QUANTIFICATION OF CARBON POOLS AND FLUXES IN RELATION TO

LAND-USE IN SUB-BOREAL BRITISH COLUMBIA

by

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Abstract

The objective of this study was to assess aboveground and belowground carbon (C) and C fluxes present in an even-aged mature forest, a regenerating forest, and a pasture in sub-Boreal British Columbia near Hixon, and in the Aleza Lake Research Forest. Net flux of carbon dioxide (CO₂) for the pasture ecosystem was measured by a combination of eddy correlation and Bowen ratio energy balance methods in 2001 for the growing season (23 May to 24 October) and measurement of belowground respiration for the non-growing season (24 October to 8 February), resulting in finding a net flux of 0.90±0.1 g C m⁻². In an 8-year-old regenerating clearcut net CO₂ flux was measured over the growing season (6 June to 28 August) using eddy correlation methods, resulting in finding a small net sink of -0.05±0.004 g C m⁻². The multiple regression of belowground respiration on soil moisture, temperature, C, nitrogen, C:N ratio, microbial biomass and root surface area resulted in a significant r^2 of 0.37. Measurement of total C pools showed that the mature forest had the highest total C (376 Mg C ha⁻¹), the 18-year-old regenerating forest was second highest (124 Mg C ha⁻¹), and the 18-year-old pasture was lowest (115 Mg C ha⁻¹) although it had the highest soil C. Analysis of an orthophoto with a novel land type classifier program (Galpin, 2002) showed that the area surrounding the Hixon study sites was 66% mature forest, 16% regenerating forest, and 13% pasture, which indicates that the maximum possible C pools are not obtained on the landscape due to land use changes. If clearing of the mature forest had not occurred there would be more aboveground C, less soil C, and higher belowground respiration.

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Table of Contents		
Abstract	•	
Table of Contents		İİİ
List of Tables		iv
List of Figures		V
Acknowledgement		vii
Chapter 1	Introduction	1
Chapter 2	A comparison of seasonal CO ₂ fluxes from sub-Boreal pastures and regenerating clearcuts	8
Chapter 3	The relationships between belowground respiration, soil organic matter C and N, soil microbial biomass, root surface area, soil temperature, and soil moisture in a sub-Boreal pasture	32
Chapter 4	A comparison of carbon pools and belowground CO_2 fluxes between pasture, regenerating forest, and mature forest in sub-Boreal British Columbia	57
Chapter 5	Land type classification and distribution of carbon pools and belowground fluxes in a sub-Boreal pasture-forest mosaic	82
Chapter 6	Summary	99

ii

List of Tables

Table 2.1.	Net ecosystem flux and mean daily flux for the pasture and regenerating clearcut sites.	22
Table 2.2.	Comparison of net ecosystem flux in the regenerating clearcut across years for the period of 27 June to 28 August.	23
Table 3.1.	Data for weekly mean statistics.	53
Table 3.2.	Data for site mean statistics.	54
Table 3.3.	Multiple regression parameters for site means.	54
Table 3.4.	Multiple regression parameters for weekly means.	55
Table 5.1	Proportions of land area of each land type and associated C pools and belowground respiration for the land type and the area represented by the land type in the orthophoto.	88

List of Figures

Figure 2.1	Daily totals of net CO_2 flux over the pasture site from May to October 2001 showing a net sink for CO_2	20	
Figure 2.2	Mean belowground respiration values for samples taken from September 2001 to February 2001 in the pasture site.	21	
Figure 2.3	Daily totals of CO_2 flux over the regenerating clearcut site from June to August 2002 showing a net sink for CO_2	22	
Figure 2.4	Example diurnal flux patterns for 11 and 12 July 2001. Fluxes measured for 11 July were variable due to partially cloudy conditions, and the nighttime fluxes were lower due to stable atmospheric conditions with low winds compared to 12 July. The drop in nighttime flux seen from 3:30 to 4:30 on 12 July was also due to low wind speeds.	26	
Figure 3.1	Linear regression of seasonal belowground respiration and soil temperature (a) showing a r^2 of 0.20 and soil moisture (b) showing a r^2 of 0.25. Data used were weekly averages of all sample points between 28 May and 31 August 2001	39	•
Figure 3.2	Linear regression of belowground respiration showing positive relationships with (a) SOC, (b) SON, (c) SMB, (d) soil moisture and (e) soil temperature. Data used were seasonal averages for each sample point.	41	
Figure 3.3	Linear regression of belowground respiration showing no relationship with root surface area.	42	
Figure 3.4	Linear regression of belowground respiration showing a negative relationship with soil C:N ratio.	42	
Figure 3.5.	Relationship between belowground respiration and soil moisture following a polynomial curve.	44	
Figure 4.1	Schematic representation of the sample site locations near Hixon, BC.	60	
Figure 4.2	SOC levels from left to right in the pasture, adjacent mature forest and regenerating forest sites for the litter laver and at depths of 0-10 cm, 10-20 cm, 20-30 cm, as		

well as total SOC including soil and litter for the site. SOC values with the same letter at each soil layer were not significantly different at the p=0.05 level using single factor ANOVA.

- Figure 4.3 Belowground respiration levels in the pasture, mature and regenerating forest sites at two-week intervals from June to August 2002.
- Figure 4.4 Aboveground biomass and total carbon measured for pasture, and adjacent mature and regenerating forest sites from left to right. Carbon pool values with the same letter for each land type were not significantly different at the p=0.05 level using single factor ANOVA.

Figure 5.2 Output image of the land types classified by the LTC. The Hixon study site is outlined.

98

65

66

67

Figure 5.1 Orthophoto image used for the analysis with the LTC program. The Hixon study site is outlined.

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Chapter 1: Introduction

Recently there has been concern over the increase in atmospheric carbon dioxide (CO₂) and the potential changes to the earth's climate that may result from this increase in CO₂ and associated warming (Tans et al., 1990; Batjes, 1998; Ramanathan, 1998). Increases in atmospheric CO₂ are estimated to total 122 \pm 40 Gt carbon (C) between 1850 and 1990 with average emissions of 1.6 \pm 1.0 Gt C y⁻¹ in the 1980s from changes in land use and 5.4 \pm 0.5 Gt C y⁻¹ due to fossil fuel emissions (Dixon et al., 1994; Schimel, 1995). Due to concerns over the effects that increases in CO₂ could have on the environment the Kyoto Protocol was signed in 1997 with the goal of reducing greenhouse gas emissions to 5.2 % below 1990 levels by between 2008 and 2012 (United Nations Framework Convention on Climate Change, 1998).

Within the Kyoto Protocol there is the potential for use of terrestrial carbon sinks that could be used to offset emission reduction targets (IPCC, 2000). It has been demonstrated that many terrestrial ecosystems can sequester C, but the extent of these C sinks has not been thoroughly quantified (Botkin, 1977; Tans et al., 1990; Mäkipää et al., 1999; Piccolol and Spaccini, 1999; Frank and Dugas, 2001). In order to quantify terrestrial C sinks, relationships between C pools and C fluxes need to be assessed both spatially and temporally over a wide range of landscapes including grasslands, forests and lands that have been altered for

human use through clearing or cultivation (Buyanovsky et al., 1987; Dixon et al., 1994; Houghton and Hackler, 2000; Frank and Dugas, 2001).

In the event of global warming, there is the potential that changes will occur in C sequestration in soils, perhaps mediated by changes in organic matter decomposition rates (Mäkipää et al., 1999). Changes in storage and decomposition of C-containing compounds along with the potential changes to photosynthetic systems due to increases in CO₂ and associated changes in climate could potentially change the overall cycling of C in terrestrial systems (Schimel, 1995; Polley, 1997). Because of the potential for a positive feedback loop between C-cycling within ecosystems and rising atmospheric CO₂, it is critical that we further understand how land-use conversion impacts C-cycling.

One of the major ways that humans impact the landscape is through the conversion of native ecosystems to managed ecosystems. In these ecosystems, the storage of C can be altered through decreases in soil organic carbon (SOC) or changes in vegetation composition (Burke et al., 1989; Römkens et al., 1999; Houghton and Hackler, 2000; Thuille et al., 2000; Conant et al., 2001; Ellert et al., 2001). Since soils store large amounts of carbon, approximately twice that contained in the atmosphere, they play a major role in C-cycling (Gill et al., 1999; Liski et al., 1999; Mielnick and Dugas, 2000; Percival et al., 2000; Conant et al., 2001). Changes in C-storage are of major concern when considering changes in atmospheric CO₂, thus processes that affect C-storage are of concern when quantifying terrestrial C fluxes and pools (Pennock and van Kessel, 1997; Price et al., 1999).

