomass	3.8 (0.1)	3.8 (0.4)								
Stem			1.0 (0.1)	1.8 (0.3)	31.4 (4.0)	41.0 (3.5)			41.9 (13.8)	28.8 (6.4)
Branches			0.6 (0.1)	1.2 (0.2)	6.0 (0.8)	7.8 (0.7)			5.5 (1.5)	3.5 (0.9)
Foliage			0.5 (0.1)	0.8 (0.1)	4.6 (0.6)	4.7 (0.4)			8.0 (2.6)	5.5 (1.2)
Grass					1.8 (0.3)	2.3 (0.4)	1.8 (0.3)	2.3 (0.4)		
Understory									1.0 (0.7)	0.2 (0.0)
Belowground biomass C (0-30 cm)	0.7 (0.2)	0.1 (0.0)	1.0 (0.1)	1.3 (0.2)	10.9 (0.8)	12.5 (1.2)	2.8 (1.2)	0.7 (0.3)	32.8 (9.6)	23.2 (5.2)
Coarse root			0.5 (0.1)	0.9 (0.2)	9.4 (1.2)	12.3 (1.0)			31.6 (10.0)	21.5 (4.8)
Fine root	0.7 (0.2)	0.1 (0.0)	0.5 (0.2)	0.4 (0.3)	1.4 (0.4)	0.2 (0.2)	2.8 (1.2)	0.7 (0.3)	1.2 (0.5)	1.7 (0.4)
Soil C	114.7 (7.0)	120.9 (3.6)	106.8 (2.3)	108.7 (3.9)	119.4 (14.2)	127.4 (12.1)	113.6 (8.3)	123.4 (12.1)	136.8 (17.0)	184.9 (9.6)
LFH									11.1 (1.5)	13.3 (1.5)
Snags									8.3 (4.3)	16.0 (2.4)
Coarse woody debris									14.8 (1.3)	13.9 (4.4)
Mineral soil (0-20 cm)	75.4 (2.3)	81.5 (3.0)	67.6 (2.5)	69.5 (4.1)	78.2 (8.8)	86.2 (4.1)	75.6 (4.6)	85.4 (8.3)	55.3 (17.6)	74.3 (14.2)
Coarse fraction	4.5 (0.0)		4.8 (0.0)		9.4 (0.0)		9.8 (0.8)		6.6 (0.0)	
Medium fraction	20.9 (0.8)		19.8 (0.0)		24.3 (0.8)		24.2 (0.8)		18.2 (0.6)	
Fine fraction	50.0 (0.8)		43.0 (0.7)		44.6 (0.0)		41.6 (0.8)		30.4 (0.6)	
Mineral soil (20-50 cm)	39.3 (5.5)	39.3 (5.5)	39.1 (0.3)	39.1 (0.3)	41.2 (8.2)	41.2 (8.2)	38.0 (3.7)	38.0 (3.7)	47.3 (5.9)	47.3 (5.9)
Total ecosystem C storage	118.6 (6.8)	124.3 (3.3)	109.8 (2.2)	113.7 (4.5)	174.1 (20.5)	195.8 (17.4)	118.2 (7.4)	126.5 (12.0)	226.1 (13.5)	225.8 (16.2)

[*]: Data for 2006 has been shown in Chapter 2.

[†]: Aboveground biomass C for agricultural site was the difference between total crop biomass production and crop biomass harvested.

Table 5-2. Annual carbon budgets for various land uses in Linaria, AB (AG – agriculture; YHP – young hybrid poplar plantation, OHP – older hybrid poplar plantation; GRA – grassland; NAT – native aspen stand). Values (SE) are in Mg C ha^{-1} yr^{-1}.

	AG	YHP	OHP	GRA	NAT
ΔB	0.00 (0.00)	0.94 (0.00)	7.15 (0.00)	0.00 (0.00)	-12.65 (0.00)
D	1.5 (0.15)	1.07 (0.08)	5.51 (0.41)	3.84 (0.48)	13.32 (4.21)
NPP	1.5 (0.15)	2.02 (0.52)	12.66 (0.28)	3.84 (0.15)	0.66 (0.80)
R_h	3.15 (0.32)	2.76 (0.48)	1.85 (2.04)	1.54 (0.48)	4.98 (3.31)
NEP	-1.65 (0.15)	-0.75 (0.08)	10.81 (0.41)	2.30 (0.48)	-4.32 (4.21)

The values (SE) are based on the mean over two years (2006 and 2008). ΔB is the annual change in plant biomass above- and belowground + fine root production; *D* (detritus) is the litter production: foliage + coarse woody debris + understory + fine root; *NPP* (net primary production) = $\Delta B + D$; *NEP* (net ecosystem production) = NPP - R_h; a positive NEP value represent C input to the ecosystem; and a negative value denotes C outputs from the ecosystem to the atmosphere.

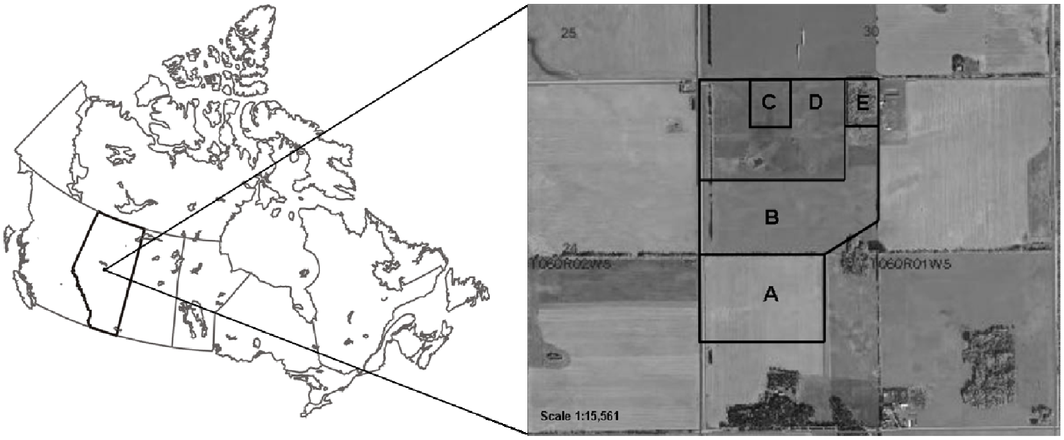

Figure 5-1. Location of land use study sites (A – agriculture; B – young hybrid poplar plantation, C – older hybrid poplar plantation; D – grassland; E – native aspen stand) in Linaria, Alberta, Canada. The source of the aerial photo is http://www.agriculture.alberta.ca. The use of the aerial photo by the author is done without any affiliation with or endorsement by the Government of Alberta. Reliance upon the author's use of this material is at the risk of the end user.

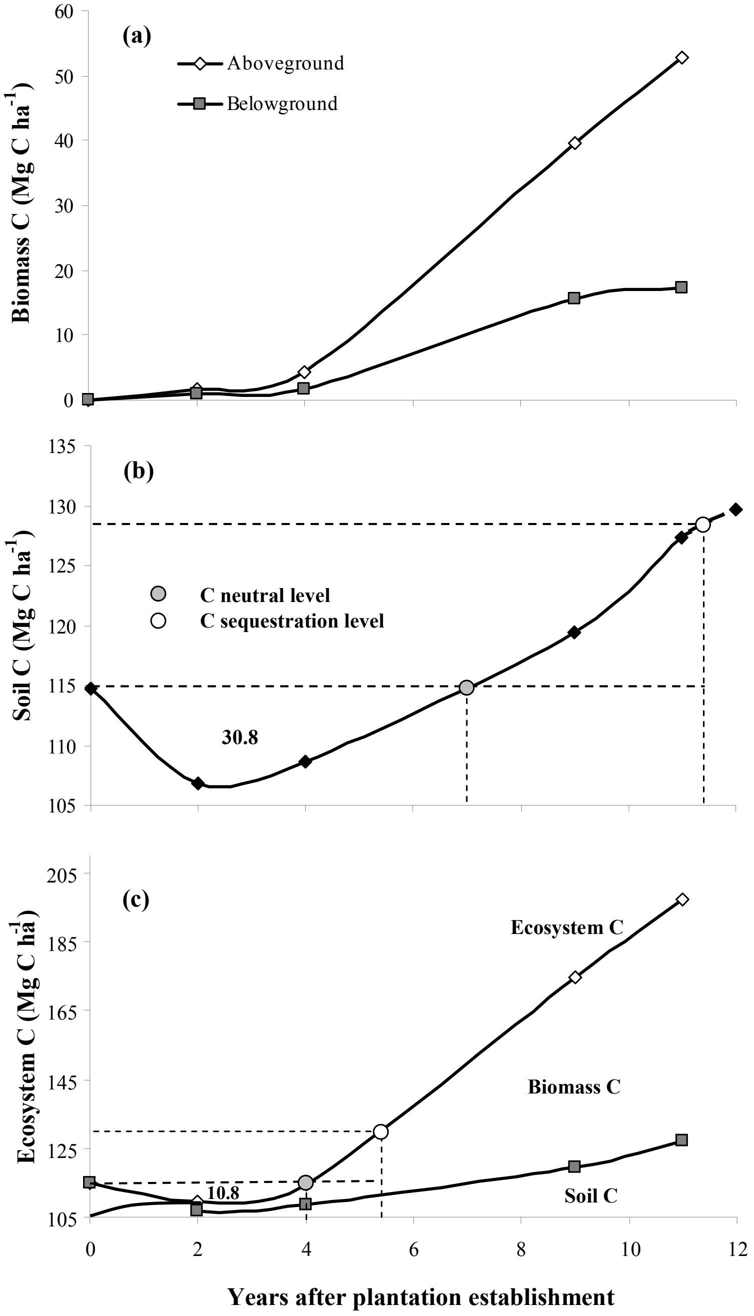

Figure 5-2. Carbon storage in above-, belowground biomass and soil pools 2 to 11 years after hybrid poplar plantation establishment on agricultural soil in Linaria, AB. C storage values for Time=0 were obtained from an adjacent agricultural site.