As global temperature increases it is possible that some of the C stored in soils could be released due to increased decomposition of organic matter (Schimel, 1995; Batjes, 1998; Liski et al., 1999). Certainly, soil temperature, along with soil moisture, oxygen supply, soil nutrient supply, clay content and mineralogy are known to be important controls of soil organic matter (SOM) decomposition (Batjes, 1998; Swanson et al., 2000; Percival et al., 2000). The potential for release of C from soils could significantly influence the balance between terrestrial and atmospheric C pools and change patterns in terrestrial C storage (Pennock and van Kessel, 1997; Uri, 2000).

Management of soils in cultivated landscapes may provide a means of offsetting or ameliorating losses of stored C and could even result in an overall increase in SOM-C in some systems (Mann, 1986). Use of cropping systems involving reduced tillage or no tillage, improved fertilizer management, use of cover crops, and other soil conservation methods provide a means to increase SOM-C and represent a way to potentially increase the terrestrial C sink (Batjes, 1997; Houghton and Hackler, 2000; Robertson et al., 2000; Uri, 2000; Conant et al., 2001).

Although C-storage in soils could be increased in some cases, it is questionable if increased storage in soils could balance the aboveground C lost when forests are cleared for agricultural purposes (Piccolo and Spaccini, 1999). As well as losing the aboveground C in the form of tree biomass, clearing of land and cultivation often results in loss of SOM-C both in forest and grassland

systems (Piccolo and Spaccini, 1999; Rosell et al., 2001). Eventually these systems will return to an equilibrium state proportional to the amount of residue C returned to the soil after 50 to 60 years, but initially after land use conversion there is rapid loss of C from soils (Voroney et al., 1981; Rosell et al., 2001).

It is the goal of this study to quantify C pools and fluxes in managed sub-Boreal environments in order to better understand the overall C cycling and the factors influencing C pools and fluxes in these systems. For this study, sites were chosen in and surrounding a pasture system near Hixon, BC and in a regenerating clearcut in the Aleza Lake Research Forest east of Prince George, BC.

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Chapter 2: A comparison of seasonal CO₂ fluxes from sub-Boreal pastures and regenerating clearcuts

Introduction

The debate over the ratification and implementation of the Kyoto Protocol has inspired the need for both greater understanding of carbon (C) dynamics in terrestrial ecosystems, as well as the need for quantification of the pools and fluxes that drive the C dynamics. There is uncertainty over the magnitude and locations of terrestrial C sinks in many ecosystems around the world (Malhi et al., 1999). Information is also lacking on how the C cycle functions in different ecosystems (Baldocchi et al., 1996). All aspects of C cycling need to be considered when making policy decisions related to energy consumption and land use, and an extensive measurement program is needed to quantify C fluxes and pools at a global scale (Tans et al., 1990; Baldocchi et al., 1996).

When examining C dynamics the landscape as a whole must be examined. Both natural ecosystems and those disturbed by clearing, fire or other natural phenomena must be studied due to the importance of such ecosystems in the global C cycle (Botkin, 1977; Woodwell et al., 1983; Tans et al., 1990; Dixon et al., 1994). It is also important to consider all types of ecosystems when examining C dynamics so that parts of the overall terrestrial C cycle are not overlooked (Baldocchi et al., 1996; Polley, 1997; Frank and Dugas, 2001).

Terrestrial ecosystems at high latitudes, those above 50°N, have been shown to contain large quantities of C (Dixon et al., 1994; Schimel, 1995). Many

of the studies to date that include high latitude ecosystems have focused on Boreal environments due to the large amounts of C stored in them and their high potential for C sequestration and loss (Woodwell et al., 1983; Mäkipää et al., 1999; Price et al., 1999; Harding et al., 2001). Sub-Boreal ecosystems in British Columbia have the potential to store and sequester large amounts of C due to milder temperatures and increased moisture compared to Boreal ecosystems in other parts of Canada, which have received more attention. At the same time, sub-Boreal forests experience lower rates of decomposition compared to midlatitude and low-latitude ecosystems, thus increasing their relative potential for C sequestration (Meidinger and Pojar, 1991; Dixon et al., 1994).

When examining the potential of ecosystems to sequester C, the sign and magnitude of C fluxes are examined, but the results obtained regarding C fluxes are often conflicting (Schimel, 1995; Thornley and Cannell, 1997; Lloyd, 1999). The results from studies and models depend on the ecosystem being examined, the location of the ecosystem, the data used for calculation of fluxes and the assumptions used in the study (Schimel, 1995). In forested ecosystems some have been shown to be a net source of C (Dixon et al., 1994; Goulden et al., 1996; Mäkipää et al., 1999), while the majority have been shown to be a net sink for C (Malhi et al., 1999; Bateman and Lovett, 2000; Newell and Stravins, 2000; Thuille et al., 2000; Read et al., 2001; Robert, 2001). Conflicting results have also been found in grassland and pasture ecosystems where some ecosystems have been shown to be net sources of C (Thornley and Cannell, 1997; Kucharik

et al., 2001), and others have been shown to be net sinks (Robertson et al., 2000; Frank and Dugas, 2001). It is also important to note that few of the studies performed in grassland ecosystems have included grazing as part of the experimental treatment (Campbell and Stafford-Smith, 2000). Due to conflicting results about the sign and magnitude of C sources and sinks it is difficult to generalize across broad geographic areas, further indicating the need to examine specific landscape and ecosystem types under a greater variety of climatic and edaphic conditions.

In order to better understand the fluxes of C and the potential for sequestration, it is important to examine how C fluxes contribute to C dynamics. Sequestration of C depends on the balance between photosynthesis and respiration in an ecosystem (Robertson et al., 2000). The photosynthesis component of this balance is the uptake and assimilation of CO_2 by plants (Polley, 1997). The respiration component is due to the cellular activity of plant components and decomposition of organic matter by soil biota resulting in the release of CO_2 (Duenas et al., 1995; Polley, 1997; Grogan and Chapin, 1999; Mielnick and Dugas, 1999). If the uptake of CO_2 by photosynthesis is greater than the release of CO_2 through respiration by plants and soil biota, the ecosystem is a net sink for C. If the opposite situation occurs, with a greater release of CO_2 than uptake of CO_2 , the ecosystem is a net source for C. The balance between these processes is controlled by many factors including available photosynthetically active radiation, air and soil temperatures, soil

moisture, available nutrients and the composition of the soil biotic community (Woodwell et al., 1983; Solomon and Cerling, 1987; Schimel, 1995; Mäkipää et al., 1999; Malhi et al., 1999).

To completely account for all the fluxes due to photosynthesis and respiration, measurements need to be made over the whole year (Solomon and Cerling, 1987; Schimel, 1995). In the past, studies have focused on growing season fluxes, which include both photosynthesis and respiration (Mielnick and Dugas, 2000). Fewer studies have measured non-growing season fluxes, which often have an increased dependence on respiration and are typically more difficult to quantify (Baldocchi et al., 1996; Brooks et al., 1997; Grogan and Chapin, 1999; Frank and Dugas, 2001) as a result of variability in response to availability of nutrients, soil moisture, soil temperature, timing and depth of snowfall, type of microorganisms present, and regional differences in climate (Ivarson and Sowden, 1966; Brooks et al., 1997; Lomander et al., 1998; Grogan and Chapin, 1999).

In this study our objective was to quantify C fluxes for two disturbed ecosystems in the Sub-Boreal Spruce biogeoclimatic zone of British Columbia, a pasture south of Prince George and regenerating clear-cut east of Prince George, both on similar soil types. We anticipated that the pasture site would be a net sink or near equilibrium for CO_2 and could represent a potential means of sequestering atmospheric CO_2 , while the regenerating clear-cut was anticipated

to be a net source for CO₂ based on the assumption in many models that clearcuts remain sources until approximately ten years after harvest (Kurtz and Apps, 1994).

Methods

Site description

The pasture site was located in central British Columbia, south of Hixon, BC (53°20'N, 122°35'W) in the Sub-Boreal Spruce dry warm (SBSdw) biogeoclimatic subzone (Meidinger and Pojar, 1991). The site, dominated by lodgepole pine (*Pinus contorta* var. *latifolia*), was cleared and pile-burned 18 years prior to measurement (1985) and was subsequently seeded with timothy grass (*Phleum pratense*). Other species present in the pasture include buttercup (*Ranunculus acris*), clover (*Trifolium hybridum*), orange hawkweed (*Hieracium aurantiacum*), other grass and sedge species (*Bromus* ssp., *Festuca* ssp., *Poa* ssp., *Carex* ssp.). The pasture was grazed for portions of the 5 years before the study was conducted. The soils on the site were predominantly Dark Gray Luvisols, with Gleyed Dark Gray Luvisols and Humic Luvic Gleysols in low-lying areas with clay or heavy clay textures.