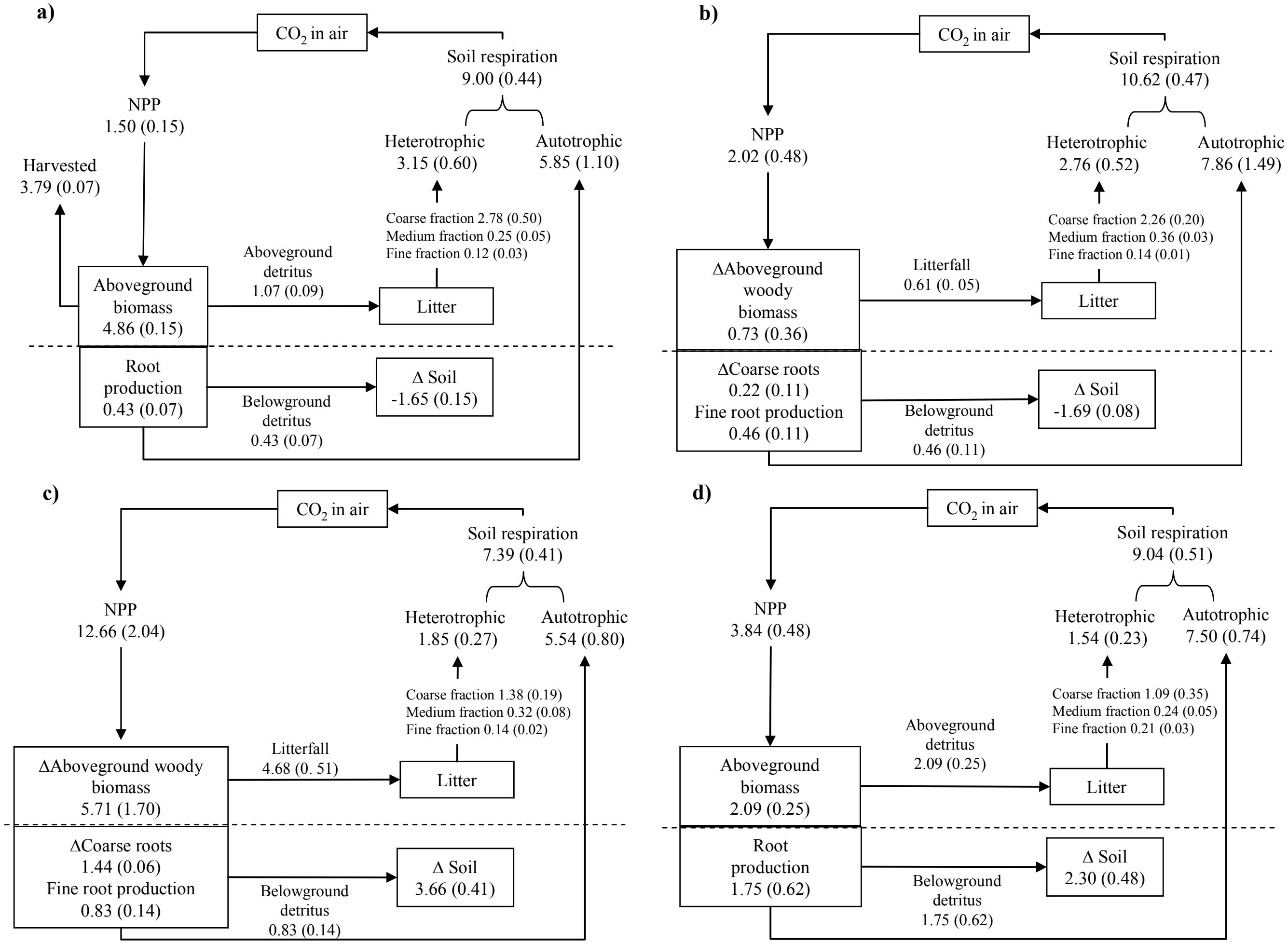

Figure 5-3. Estimated annual total C pools and fluxes in a) agricultural, b) young hybrid poplar plantation, c) older hybrid poplar plantation, and d) grassland systems in Linaria, AB. Values (SE), Mg C ha^{-1} yr^{-1}, were averaged over two years (2006 and 2008). Squares indicate C pools while arrows indicate C fluxes, which were measured or estimated biometrically in this study. Fine root biomass includes both live and dead fine roots.

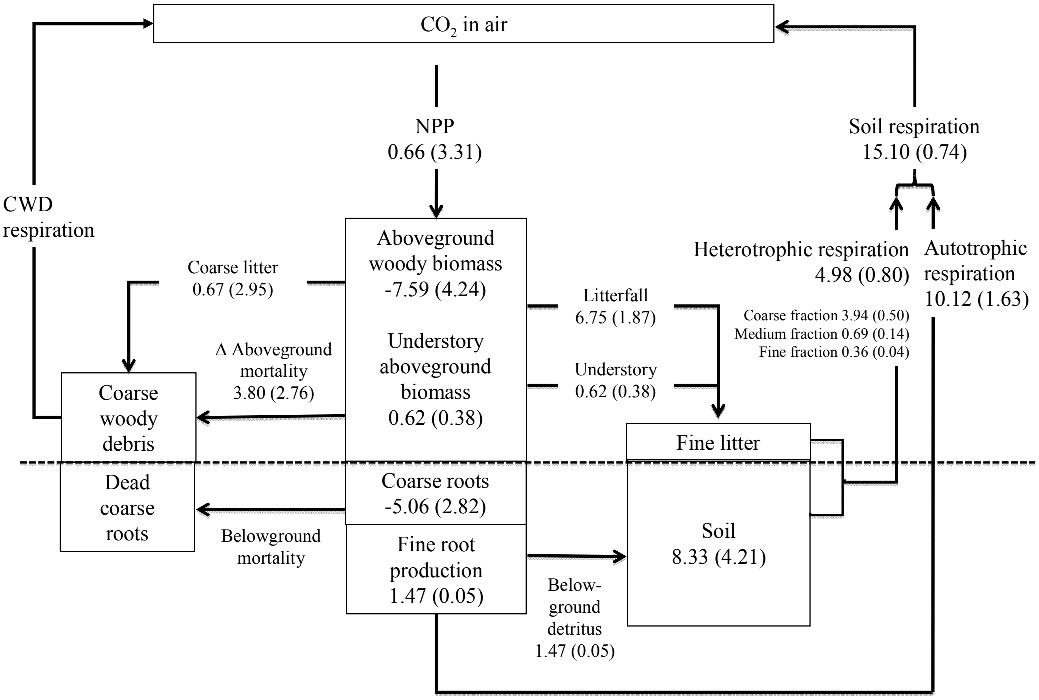

Figure 5-4. Estimated annual total C pools and fluxes in a native aspen forest in Linaria, AB. Values (SE), Mg C ha^{-1} yr^{-1}, were averaged over two years (2006 and 2008). Squares indicate C pools while arrows indicate C fluxes, which were measured or estimated biometrically in this study. Fine root biomass includes both live and dead fine roots.

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Chapter 6. Summary and Conclusions

Summary

The main objective of this thesis research was to compare ecosystem C stocks and distribution, soil respiration, soil C mineralization, and net ecosystem production across different land uses and determine how these aspects of the ecosystem C cycle changed with land use change. The specific objectives of this study were:

1. To characterize trends in C storage and distribution in ecosystem components and to understand the influence of land use change on the dynamics of ecosystem C storage.
2. To determine seasonal and diurnal changes in soil respiration rates under different land uses; partition soil respiration into its heterotrophic and autotrophic components; and examine the factors affecting variations in soil respiration and its components
3. To determine mineralization rates and mean residence time of C in soils under different land uses; to determine the contribution of different particle size fractions to total mineralizable C from the bulk soil; and to examine the effect of changes in temperature on the mineralization and turnover rates of C in bulk soils and particle size fractions from different land uses
4. To examine the effect of land use and land use change on ecosystem C production potential and characterize the C source-sink dynamics in different land use systems

In the first study, it was determined that ecosystem C stock in the native aspen stand (223 Mg C ha^{-1}) was similar to that of the 9-yr-old hybrid poplar plantation (174 Mg C ha^{-1}) and both were significantly greater than in the agriculture (122 Mg C ha^{-1}), grassland (121 Mg C ha^{-1}), and 2-yr-old hybrid poplar plantation (110 Mg C ha^{-1}). Above- and belowground biomass C stocks accounted for the observed differences between the native aspen stand and other land uses i.e., the native aspen stand had 6 times greater plant biomass C compared to the other land uses. Although soil C content in the 0-50 cm soil layer was 4-59 times greater than biomass C in the different land uses, significant differences in soil C among the land uses were not detected. Even 80 years of conventional agricultural use, of a native forest soil, had only resulted in a slight non-significant decrease of soil C. It should be noted however, that a non-significant decrease of soil C was observed two years after hybrid poplar plantations were established and a non-significant increase of soil C occurred after nine years. Soil C loss in the early stages of plantation development may be due to greater C outputs via soil respiration brought about by cultivation during site preparation and weed management.

For the soil particle size fractions, C stocks decreased in the order of: fine (< 53 μm) > medium (53-250 μm) > coarse (250-2000 μm) for all land uses, except in the native aspen stand where C was uniformly distributed among soil particle size fractions. High C stock in the fine particle size fraction was related to the high recovery of silt and clay (average of 59 %) plus their high C. High C content in fine particle size fractions could result from a combination of factors including: lower decomposition rates of silt- and clay-associated organic matter, the transfer of stabilized decomposition products from other size separates, and the accumulation of more stable soil organic matter. In terms of land use change, C stock doubled in the coarse-size fraction nine years after hybrid poplar plantation was establish on agricultural soil whilst no other changes within the other particle size fractions were observed. This suggests that after the soil disturbance and mixing by cultivation stopped in the hybrid poplar plantations there was build up of organic matter in the coarse-size fraction nine years after plantation establishment. Similarly, a slower soil organic C incorporation rate in smaller particle size fractions was expected in the native aspen stand as the ecosystem was undisturbed and where soil aggregation was expected to be high compared to the other land uses.