The regenerating clearcut site was the same as that used by Pypker and Fredeen (2002) located in the University of Northern British Columbia/University of British Columbia Aleza Lake Research Forest (54°01'30"N, 122°07'30"W). In brief, this site was within an 84.15 ha 1994 clearcut of sub-Boreal spruce-fir

forest (SBSwk biogeoclimatic zone) planted with 2-year-old hybrid white spruce and lodgepole pine after mounding in 1995. Soils on the clearcut site were largely classified as Orthic Luvic Gleysols (Arocena and Sanborn, 1999).

Bowen ratio energy balance (BREB) method

BREB measurements were made on the pasture site from 23 May until 24 June 2001 using a Bowen Ratio Energy Balance system (023/CO₂, Campbell Scientific, Edmonton, AB) and an infrared gas analyzer (IRGA) (LI-6262, LI-Cor. Inc., Lincoln, NE) placed on a 3.2 m tall tower located so that there was 250 m of fetch in all directions. The IRGA was calibrated 3 times per week and is accurate to ±1 ppm. The top arm of the BREB system was mounted at 2.8 m and the lower arm was mounted 1.4 m below, well above the height of the grass canopy (maximum grass height was approximately 0.7 m). Net radiation (Q*) was measured using a Q7 net radiometer, accurate within 6% of the measurement, (Campbell Scientific) mounted 3.0 m from ground level. Air temperature was measured using 75 µm chromel-constantan thermocouples mounted at the end of each arm. Soil heat flux (G) was measured using two soil heat flux plates placed 8 cm below the ground surface (HFT-3, Campbell Scientific), which were accurate within 5% of the measurement. Soil temperature was measured using four soil thermocouples (TCAV, Campbell Scientific) placed in pairs at 2 and 6 cm depths, as well as at 10 cm at three additional locations. Wind speed and direction were also measured at 3.2 m (R.M. Young, Wind Sentry, Campbell Scientific) and relative humidity was measured at 2.1 m, accurate within ±2%RH

(HMP35C, Campbell Scientific). Rainfall was measured using a tipping rain gauge, accurate within ±1% of the measurement (TE525WS, Campbell Scientific). All data were averaged over 20 min intervals and were stored on two dataloggers, a 23X (Campbell Scientific) for the majority of the measurements and a 21X (Campbell Scientific) for the soil thermocouples placed at 10 cm depth and the rain gauge.

The BREB method is based upon energy balance at the surface represented by the simplified energy balance equation:

$$Q^* = H + LE + G \tag{1}$$

Where Q^* is the net radiation, *H* is the sensible heat flux, LE is the latent heat flux, and *G* is the soil heat flux, all expressed in W m⁻². To calculate the values of the variables in the energy balance equation the computer program SPLIT (PC208W Software, Campbell Scientific) was used. Soil heat flux was calculated first using the heat capacity of moist soil, c_s (J m⁻³ K⁻¹):

$$c_s = \rho_b(c_d + \theta_m c_w) \tag{2}$$

where ρ_b (kg m⁻³) is the bulk density of the dry soil, c_d (J kg⁻¹ K⁻¹) is the specific heat of dry mineral soil, θ_m (dimensionless) is the fraction water content of the soil on a mass basis, and c_w (J kg⁻¹ K⁻¹) is the specific heat of water. The rate of energy storage between the thermocouples belowground (*S*) (W m⁻²) over each 20 min interval was calculated:

$$S = \Delta T_s c_s d \tag{3}$$

t

where ΔT_s (K) is the average change in soil temperature at the two thermocouples, *d* (m) is the distance between the sensors and *t* is the time over which the change occurred. Values for *G* were then calculated by summing the heat flux values from the heat flux plates (*G_g*) (W m⁻²) and *S*:

$$G = G_g + S \tag{4}$$

Calculation of *H*, LE, and the flux density for CO_2 (*F_c*) were based on the following equations:

$$LE = -L_v K_E \delta \rho_v \tag{5}$$

$$H = -\rho C_{\rho} K_{H} \delta T$$

$$\delta z$$

$$F_{c} = -K_{c} \delta \rho_{c}$$

$$\delta z$$
(6)
(7)

where K_E , K_H , and K_C are the eddy diffusivities for water vapour, heat and CO₂ respectively; ρ_v , ρ_c , and ρ are the densities (kg m⁻³) for water vapour, CO₂, and air respectively; C_ρ (J kg⁻¹ K⁻¹) is the heat capacity of air; L_v (J kg⁻¹) is the latent heat of vaporization; T (K) is the air temperature; z (m) is height; and δ is a partial derivative of ∂ .

Equations 5 and 7 can be transformed using the universal gas law to:

$$LE = -L_{v}K_{E}PM_{W}\delta W$$
(8)

TRδz

$$F_{c} = -Kc \underline{PM_{c}\delta C}$$
$$TR\delta z$$

Where *P* (kPa) is total atmospheric pressure M_w (g mol⁻¹) is the molecular weight of water, *W* is the mole fraction of water to air, *R* (J kg⁻¹ mol⁻¹) is the universal gas constant, M_c (g mol⁻¹) is the molecular weight of CO₂, and *C* is the mole fraction of CO₂ to air.

(9)

Using equation 1 and the similarity assumption ($K_H = K_E = K_c$) (Denmead and Raupach, 1993) we calculate the Bowen ratio, which is defined as the ratio of sensible to latent energy:

$$\beta = H = \frac{C_p \delta T}{LE}$$
(10)
LE $L_v \epsilon \delta W$

where δT (K) is the temperature gradient and ϵ is the ratio of molecular weights of water vapour to dry air. Combining equation 10 with equation 1 to eliminate LE gives:

$$H = \beta(Q^* - G) \tag{11}$$

The eddy diffusivity of CO_2 was estimated by rewriting equation 6 with *H* from equation 11 to give:

$$K_{c} = K_{H} = (z_{1} - z_{2})H$$
(12)
$$\overline{(T_{1} - T_{2})\rho C_{\rho}}$$

Equation 9 was then used to calculate the CO_2 flux and corrected using the equations presented in Webb et al. (1980).

Eddy correlation/covariance (EC) method

From 8 June to 24 October 2001 EC measurements were made on the pasture site and from 6 June to 28 August 2002 the EC system was moved to the regenerating clearcut site. CO₂ fluxes were measured using an open path IRGA, accurate within ±0.3 µmol mol⁻¹, which was calibrated at the beginning of the field season (LI-7500, LI-Cor Inc.) mounted at 2.2 m on a 2.9 m tall tower. Wind speed and direction were measured in three dimensions using a sonic anemometer, accurate to ±4 cm s⁻¹ (CSAT3, Campbell Scientific) mounted at 2.7 m. Air temperature was measured using a 1.27 µm chromel-constantan thermocouple mounted on the sonic anemometer. Data was averaged over 15 min intervals and stored on a datalogger (23X, Campbell Scientific). In 2001 both BREB and EC systems were placed in the pasture site in an 8.5 by 3.5 m enclosure to prevent grazing cattle from damaging the instrumentation and interfering with measurements. The site had a minimum of 450 m fetch in all directions, which is well within the distance needed for equilibrium to be reached in the boundary layer (Gash, 1986). In 2002 the IRGA was mounted at 2.4 m and the sonic anemometer and thermocouple were mounted at 2.8 m on a 2.9 m tall tower located a minimum of 300 m from the forest edge.

The EC method is based on measurement of gas flux densities, in this case CO_2 . The mean flux density for CO_2 (F_c) is given by:

$$F_{c} = \overline{w'\rho_{c}}'$$
(13)

Where ρ_c' (kg m⁻³) is the density of CO₂, and *w'* (m s⁻¹) is the vertical component of the wind speed as it fluctuates from their mean values. Both of these variables are measured directly using the EC method and F_c is calculated within the datalogger program (Campbell Scientific) using equation 13. All interfacing and downloading with the dataloggers was performed using the PC208W 3.1 software (Campbell Scientific), communications directly with the sonic anemometer used the CSAT32 software (Campbell Scientific), and communication with the open path IRGA used the LI7500 software (LI-Cor Inc.).

Non-growing season belowground respiration measurements

On the pasture site, belowground respiration was measured from 20 September 2001 to 8 February 2002 using a portable closed gas exchange system (LI-6200, LI-Cor Inc) with the soil chamber attachment (6000-09, LI-Cor Inc), accurate within ±0.3 ppm and calibrated daily when in use. Four soil collars placed within the tower enclosure were measured once weekly as weather permitted until 11 January 2002 when freeze-thaw events resulted in large areas of the soil surface covered in ice. Samples were then taken in the ice-free areas of the pasture until 8 February 2002 when further freeze-thaw events resulted in coverage of the entire surface of the field with ice. Total belowground respiration for the non-growing season was calculated using the mean belowground respiration values for the period and the number of days above freezing temperatures. It was assumed that the belowground respiration values for the days with below freezing temperatures were zero. This was a valid assumption because of the minimal snow cover in 2002 and the soil freezing that occurred.

Data processing

Data were lost from the BREB and EC measurements due to low wind speed, precipitation events, or changes in boundary layer stability, such as at dawn and dusk (Ham and Knapp, 1998). Gaps in the data set due to these omissions and from equipment failures were modelled using a combination of extrapolation from surrounding measurements for small gaps (less than 3 hours) and a regression between CO_2 flux and solar radiation during the day and mean night time respiration values from previous and following nights for larger gaps based on the method in Ham and Knapp (1998). Net growing season flux error estimates were based on the range of data between the maximum and minimum values observed and the mean net flux error was based on the standard error of the estimate. Closure of the energy budget for the pasture site was performed using data from the BREB system with a linear regression of the eddy fluxes (LE + H) and the available energy ($R_n - G$) based on Hunt et al. (2002).

Results

Pasture site

Net ecosystem flux from the pasture site as measured by the BREB and EC method for 2001 showed a small net growing-season sink of approximately -0.033±0.002 g C m⁻² and a mean net flux of -0.21±0.03 mg C m⁻² d⁻¹ (Table 2.1). Peak rates of CO₂ uptake were seen in July and early August, and peak rates of CO₂ release were seen in May and October with some high values intermittently during August and September (Figure 2.1). Energy budget measurements showed that the eddy fluxes agreed with the available radiation (R_a = 0.90(H + LE) – 25 (W m⁻²), r² = 0.80, n = 2417).



Figure 2.1. Daily totals of net CO_2 flux over the pasture site from May to October 2001 showing a net sink for CO_2 .

Data obtained from measurement of belowground respiration during the non-growing season showed higher respiration rates in September and early October, and lower respiration rates later in the season (Figure 2.2). Modelling the non-growing season fluxes for the interval between the end of micrometeorological measurements and ice-over of the pasture using freeze or thaw conditions based on air temperature measurements showed a net source of 32.1 ± 0.1 g C m⁻² for that period with 69 days of the 101 day sampling period under freezing conditions.



Figure 2.2. Mean belowground respiration values for samples taken from September 2001 to February 2002 in the pasture site. Data from the end of October 2001 to February 2002 was used to model non-growing season fluxes for the pasture site.

Table 2.1. Net ecosystem flux and mean daily flux for the pasture and regenerating clearcut sites over the growing season.

Site	Growing Season Ecosystem Flux	Daily Flux Range	Mean Daily Flux	Standard Error
	(g C m ⁻²)	-	(mg C m ⁻² d ⁻¹)	
Pasture	-0.033	0.003	-0.212	0.038
Regenerating clearcut	-0.049	0.001	-0.590	0.027

Regenerating clearcut site

Net ecosystem flux as measured by the EC method for the 2002 growing season showed the regenerating clearcut was a small sink of approximately -0.05 ± 0.001 g C m⁻² with a mean daily flux of -0.6 ± 0.03 mg C m⁻² d⁻¹ (Table 2.1). Peak rates of CO₂ uptake were seen in mid-July with periods of high uptake in mid-June and early August (Figure 2.3).



Eddy Covariance Modelled Data — EC Moving Average

Figure 2.3. Daily totals of CO_2 flux over the regenerating clearcut site from June to August 2002 showing a net sink for CO_2 .

Discussion

The net source of 32.0±0.1 g C m⁻² observed for the pasture site disagrees with what has been found in previous studies. Other sites used for agricultural purposes have been found to be net sinks for CO₂ in temperate regions (e.g. Frank and Dugas, 2001). Previous studies have mainly focused on intensively managed natural grassland ecosystems that lack the influence of grazing and often sampling only occurs over the growing season, which could explain the difference between the net sink observed in these studies and our results (Thornley and Cannel, 1997; Batjes, 1998; Campbell and Stafford-Smith, 2000).

In the regenerating clearcut the results we found agree with those previously observed on the site in 1999 by Pypker and Fredeen (2002a). The observations used by Pypker and Fredeen (2002a) were for a similar observation period of 27 June to 3 September 1999 resulting in a net sink of –22.4 g C m⁻² compared to this study with measurements from 6 June to 28 August 2002

resulting in a net sink of -0.05 g C m⁻², but the results for the site from the 2000 growing season (over a longer time period of 24 May to 20 September 2000) showed the site to be a net source of 142 g C m⁻² (Pypker and Fredeen, 2002b). A comparison of data from all three years over the common interval of

Table 2.2. Comparisonof net ecosystem flux inthe regenerating clearcutacross years for theperiod of 27 June to 28August.

Net ecosystem flux	4
(g C m ⁻²)	
-31.1	
56.6	
-0.0365	
	Net ecosystem flux (g C m ⁻²) -31.1 56.6 -0.0365

27 June to 28 August showed a similar pattern to the whole growing season fluxes (Table 2.2). The variation observed between sink and source status of the regenerating clearcut are most likely due to changing environmental conditions between years (Pypker and Fredeen, 2002b). Over time a trend towards a C sink in the regenerating clearcut should occur, with observations of net sink status occurring after approximately ten years of growth (Kurtz and Apps, 1994). Increases in biomass of growing trees over time should facilitate this trend towards a C sink due to increases in photosynthetic plant material as trees grow.

There is some question about the accuracy of the micrometeorological methods used to determine the net ecosystem fluxes on each of the sites. The BREB method has been shown to have problems accurately estimating fluxes during periods of stable atmospheric conditions resulting in thermal stratification, often at night, and during periods where wind speed is low (Perez et al., 1999; Polonio and Soler, 2000; Frank and Dugas, 2001). In these cases the diffusivities for heat and water vapour may not be equal or the fluxes can become very small and difficult to measure (Frank and Dugas, 2001). Gaps in the data for fluxes were caused by rainfall events and equipment failures for both the BREB and EC methods. The EC method has also been shown to have problems with thermal stratification due to stable atmospheric conditions and functions best during windy periods (Baldocchi et al., 1996; Goulden et al., 1996; Malhi et al., 1999). There is also the potential for errors to occur in the measurement of turbulent velocities due to interference with normal flow by the instrumentation

and supporting structures (Baldocchi et al., 1988). Partially cloudy conditions can also result in a mosaic of sunlit and shaded patches over the landscape causing the fluxes over the landscape to be highly variable in space (Baldocchi et al., 1996). These errors in measurement of the fluxes may have resulted in an underestimation of nighttime fluxes due to inaccurate data or loss of data during those stable atmospheric periods (Goulden et al., 1996) (Figure 2.4).

Both sites were found to be small sinks for CO_2 during the growing season (23 May to 24 October 2001 for the pasture) and (6 June to 28 August 2002 for the regenerating clearcut). There is the possibility that winter respiration from belowground sources could change the magnitude and sign of the net ecosystem flux in the regenerating clearcut site. Measurement of the belowground respiration over the winter in the pasture site changed the sign and magnitude of the net ecosystem flux from a sink to a source of CO_2 of 32.0 ± 0.1 g C m⁻² for the year. Errors in measurement of the belowground respiration due to difficulties in creating a good seal between the soil chamber and the frozen soil could result in overestimation of the fluxes due to leakage into the chamber from the atmosphere resulting in magnified increases in CO_2 concentration over time.

The belowground respiration observed in the pasture site follows patterns that have been previously observed in other studies. Specifically, many have observed similar patterns of CO_2 release from soils, with higher values of belowground respiration as soil temperatures initially cool, but decreasing rapidly



Figure 2.4. Example diurnal flux patterns for 11 and 12 July 2001. Fluxes measured for 11 July were variable due to partially cloudy conditions, and the nighttime fluxes were lower due to stable atmospheric conditions with low winds compared to 12 July. The drop in nighttime flux seen from 3:30 to 4:30 on 12 July was also due to low wind speeds.

as soils begin to freeze (Solomon and Cerling, 1987; Winston et al., 1995;

Lomander et al., 1998). Once soils reach a minimum temperature of between -4

and -7°C little or no heterotrophic activity of soil microorganisms is detected in

most soils (Winston et al., 1995; Lomander et al., 1998; Grogan and Chapin,

1999). The decrease of belowground respiration observed in late October and

throughout the rest of the sampling period corresponds to the freezing of the soil,

especially where little snow cover was present. The lack of the insulating

properties of a snowpack has been shown to prevent heterotrophic activity during

the winter (Brooks et al., 1997).

The results of this study indicate that the conversion of natural forests to forest plantations does not necessarily result in creation of carbon sources over the growing season, especially after the ecosystem has had time to reach a new equilibrium carbon balance or recover to a previous equilibrium carbon balance. In the case of conversion of forests to non-forest use, such as pasture, the site was shown to be a net source for CO₂ for this period. It is also important to note that although the regenerating clearcut site showed a small net sink, the inclusion of winter respiration measurements could alter the sink status of the site. With time cleared forest sites have the potential to regain their sink status, but the loss of biomass C found in trees is permanent after conversion to non-forest use such as pastures, and it takes a long time to return to the high levels of biomass C found in mature forests from regenerating clearcuts. Overall there is a high potential for loss of C with conversion of forest ecosystems to other land uses, both through changes in C fluxes and standing biomass. These considerations have to be taken into account when determining the C balance of any landscape and have implications for land use management in terms of C sequestration in terrestrial ecosystems.