In the second study, soil respiration followed a pronounced seasonal trend – increasing during the growing season and converging to minimum in the fall. Diurnally, soil respiration showed early morning minima (generally before 09:00) and late afternoon maxima (generally after 15:00 h). Differences in soil respiration between the different land uses were explained mainly by site differences in soil temperature (88-94%). Vegetation cover can alter soil temperature and soil moisture conditions which in turn, can significantly influence soil respiration rates. I found that soil temperature in the native aspen stand was consistently lower compared to the agricultural field. This is likely due to the shading of trees and the presence of the LFH layer. The LFH layer acts as an insulating mulch to reduce heat gain (from solar radiation) during daytime and loss during nighttime and hence moderate temperature conditions.

Cumulative soil C loss via soil respiration averaged over two growing seasons increased in order of: 2-yr-old hybrid poplar plantation (4.28 Mg C ha^{-1}) < agriculture (5.02 Mg C ha^{-1}) < grassland (5.23 Mg C ha^{-1}) < 9-yr-old hybrid poplar plantation (5.51 Mg C ha^{-1}) < native aspen stand (7.81 Mg C ha^{-1}). Differences in cumulative soil respiration between the land uses were related to 1) ecosystem C stock, 2) temperature sensitivity (Q_{10}) of organic matter present, and 3) organic matter decomposability (determined from the natural abundance of δ^{13}C). Cumulative soil C loss via heterotrophic and autotrophic respiration among the land uses ranged from 0.97 to 2.72 and 3.33 to 5.60 Mg C ha^{-1}, respectively; with respective contributions to soil respiration of up to 35 and 83%. Heterotrophic respiration was influenced mainly by soil temperature while autotrophic respiration was influenced by fine root biomass and nutrient availability (NO_3^- and P).

In the third study, mineralized C from the bulk soil across the different land uses ranged between 2 to 5% of initial total organic C, with mineralization rates ranging from 0.06 to 0.12 µg C mg^{-1} C_i d^{-1} and mean residence times ranging from 30 to 51 yrs. Across particle size fractions, % C mineralized, mineralization rates and turnover times of C were increasing in the order of: native aspen stand < grassland < hybrid poplar plantations < agriculture. Lower % C and mineralization rates with higher MRT under the native aspen stand may be due to the exclusion of the LFH layer, where recent inputs of C undergo much of the microbial processes before they translocate into the mineral soil. Once C reaches the mineral soil and forms mineral-organic complexes, it becomes more stable and less prone to further decomposition. When native soil however was converted to agriculture, % C, mineralization rate and turnover of C significantly increased. This may be due to fresh litter from crop residues and roots annually added and incorporated into the system by cultivation. Mixing by cultivation causes the breakdown of soil structure that enhances oxidation of physically and chemically protected C inside aggregates. The combination of constant additions and mechanical disintegration of new organic matter every year resulted in higher C losses from agriculture. Conversion from agriculture to either hybrid poplars or grassland, on the other hand, did not lower % C, mineralization or turnover rates. Although, there was some indication (^{13}C and ^{15}N in the bulk soil) that litter decomposability in the plantations decreased. Conversion of agriculture to plantation or grassland started reversing the effect of cultivation on organic C pool. Although these changes maybe slow to occur, lower rates of mineralization are of significance for soil management.

Across land uses, the coarse fractions, representing labile C, were the main source of mineralized C (79%), followed by the medium (14%) and fine fractions (7%). Carbon within the coarse and medium fractions had higher temperature sensitivity when a hybrid poplar plantation was established on agricultural land, and mineralized up to 7 times more C when temperature increased. With plantation establishment, litter returned to the system becomes rich in lignin. Lignin, being composed of aromatic polymers with higher molecular weight, is more resistant to decomposition and is more responsive to temperature change compared to labile OM. Because of this, enzymatic reactions required to metabolize structurally complex C substrates involve higher activation energies than reactions metabolizing simpler or more labile C substrates. With increase in temperature, there is higher activation energy available resulting in increased rate of decomposition of these complex C. While the data suggest that the change in C substrates with plantation establishment on agricultural land results in lower degree of mineralization and turnover of C, it was also found that with land conversion, soil C mineralization also became more responsive to change in temperature. This may have important implications in the future ecosystem models that, even with small increases in temperature, it may prompt large releases of C as soils from these types of land use respond to global climate change.

In the final study, net ecosystem productivity across land uses was calculated to range between -1.65 (AG) and 11 Mg C ha^{-1} yr^{-1}(OHP) ; while changes in C storage over the course of two years (2006-2008) ranged between 0 and 7 and between -2 and 8 Mg C ha^{-1} yr^{-1}, as biomass and soil organic C, respectively. Conversion from native aspen forest to agriculture led to a 10-fold and 27% decrease in biomass and soil C in the 0-50 cm depth, respectively. When agriculture was converted to grassland, heterotrophic respiration decreased by half, and net ecosystem productivity increased 2.8 times (from -1.65 in agriculture to 2.3 Mg ha^{-1} yr^{-1} in grassland). When agriculture was converted to hybrid poplar plantations, the stands were initially large sources of C. As cultivation ceased and trees grew bigger, the soils under plantations became C sinks by year 7, and recovered soil C lost from the early plantation establishment phase by year 11, indicating that growing hybrid poplars on rotations longer than 11 years would sequester significant quantities of C. At the ecosystem level, however, hybrid poplar plantations were a source of C in the first 4 years. Due to the fast growing nature of the hybrid poplars, plant biomass C accretion (11 Mg ha^{-1} yr^{-1}) caused the plantation to become C sink by year five. This characteristic will indeed contribute to the attractiveness of growing short-rotation tree plantations as a strategy for aiding the sequestration of atmospheric CO_2 especially under the pressing need to mitigate the effects of future climate change.