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<u>Chapter 3: The relationships between belowground respiration, soil organic</u> <u>matter C and N, soil microbial biomass, root surface area, soil temperature,</u> <u>and soil moisture in a sub-Boreal pasture</u>

Introduction

In many terrestrial systems much of the carbon in the system is found in soils (Gill et al., 1999). Globally, soils also contain approximately two times as much carbon as the atmosphere, making them especially important when considering carbon cycling (Liski et al., 1999; Mielnick and Dugas, 2000; Ellert et al., 2001). The large amounts of carbon stored in soils have the potential to release large quantities of CO₂ to the atmosphere, especially in response to changes in climate (Batjes, 1998), land-use (Thuille et al., 2000), and associated microclimate (Liski et al., 1999). The release of CO₂ following land-use change such as deforestation results from physical disturbance of soil organic matter decomposition of tree and plant roots, and attendant shifts in belowground conditions such as temperature and moisture (Fearnside and Barbosa, 1998; Ellert et al., 2001).

When land-use is altered or natural vegetation removed, changes in root respiration and decomposition of organic matter by soil microorganisms occur (Pongracic et al., 1997; Janssens et al., 2000; Mielnick and Dugas, 2000). Measurement of this belowground CO₂ efflux is critical for the accurate determination of ecosystem C pools and dynamics of C-cycling (Janssens et al., 2000). When an ecosystem is disturbed it usually becomes a source for CO₂, but if consistent management practices are observed, the system is able to reach a

new steady-state (Batjes, 1998; Ellert et al., 2001). Some management practices have been developed for agricultural soils that are designed to increase sequestration of carbon in the soil carbon pool (Batjes, 1998; Uri, 2000).

Because root systems are a major influence on belowground respiration through cellular activity and also through inputs of soil organic matter (SOM) through decomposition of plant parts, it is important to consider them when examining carbon pools and fluxes (Paul and Clark, 1996; Pongracic et al., 1997; Silver and Miya, 2001). The distribution and structure of root systems in the soil is essential for the assessment of C-cycling through the plant-soil-atmosphere system (Al-Khafaf et al., 1977). The patterns and magnitude of CO_2 efflux from roots differs between species, largely as a result of rooting pattern differences (Ben-Asher et al., 1994). Root systems have been shown to contribute up to 30% of soil respiration in grassland ecosystems with the remainder coming from SOM decomposition (Buyanovsky et al., 1987). The control of CO_2 efflux from roots is dependent on a combination of factors including soil temperature, soil moisture, root chemical composition and net radiation flux (Pongracic et al., 1997; Fitter et al., 1998; Kelliher et al., 1999; Silver and Miya, 2001).

The amount of carbon present in the soil carbon pool is determined by a balance between organic inputs and organic matter decomposition (Uri, 2000). The balance between these two processes is regulated by soil microbial biomass (SMB) through the process of decomposition (Howarth and Paul, 1994; Beyer et

al., 1999). This interaction between SMB and soil organic matter (SOM) is complex and results in the regulation of nutrient cycling, soil fertility, and SOM turnover (Howarth and Paul, 1994; Turco et al., 1994; Franzluebbers et al., 1996; He et al., 1997; Blagodatsky et al., 2000; Haney et al., 2001).

As organic matter is decomposed by the active SMB pool, CO₂ is evolved through microbial respiration (Duenas et al., 1995). Due to the rapid cycling of SOM through SMB pools, small changes in SOM can be observed using SMB as an indicator (Piao et al., 2001). A change in land management resulting in changes to C inputs, such as cultivation, can result in major changes in both the SMB community structure and activity, which could in turn change the cycling of SOM in the system (Voroney et al., 1981; Bossio et al., 1998; Dilly and Munch, 1998; Ananyeva et al., 1999).

The chemical and physical composition of the SOM as a substrate for SMB, especially carbon (C), nitrogen (N), and phosphorus content, determine the decomposition rate of available substrate (McGill and Cole, 1981; Paul and Clark, 1996; Hyvonen et al., 1998; Lomander et al., 1998; Broughton and Gross, 2000; Wright and Reddy, 2001). Because of the need for available substrates, the SMB is not evenly distributed throughout the soil and populations have a tendency to be concentrated in the rhizosphere; the community present in each rhizosphere is dependent on the plant community (Turco et al., 1994; Liang et al., 1998; Ohtonen and Vare, 1998; Broughton and Gross, 2000; Piao et al., 2001).

Portions of the total C and N pools can be lost if soil is exposed through clearing or cultivation, but C and N can be sequestered if soil is not disturbed and the plant community deposits organic matter to act as a substrate for SMB (Buyanovsky et al., 1987; Sparling et al., 1994; Fearnside and Barbosa, 1998). The balance between C and N losses and gains from the SOM pools can be altered through agricultural practices and land-use and therefore has to be considered when determining management decisions (Uri, 2000).

Accumulation or loss of soil C and N is also influenced by soil moisture and soil temperature. Soil moisture content determines amount of water-filled pore spaces and through this the oxygen conditions for SMB and roots (Paul and Clark, 1996). If soil moisture is high then most of the pores are filled and anoxic conditions result and anaerobic processes are dominant, if soil moisture is low there is little SMB activity due to desiccation of the organisms (McGill et al., 1981). Soils are a combination of both aerobic and anaerobic conditions due to variability of soil moisture conditions in space and time, which can result in varied soil respiration across a small area (Paul and Clark, 1996).

Soil organisms, including SMB, work at optimal soil temperatures, and the optimum is dependent on species (Paul and Clark, 1996). Activity of organisms found in soils has been seen between –12°C and 45°C with lower activities seen at the extremes of this range and optimal activities around 30°C (Paul and Clark, 1996). Soil temperature and moisture conditions are related since increased

temperatures result in increased evaporation of water and decreased soil moisture. Such interactions make it difficult to entirely separate the effects of environmental properties on soil respiration in field conditions (Paul and Clark, 1996).

There is a need to examine the relationships associated with belowground respiration and other soil properties as part of C-cycling over many different land types in order to better understand these relationships (Halldin et al., 1999). Several ecosystems have become increasingly of interest in C-cycling, including those in Boreal regions. Boreal ecosystems have large C pools present which account for approximately 15 % of the world's terrestrial C (Harding et al., 2001; Banfield et al., 2002; Falge et al., 2002; Lee et al., 2002). The large amounts of C stored in Boreal ecosystems have a high potential to release C as climate changes due to alterations of temperature and growing season length (Billings et al., 1998; Rayment and Jarvis, 2000; Harding et al., 2001; Falge et al., 2002). In Boreal ecosystems carbon is also released through both natural and anthropogenic disturbance and changes in the disturbance through land management need to be quantified across all land types (Halldin et al., 1999; Bhatti et al., 2002; Lee et al., 2002). There is also a need for greater understanding of C dynamics within these systems to fully understand the cycling of C and how changes to climate could affect C pools and fluxes. The purpose of this study was to determine the relation of belowground respiration to soil organic matter C and N, SMB, root surface area, soil temperature, and soil moisture in a

sub-Boreal pasture soil in order to asses the influences of environmental and biological variables on belowground CO₂ fluxes.

Methods

Soil respiration, temperature and moisture

Soil properties were measured at thirty randomly selected points from 28 May to 31 August 2001 in a pasture site near Hixon, BC (Chapter 2). At these points soil respiration and soil temperature were measured using an LI-6200 portable photosynthesis system with an LI-6000-09 soil respiration chamber (LI-COR Inc., Lincoln, NE), accurate within ± 0.3 ppm and calibrated daily when in use. Soil moisture was determined at each respiration sampling point using a Hydrosense soil moisture probe, accurate within $\pm 3\%$ (CS620, Campbell Scientific, Logan, UT).

Soil organic matter

Soil samples from each point were also taken to a depth of 10 cm during the first week of July 2001 for determination of soil organic carbon (SOC), and soil organic nitrogen (SON). Representative sub-samples of the soil samples were taken and oven dried at 105 °C for 48 hours and finely ground using mortar and pestle for elemental analysis. Both SOC and SON were determined using elemental analysis with a NA 1500 Elemental Analyzer (Fissions Instruments SPA., Milano, Italy). The SOC and SON results were corrected for equal mass of the soil following the method outlined in Ellert et al. (2001) (see Appendix B for example).

Root surface area

Soil cores 23 cm in length and 5 cm diameter were also obtained during the first week of July 2001 at each soil respiration sampling point for determination of root properties. Root-containing soil was washed through a 0.7 mm soil sieve to obtain roots and the roots were soaked overnight in water so that they were saturated. The washed root samples were then weighed and 10% of the sample was selected for digital analysis. Roots were then placed in 100 mL of water with 30 mL of acetic acid and 10 mL of purple food colouring so that they were visible when scanned against a white background. Dyed roots were then isolated in water filled trays and scanned digitally for surface area determination (See Appendix A for calculations) (Delta – T Devices Ltd., Cambridge, UK).

Soil microbial biomass

Sub-samples of the fresh soil taken for SOC and SON determination were taken from the upper 10 cm of soil at twenty-five of the selected points on the pasture. The portion of the soil sample used for SMB determination was kept at field moist conditions and aggregates were broken up in preparation for chloroform fumigation-incubation (CFI) biomass determination. Soil moisture for the samples used for CFI was also determined by oven drying a portion of the samples.

For the CFI biomass (Jenkinson and Powlson, 1976a, 1976b), two 25 g soil samples were used, one fumigated and the other not fumigated. After fumigation, the soil samples and three blanks were incubated with 25 mL of 0.25 M NaOH for six days. Ten mL of NaOH and 1 mL of BaCl₂ were then titrated using standardized 0.25 M HCl. The CO₂-C and SMB were calculated using the amount of HCl needed to neutralize the NaOH and the equivalent dry mass of soil (see Appendix A for sample calculations).

Data analysis

Single factor and multiple regression analyses were performed between SOC, SON, SMB, soil moisture, soil temperature, root surface area, and soil respiration to determine the contribution of each variable on soil respiration. Data were analyzed both using the mean data for all sample points in the pasture each week of sampling, and using the mean data for each sample point over the whole sampling period.

Results

Linear regression of belowground respiration, both heterotrophic and autotrophic, from the mean weekly data of all points in the pasture with soil moisture and soil temperature resulted in r^2 values of 0.20 and 0.25, respectively which were not significant (p = 0.05). Multiple regressions of the two variables

and belowground respiration resulted in an r^2 value of 0.48, which was significant (p = 0.05). Patterns of belowground respiration using the mean weekly values show that as soil temperature increases, so does respiration, while as soil moisture increases, belowground respiration decreases (Figure 3.1).

Patterns of belowground respiration were slightly different when examined using the sample point means for each sampling point in the pasture from when soil samples were taken in early July. Belowground respiration increased with SOC ($r^2 = 0.06$), SON ($r^2 = 0.13$), SMB ($r^2 = 0.08$), soil moisture ($r^2 = 0.02$), and soil temperature ($r^2 = 0.10$) (Figure 3.2). No relationship was seen between belowground respiration root surface area ($r^2 = 0.005$) (Figure 3.3). A negative relationship was seen between belowground respiration and soil C:N ratio ($r^2 = 0.005$)



Figure 3.1. Linear regression of seasonal belowground respiration and soil temperature (a) showing an r^2 of 0.20 and soil moisture (b) showing a r^2 of 0.25. Data used were weekly averages of all sample points between 28 May and 31 August 2001

0.07) (Figure 3.4). None of the single factor regressions showed significant relationships, but in multiple regressions using all of the above parameters resulted in an overall r^2 value of 0.37, which explained a significant amount of variation in belowground respiration (p = 0.05). From the regressions, although they were not significant, it is apparent that the major controls of belowground respiration of the variables examined in this study are soil N content and soil temperature.







Discussion

The importance of temperature on regulation of soil processes, such as belowground respiration has been demonstrated in many soil experiments (Thornley and Cannell, 1997; Fitter et al., 1998; Knorr, 2000; Jones and Cox, 2001). In general there is a positive correlation between temperature and belowground respiration due to the warming of the soil creating conditions that are closer to optimal for decomposing organisms (Paul and Clark, 1996; Thornley and Cannell, 1997; Kudeyarov and Kurganova, 1998; Thuille et al., 2000; Silver and Miya, 2001). Strong relationships have also been seen between net radiation flux, which controls temperature, and belowground respiration (Fitter et al., 1998). Although soil temperature was found to have the most influence on









belowground respiration in this study, no relationship was seen between belowground respiration and net radiation, possibly due to lack of data caused by equipment failures.

It has also been recognized that although soil moisture has a weaker relationship with belowground respiration than soil temperature, a minimum level of moisture in the soil is required for respiration to occur (Kudeyarov and Kurganova, 1998; Knorr, 2000). High levels of moisture have also been found to decrease belowground respiration (Kudeyarov and Kurganova, 1998), most likely due to the reduction of aerobic environments through filling of soil pores. At optimal soil moisture conditions of approximately 60% water-filled pore space (WFPS), belowground respiration has been found to be highest (Paul and Clark, 1996; Kudeyarov and Kurganova, 1998). The correlations seen between soil moisture and belowground respiration have been either positive or negative, but this is dependent on soil type (Kudeyarov and Kurganova, 1998; Silver and Miya, 2001). Our results agree with a weak positive, but not significant relationship between soil moisture and belowground respiration (Figures 3.1 and 3.2), which is similar to the relationship found in temperate Gray Luvisols by Kudeyarov and Kurganova (1998). Due to the optimal soil moisture conditions of 60% WFPS, it would be expected that belowground respiration would be highest under these conditions (Scott et al., 1996). If a polynomial curve is fitted to the site mean data, as opposed to the linear relationship used in the regressions, to account for a maximum respiration at mid-range soil moisture levels a much higher r² value

of 0.23 is obtained (Figure 3.5). Applying a non-linear relationship to the weekly mean data did not show an increase in r^2 .

Soil nutrient availability, in the form of C and N, influences the respiratory activity of both SMB and roots (Paul and Clark, 1996; Fitter et al., 1998). In soils where there are higher concentrations of





C and N there is more SOM, thus more substrate for SMB and roots to utilize. Carbon and nitrogen are both major components of SOM and the two elements are directly bonded to each other in organic molecules. As soil carbon increases there should be a corresponding increase in soil nitrogen, due to linkages between the cycling of C and N. Increases in SOM have been shown to increase belowground respiration (Kudeyarov and Kurganova, 1998; Silver and Miya, 2001), and the same pattern was seen for both SOC and SON in our study (Figure 3.2). A negative correlation has also been observed with C:N ratios and belowground respiration, as was seen in our study, with C:N ratios between 20 and 25 resulting in the highest belowground respiration values (Silver and Miya, 2001). Correlations between root respiration and SON content have been found to be stronger than those found with SOC content (Jones and Cox, 2001).

Strong associations have also been found between N mineralization and belowground respiration where the N mineralization was controlled by SMB and enzymes in the soil (Deng et al., 2000). The association between belowground respiration and SON levels could possibly be due to the similarities between the relationships of each to soil temperature and moisture and their dependence on SMB activity (Zak et al., 1999).

The total flux of C from belowground sources is due to the cellular activity of both roots and SMB. Each of these factors is regulated by nutrient or substrate availability, soil moisture and soil temperature (Voroney, 1981; Mann, 1986; Saggar et al., 1997). The available nutrients, such as C and N, in the soil are also controlled by soil moisture and soil temperature (Saggar et al., 1997; Zak et al., 1999). Wright and Reddy (2001) found that SMB activity increased with soil organic C and N. The different levels of soil moisture found over most soil landscapes result in differing levels of plant productivity and SOM inputs to the soil (Pennock and van Kessel, 1997). This relationship means that as soil moisture increases there is an increase in SOM, due to increased organic inputs from plants, and the increased SOM results in more substrate and nutrients available for SMB to utilize. In root systems if levels of nutrients are low more photosynthate is allocated to belowground systems to obtain nutrients and the increased root biomass would result in higher belowground respiration values compared to areas with higher nutrient content (Saggar et al., 1997).

This interaction of soil properties results in a complex relationship between the controls of belowground respiration since many of the variables are linked to each other. Changes in substrate quality, SMB populations, root activity, soil temperature or soil moisture can result in varied levels of belowground respiration through both space and time (Pongracic et al., 1997). The level of belowground respiration measured also depends on soil type, plant cover species, vegetation age and land-use, with agricultural soils showing the least variation in respiration due to management practices which create similar conditions across a field (Kudeyarov and Kurganova, 1998). Due to complex relationships involved with below around respiration the low r^2 values in the single factor regressions compared to the multiple regressions are to be expected since much of the variation in respiration is likely due to a combination of factors, not all of which could be measured using a simple experimental design. The r² values were higher for the weekly data instead of site data, perhaps because the temporally separated data is a better predictor of belowground respiration (Silver and Miya, 2001).