Project Limitations

In this thesis research, one limitation that I encountered was that the combination of land uses that I studied was very complex and unique, making it difficult to have the experimental design in this study truly replicated. Because of the nature of pseudo-replication, it should be noted that the sampling plots established in each of the land uses did not represent independent units with the same treatment (land use) but rather should be considered as sub-samples within non-replicated land use treatments. Results obtained from this study can therefore only be applied to the research plots studied, and that they cannot be used to make wider inferences to the greater population of possible plots and treatments.

Ecosystem stocks and the distribution of C were measured and compared across four different land uses. One possible limitation of this experiment was the assumption that the soils were similar prior to the imposition land use change. Soil C at the 20-50 cm under the different land uses were determined to be statistically similar which may support this assumption. Although I was not able to collect data from the C horizons, I am confident that the sites developed from the same (or at least, similar) parent material considering that study sites 1) were all under the same land use (native aspen forest) before they were all cleared for agricultural expansion 80 years ago; 2) they are adjacent to each other; 3) and they are located in a relatively flat land. For other soil properties (K, Ca, Mg, pH, proportions of silt and clay), I recognize that NAT was slightly different compared to the other land uses in this regard and attribute this to the greater influence of land use conversion and cultural practices/land management employed over 80 years than to dissimilarities of parent materials.

Soil C at the 0-50 cm layer were determined to be statistically similar among the different land uses in this study even though my measurements indicated soil C in NAT in 2006 (136.8 Mg C ha^{-1}) was 19% more than soil C measured in AG (114.8 Mg C ha^{-1}, see Table 2-3); and that cumulative soil C lost over the course of 7 years when agriculture was converted to plantation was 31 Mg C ha^{-1} or 27% pre-plantation soil C levels (Fig. 5-2b). In the field of soil science, 19-27% difference of soil C is quite significant when comparing two systems but this was not the case in this study, statistically-speaking. Non-significant differences between the land uses may be due to 1) possible differences in initial soil C stocks; 2) large spatial variabilities within each of the systems that I may not have been able to capture; and/or 3) I may not have had collected enough number of samples to increase the power of my statistical tests.

Tree biomass C estimates in this study were obtained using a combination of allometric equations. These equations were not clone-, age-, and site-specific to my study sites but rather, these equations were developed for other clones of hybrid poplars grown under different field site conditions. Using allometric equations that are not clone-, age-, and site-specific may cause under- or overestimation of the biomass estimates. It is also recognized, that allometric equations use parameters that are located aboveground e.g. DBH and height, and that changes aboveground may not necessary correspond to changes belowground as growing patterns are, again, clone-, age-, and site-specific. The estimates for coarse roots in this study were based on allometric equations using DBH as the dependent variable. The magnitude of uncertainties for above- and belowground estimates in this study was not able to be estimated.

In terms of estimating C storage along various soil C pools, I fractionated bulk soil to its particle size fractions using the ultrasonic dispersion and wet-sieving technique. I originally opted to separate bulk soil into the following fractions: 250-2000, 53-250, 2-53, and <2 μm. With the intention of preserving aggregates, ultrasonic dispersion was conducted at 60% of maximum power for 2 min. After fractionating and comparing the proportions of recovery with particle size analysis previously conducted for the same soil samples, however, I found that I was not able to fully separate the finer particle size fractions i.e., I was not able to fully separate the <2 μm from the 2-53 μm fraction. Because of this, further experimentation on this technique would become useful. I am assuming that prolonging the ultrasonic dispersion or increasing the dispersion power would increase the recovery of <2 μm fraction. However, this will, in turn, rid the samples of aggregation. It may be possible that another form of separation would be more appropriate for finer particle size fractionation.

The partitioning method (root exclusion) I used involved severing (but not removing) all the roots around the perimeter of collars in order to eliminate root respiration and inhibit future root growth. One limitation of this technique is that cessation of root activity within the collars would have eliminated transpiration losses resulting to possibly higher soil moisture, which may in turn, affect decomposition and potentially overestimate heterotrophic respiration (or underestimate autotrophic respiration) in my calculations.

Finally, another limitation is related to soil CO_2 efflux measurements that I took using CO_2 sensors. In this experiment, I only had two points along the soil profile i.e., at 2- and 20 cm. In order to calculate for diffusivity, I assumed that CO_2 concentration at 0 cm was 380 ppm. If a sensor was installed at 0 cm to obtain ambient CO_2 concentration, estimates for flux values may be more accurate.