There is also the possibility that variations observed in belowground respiration were due to errors in the sampling techniques. Use of soil chambers for belowground respiration has been shown to alter the CO_2 concentration gradient between the soil and atmosphere in the chamber headspace, alter the pressure in the chamber compared to the atmosphere, and modify air motion (Pongracic et al., 1997; Janssens et al., 2000). All of these can change the CO_2

efflux between the soil surface and the chamber compared to the surrounding areas and represent a possible source of error in our measurements. There is also the possibility that some of the variables measured are not related to belowground respiration. In the case of root surface area they have been shown to have a logarithmic relationship with belowground respiration, not a linear one (Ben-Asher et al., 1994), though in the case of our results this is not what was observed.

From this analysis it can be seen that belowground respiration is controlled by the interaction of a complex set of environmental and biological variables. Of the variables that we measured soil temperature and soil nitrogen content were determined to have the most influence on belowground respiration using a linear regression model. These factors were not individually significant, but they had the highest r^2 values of the factors measured when comparing individual points in the pasture. The best r^2 values were seen when examining belowground respiration using the weekly mean values for temperature and moisture, most likely because this method minimizes the confounding variation in soil properties across the site by taking the mean of all points. Although our measurements did explain a significant amount of variation in belowground respiration when examined using multiple regression techniques, they did not explain all of the variation observed. A more detailed study over a longer sampling interval including all possible variables that could influence belowground respiration such as vegetation type and age, soil microsite

differences in texture and structure, and other nutrients found in soil would be needed to more fully understand the controls of belowground respiration. It is important to fully understand belowground respiration due to its importance in the C-cycling of terrestrial ecosystems and the potential influence it could have on C balance as climate change occurs.

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Appendix A

Root Surface Area Calculation:

$$A_r = P_0$$

 $(P_0 + P_1)$

 $A_o = A_i * A_r$

Where A_r is the area of the image that is covered by objects, P_0 is the number of 'on' pixels, P_1 is the number of 'off' pixels, A_0 is the area of objects (roots) in the image, and A_i is the image area.

<u>CO₂ Evolved from CFI Sample Calculation:</u> CO₂ = (HCl_{blank} – HCl_{sample}) * (0.25 mM HCl) * (6 mgC/mmol NaOH) * 25mL/10mL

kg m_{od}

CO₂ = (9.92mL - 3.25mL) * (0.25 mM HCl) * (6 mgC/mmol NaOH) * 25mL/10mL

12.02 kg

 $CO_2 = 2079 \text{ mg } CO_2/\text{kg soil}$

Microbial Biomass C from CFI Sample Calculation:

 $C = (F_c - NF_c)/K_c$

C = (2195 - 2079)/0.41

C = 281

Data Used for Statistical Analysis:

Table 3.1. Data for weekly mean statistics.

Date	Respiration	Temperature	H₂O
	$(\mu mol m^{-2}s^{-1})$	(°C)	(%)
28-May-01	6.79	10.32	44
04-Jun-01	11.13	13.11	51
11-Jun-01	9.36	12.38	65
25-Jun-01	15.26	9.82	41
02-Jul-01	11.74	15.80	61
09-Jul-01	12.48	16.12	61
16-Jul-01	11.55	15.87	73
18-Jul-01	9.47	9.18	64
06-Aug-01	14.21	15.50	27
13-Aug-01	13.57	16.60	46
20-Aug-01	7.99	14.74	61
27-Aug-01	7.69	13.55	55

Table 3.2. Data for site mean sta

				Root surface				
Site	Respiration To	emperature	H₂O	area	SON	SOC	C:N	SMB
	(µmol m ⁻² s ⁻¹)	(°C)	(%)	(m ² root m ⁻³ soil)	$(Mg ha^{-1})$	(Mg ha ⁻¹)		(mg C kg ⁻¹ soil)
1	9.09	14.03	49	1.21	5.34	57.82	10.83	1130.00
2	10.22	13.79	37	11.27	8.05	127,10	15.78	2189.57
3	10.03	12.72	50	6.48	4.02	53.01	13.19	814.74
4	14.86	14.31	42	5.15	7.00	101.68	14.53	2102.07
5	11.52	13.84	37	7.42	5.69	98.56	17.31	1087.30
6	9.26	14.22	48	3.89	5.07	78.93	15.58	1378.34
7	7.52	12.92	58	10.57	3.11	36.11	11.61	1460.40
8	7.47	13.15	60	6.28	4.73	71.93	15.20	1910.49
9	10.27	14.61	53	5.73	6.01	75.74	12.60	1819.21
10	9.39	12.88	31	2.31	3.60	61.92	17.19	556.25
11	12.75	14.00	39	2.03	3.29	37.13	11.27	519.42
12	11.61	13.25	52	3.55	4.33	71.18	16.42	1427.99
13	11.28	13.69	33	1.89	4.20	55.59	13.25	759.65
14	7.70	13.83	39	6.53	5.08	93.50	18.42	741.16
15	10.49	13.87	32	0.52	1.46	23.19	15.91	309.37
16	10.28	15.04	44	4.72	3.04	44.33	14.57	770.36
17	14.16	13.75	47	1.79	3.71	39.52	10.66	594.06
18	5.61	13.24	37	4.93	2.63	37.47	14.27	663.91
19	7.60	12.76	36	2.97	3.54	55.72	15.74	648.23
20	7.06	12.00	40	2.26	2.63	58.96	22.41	613.89
21	7.76	13.60	30	2.32	4.51	60.02	13.30	931.40
24	8.07	13.83	32	1.64	1.76	30.27	17.23	477.27
25	6.49	13.87	31	1.88	0.96	13.33	13.94	328.04
27	14.33	13.10	47	7.14	3.67	56.82	15.49	1443.98

Table 3.3. Multiple regression parameters for site means.

Parameter	Value	Error	t-Value	Prob> t	R^2	
					0.252	
Y-Intercept	10.858	21.748	0.499	0.624		
SOC	0.144	0.231	0.623	0.542		
smb	0.001	0.002	0.311	0.760		
temp	0.450	0.980	0.459	0.652		
h2o	0.031	0.093	0.332	0.744		
roots	-0.338	0.350	-0.964	0.349		
SON	-1.655	3.545	-0.467	0.647		
CN	-0.651	0.858	-0.759	0.459		

Table 3.4. Mul	Multiple regression parameters for weekly				
Parameter	Value	Error	t-Value	Prob> t	R^2
					0.483
Y-Intercept	7.833	5.489	1.427	0.187	
temp	0.724	0.320	2.259	0.050	
h2o	-0.133	0.067	-2.001	0.076	

<u>Chapter 4: A comparison of carbon pools and belowground CO₂ fluxes</u> <u>between pasture, regenerating, and mature forest in sub-Boreal British</u> <u>Columbia</u>

Introduction

Accumulation of soil organic matter (SOM) represents a potential means for mitigation of increasing levels of atmospheric CO₂ (Robertson et al., 2000; Batjes, 1998). The potential for enhanced carbon (C) storage in soils, especially those under pastoral and agricultural use has led to renewed interest in the C dynamics of these systems (Pennock and van Kessel, 1997; Uri, 2000). Differences in land management practices and patterns of land use determine the amount of C that can be accumulated on a landscape (Houghton and Hackler, 2000; Uri, 2000). The C dynamics for many ecosystems, especially belowground, are incomplete and need to be further examined for many ecosystem types (Buyanovsky et al., 1987; Schimel, 1995).

When SOM is incorporated into the soil, its stability can be facilitated or affected by a number of soil properties and processes. These include soil clay content, soil moisture, soil temperature, bulk density, soil aggregation, parent material, and topography. One of the most important factors contributing to SOM stabilization is soil clay content (Rosell et al., 2001; Percival et al., 2000; Romkens et al, 1999; Amelung et al., 1998; Fearnside and Barbosa, 1998; Homann et al., 1995). Much of the SOM in soils is contained in the clay fraction (Rosell et al., 2001), but the amount of SOM present in the soil is determined by the interaction of clay content and factors such as water content, temperature,

climate, vegetation and productivity, decomposition and nutrient availability, and other properties of the mineral soil (Homann et al., 1995; Mäkipää et al., 1999; Batjes, 1998; Franzluebbers et al., 1996; Schimel, 1995).

Precipitation and temperature can control the amount of SOM stored in the soil (Percival et al., 2000). Cool, temperate forests of high latitudes contain the highest amount of SOM, while warm, tropical forests at low latitudes contain relatively small amounts of SOM (Dixon et al., 1994). Therefore, the expectation is that warming of overall climate will result in declining SOM due to increased mineralization of carbon in temperate and Boreal ecosystems (Mäkipää et al., 1999). Climate and soil texture interact to control levels of SOM (Burke et al., 1989), with the effect of climate on SOM seen most strongly in the clay-sized fractions of soils (Amelung et al., 1998).