Suggestions for Future Research

The results from this thesis may have implication for improving research projects summarized above compared ecosystem C stocks and distribution, soil respiration, soil C mineralization, and net ecosystem production across different land uses (agriculture, plantation, grassland, and native forest) and determined how these aspects of the ecosystem C cycle changed with land use change in north central Alberta. There are numerous opportunities for future research on C dynamics within different land uses however I would like to suggest that more attention be geared towards understanding short rotation woody crop plantations as many questions still need to be answered in relation to its ecosystem C sequestration potential.

Based on the net ecosystem productivity of hybrid poplar plantations, I presented the C source-sink relationship of hybrid poplar plantations in the first 13 years of its life cycle. In the first two years, I found that soil C decreased by

27% relative to pre-plantation levels. Soil C storage then increased and reached neutrality by year 4, started to serve as a C sink by year 7, and fully recovered soil C lost by year 11. Would this C source-sink pattern for hybrid poplars, determined for my study sites, hold true for other sites with different soil types and/or climatic conditions? How much soil C would be lost in the first few years after plantation establishment under different soil types and/or site conditions? How long will the soil continue to lose C after plantation establishment? How fast will soil C recover?

This study was focused on hybrid poplar plantations that were managed under one management regime. The Canadian Wood Fibre Centre, through the Canadian Forest Service of Natural Resources Canada, is currently using other management regimes that are more intensive across Canada. How will ecosystem C sequestration potential differ with more intensive management regimes? How will the potential change under different hybrid clones? Under different soil types?

Finally, establishment of short rotation woody crop plantations for the purpose of sequestering C into their ecosystems seem promising, most especially if efforts were focused on a large-scale throughout Canada. However, C sequestration potential is influenced by site conditions. For instance, given appropriate moisture, productivity will increase with increasing temperatures. Although soil respiration will also increase with increasing temperatures, increased biomass production may compensate for C loss due to soil respiration. If there is a limitation in moisture, on the other hand, belowground biomass may increase in search for water availability. This will, in turn, cause shifts in the proportions of C losses via heterotrophic and autotrophic respiration i.e., heterotrophic respiration may decrease with decreasing moisture availability and autotrophic respiration may increase with decreasing moisture availability. With global concern on increasing atmospheric CO_2 concentrations, rising temperature, and changing precipitation patterns, how will ecosystem C sequestration potential of short rotation woody crop plantations respond to changes in temperature and moisture under future climate change?

Conclusions

Differences in ecosystem C stocks between the land uses were primarily the result of different plant biomass as SOC was unaffected by land use change although the litter quality of SOC as indicated by the enrichment of the natural abundances of ^{13}C and ^{15}N across the land uses suggested shifts from more labile forms of C to more humified forms of C following land use changes.

Soil C emissions via R_s averaged over two growing seasons were: NAT > 9HP > GRA > AG > 2HP. Seasonal R_h and R_a among the land uses had respective contributions to R_s of up to 35 and 83%. The variability in R_s across the different land uses was explained mainly by site differences in T_s (88-94%).

Soil C mineralization potential of bulk soil across the different land uses ranged between 2 to 5% of initial total organic C, with mineralization rates ranging from 0.06 to 0.12 μg C mg^{-1} C$_i$ d^{-1} and mean residence times (MRT) ranging from 30 to 51 yrs. Mineralization potential of soil, across particle size fractions, were in the order of: AG > HPs > GRA > NAT of which the coarse fractions were the main source of mineralized C (79%), represented the labile C, and was most affected by land use change. Mineralization potential increased when NAT was converted to AG; and decreased when AG was converted to HPs or GRA.

Net ecosystem productivity across land uses ranged between -1.65 and 11 Mg C ha^{-1} yr^{-1}. Conversion from NAT to AG led to a 10-fold and 27% decrease in biomass and soil organic C in the 0-50 cm depth, respectively. When AG was converted to GRA, NEP increased 2.8 times. When AG was converted to HP, NEP indicated the plantation to be a C source in the first four years and a C sink by year five.

Taken together, the results of these studies indicate that land use conversion from natural to agricultural systems dramatically decreases ecosystem C stocks, increases soil C emissions, destabilizes physically- and chemically-protected SOC that result in near zero net ecosystem productivity. On the other hand, land use conversion from agricultural to either hybrid poplar plantation or grassland reverses the process of C loss and increases ecosystem C storage. Although soil C emissions increased with increase in plant biomass input, hybrid poplar plantations or grassland rapidly stabilizes SOC, and increases net ecosystem productivity in the long run. Quantifying the net C balances of various land uses and determining how these balances change with land use conversions is necessary for validating the C sequestration potential of these systems and for understanding the role of these ecosystems in mitigating future climate change.

Appendix 1

Table A1-1. Correlations (Pearson's r) between various soil biophysical variables and heterotrophic (R_h) and autotrophic respiration (R_a) in all studied land uses in Linaria AB.

	R_h		R_a	
Soil variables	Correlation r	*p-value*	Correlation r	*p-value*
pH	-0.46	0.43	-0.36	0.55
Soil organic N	-0.35	0.56	-0.67	0.22
Soil organic C	-0.31	0.61	-0.68	0.20
POM-C	-0.48	0.41	0.36	0.55
NH_4^+	0.42	0.48	-0.74	0.16
NO_3^-	0.34	0.58	-0.92	0.02 *
P	0.34	0.58	-0.94	0.02 *
T_s	0.79	0.10 †	-0.47	0.42
θ_v	0.01	0.98	-0.81	0.10 †
MBC	-0.19	0.76	-0.09	0.88
MBN	-0.29	0.63	-0.90	0.04 *
Fine root biomass	0.44	0.46	0.79	0.10 †

Significance of correlation, * *p-value* < 0.05 significant, † *p-value* < 0.10 weakly significant. Soil variables: POM-C– particulate organic matter C, NH_4^+– ammonium, NO_3^- – nitrate, P – phosphorus, T_s – soil temperature, θ_v – volumetric soil water content, MBC – microbial biomass C, MBN – microbial biomass N.

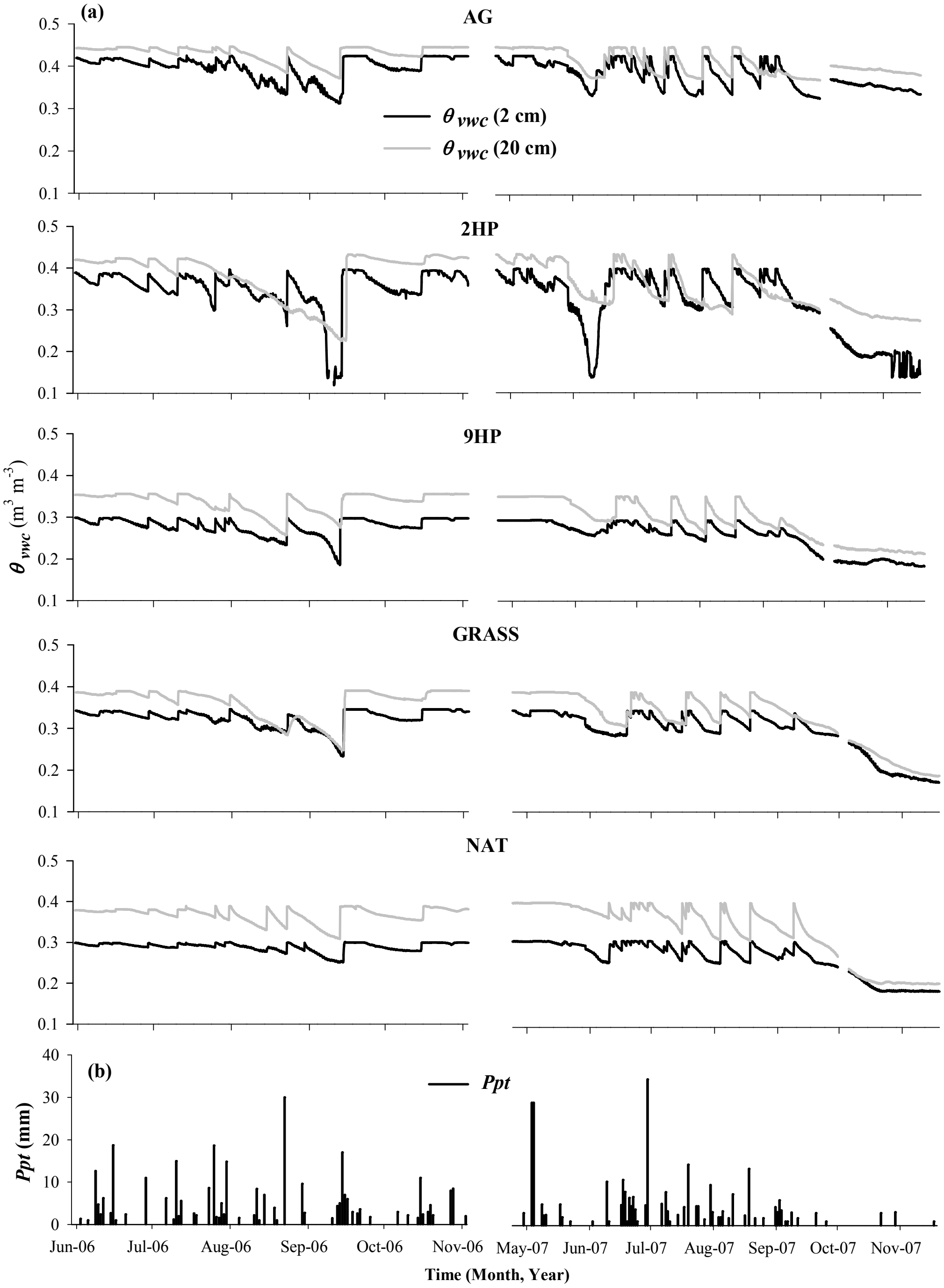

Figure A1-1. Seasonal patterns of (a) soil volumetric water content (θ_v) in various land uses and (b) precipitation (Ppt) during the two study years in Linaria, AB (AG – agriculture, 2HP – 2-yr-old hybrid poplar stand, 9HP – 9-yr-old hybrid poplar stand, GRA – grassland, NAT – native aspen stand).