How a soil's physical, chemical, and biological properties interact to stabilize SOM is also a function of land use. In undisturbed systems, SOM accumulation is slow or at steady-state (Fearnside and Barbosa, 1998; Bhatti et al., 2001). If a system is disturbed, either through land clearing or cultivation, the C balance of the ecosystem can be greatly altered. For example, clearing of forests can result in a loss of most of the aboveground biomass and loss of belowground woody biomass (Thuille et al., 2000). Forestry practices can also alter the physical and biological properties of the soil resulting in a release of CO₂ to the atmosphere and a reduction in SOM (Conant et al., 2001; Ellert et al.,

2001). Conversion of grassland to agriculture can result in similar effects on SOM. While the standing aboveground biomass is present in similar quantities for part of the growing season, harvesting in agricultural systems reduces litter inputs to these systems (Buyanovsky et al., 1987). Common agricultural practices such as tillage result in the exposure of SOM to favourable environmental conditions for soil microbial biomass (SMB) and much of the SOM can be lost through decomposition after cultivation (Voroney et al., 1981; Ellert and Gregorich, 1996; Uri, 2000).

Over time, a disturbed ecosystem can reach a new steady-state, but it is not necessarily the same as that existing prior to disturbance and the SOM can be either lower or higher than the original depending on site characteristics (Mann, 1986; Fearnside and Barbosa, 1998; Batjes, 1998; Rosell et al., 2001). Aboveground C pools can be permanently lost in forest ecosystems if they are converted to non-forest use and an overall reduction in SOM can be seen in both forest and grassland ecosystems in many cases (Voroney et al., 1981; Conant et al., 2001; Ellert et al., 2001). There is a need for quantification of the impacts of forest management and its effects on C pools and fluxes (Lee et al., 2002). When quantifying the impacts of forest management, there is also a need to examine a variety of forest types and ecosystems (Halldin et al., 1999) and to make measurements at small scales to capture natural variability in C pools and fluxes in order to permit modelling at local to regional scales (Banfield et al., 2002).

Boreal and sub-Boreal areas are ecosystems of potential importance to overall global C pools and fluxes. This importance is due to the large pools of C, which make up approximately 15 % of terrestrial C, contained in Boreal ecosystems resulting from low annual temperatures and short growing seasons, and the high potential for these C pools to diminish under climatic warming (Halldin et al., 1999; Mäkipää et al., 1999; Price et al., 1999; Harding et al., 2001; Read et al., 2001; Banfield et al., 2002; Falge et al., 2002; Lee et al., 2002). The large amount of C stored in Boreal ecosystems and the high potential for the storage of C to change as global climates change has resulted in increased interest in the quantification of the C pools and fluxes in these areas.

It is the objective of this study to quantify and assess the variability of C pools and belowground fluxes between pastures, regenerating forests and mature forests, and to determine the impacts of forestry and conversion to pasture on C pools and belowground fluxes in sub-Boreal British Columbia.

Methods

Site description

Carbon pool and belowground C fluxes were assessed in a pasture site (Chapter 2) and adjacent regenerating forest and mature forest near Hixon, BC (53°19'N, 122°34'W and 53°22'N, 122°38'W) (Figure 4.1). Regenerating forest species were predominantly lodgepole pine (*Pinus contorta*) and trembling

aspen (*Populus tremuloides*) with an understory of *Vaccinium* species, *Hieracium aurantiacum*, *Rosa acicularis*, *Comus canadensis*, and various grass species (*Poa, Bromus, Phleum*). The mature forest was dominated by Douglas-fir (*Pseudotsuga menziesii*) and spruce (*Picea englemannii*) trees, with lesser components of lodgepole pine and trembling aspen and an understory consisting of *Vaccinium* species, *Linnea borealis*, *Lonicera involucrata*, *Rosa acicularis*, *Cornus canadensis*, and various grass species. The regenerating forest sites were grazed throughout the field season. The mature forest area north of the pasture site was selectively logged 25 years before sampling occurred. Records





were not available for the volume of wood removed, but it is likely that the aboveground biomass estimates are an underestimate for the undisturbed mature forests in the area. Both forest cover sites had Dark Gray Luvisols with high clay contents, with some of the mature forest sites having a coarser texture than the pasture and regenerating forest.

Soil organic matter

Soil samples from the pasture site were taken at thirty randomly selected points during the first week of July 2001. Soil samples were also taken at five points in the regenerating forest sites, due to limited sampling area and the potential for edge effects to occur if the 20 x 20 m sampling grid was moved to include more points, and at thirty points in the mature forest sites during the 2002 growing season. Samples were taken in increments of 10 cm to a depth of 30 cm for determination of soil organic carbon (SOC). Representative sub-samples of the soil were selected, oven dried at 105 °C for 48 hours and finely ground for elemental analysis. SOC was determined by elemental analysis using an NA 1500 Elemental Analyzer (Fissions Instruments SPA., Milano, Italy). SOC results were corrected for equal mass of the soil following the method outlined in Ellert et al. (2001). Bulk densities measured in the field for the mineral soil were 0.85±0.03 for the pasture, 0.71±0.04 for the mature forest, 1.35±0.06 for the regenerating forest, assuming a bulk density of 0.1 g cm⁻³ for the forest floor samples (Federer et al., 1993; Liski and Westman, 1995). (See Appendix B for sample calculations.)

Belowground respiration, temperature and moisture

Belowground respiration, temperature and moisture were measured weekly at ten representative sampling points in the pasture site near Hixon, BC as well as at the five regenerating forest sampling point and at five of the mature forest sampling points from 4 June until 30 August 2002 as weather permitted. Fewer samples were used in the regenerating forest and mature forest due to the small size of the forest patches used for sampling. Belowground respiration and temperature were measured using an LI-6200 portable photosynthesis system in conjunction with an LI-6000-09 soil respiration chamber (LI-COR Inc., Lincoln, NE), accurate within ±0.3 ppm and calibrated daily when in use. Soil moisture was determined at each respiration sampling point using a Hydrosense soil moisture probe (CS620, Campbell Scientific, Logan, UT).

Aboveground biomass

Measurement of aboveground biomass in the pasture was performed from 2 to 5 July 2002 using a 0.25 m² quadrat immediately adjacent to the ten representative sampling points used for belowground respiration and at three additional ungrazed sample points to determine if there was a significant difference between grazed and ungrazed biomass values. These samples were oven dried at 70 °C for 24 hours and weighed. In the mature forest and the regenerating forest understory biomass was measured using the same technique as in the pasture from 2 to 30 July 2002 at each of the SOC sample points. A tree that reflected stand characteristics in the area around the sample point was
also selected at each site for measurement of diameter at 1.3 m, and the height was measured. Tree measurements were used to calculate biomass using the method outlined in Standish et al. (1985). Tree biomass over an area was calculated using stand density measurements taken over 100 m² plots in each of the sampling areas. It was assumed that all understory plant components contained 49 % C, coniferous trees contained 42 % C and deciduous trees contained 46% C for calculation of C content of the biomass (Milne and Brown, 1997; Poorter and de Jong, 1999). (See Appendix B for sample calculations.)

Coarse woody debris

A quantification of coarse woody debris (CWD) was performed along four transects in the mature forest areas following the method in Keisker (2001). The regenerating forest did not contain any CWD as defined by a minimum diameter of 7.5 cm. Transects of 24 m in length were laid out parallel to the adjacent road and woody debris piece length and diameter at the point of crossing were measured. CWD densities were calculated using the method outlined in Van Wagner (1982). A sample of the CWD was also taken for wood density determination and these samples were measured, oven dried at 70 °C for 24 hours, and weighed. It was assumed that all pieces of CWD measured contained 49 % C for calculation of C content of the CWD (Laiho and Prescott, 1999). (See Appendix B for sample calculations.)

Belowground biomass

In the pasture, soil cores 23 cm in length and 5 cm diameter were obtained during the first week of July 2001 at each soil respiration sampling point for determination of root properties. Root-containing soil was washed through a 0.7 mm soil sieve to obtain roots and the roots were soaked overnight in water so that they were saturated. These roots were then weighed to determine root biomass and the value converted to Mg root ha⁻¹. For the mature and regenerating forests belowground biomass was calculated using aboveground biomass and formulas from Li et al. (2003) of:

$$RB_s = 0.222 * AB_s$$
 (1)

$$RB_{\rm h} = 1.576 * AB_{\rm h}^{0.615} \tag{2}$$

where RB is the root biomass, AB is the aboveground biomass, the subscript s designates softwood species, and the subscript h designates hardwood species.

Results

Soil organic carbon

The lowest levels of total SOC were measured in mature and regenerating forest areas with a mean total SOC to a depth of 30 cm of 89 ± 13 Mg C ha⁻¹ and 66 ± 18 Mg C ha⁻¹, respectively. The pasture had the highest total SOC (196±21 Mg C ha⁻¹). The much higher values for the pasture were due to high levels of SOC present in the mineral soil (Figure 4.2). The regenerating forest had lower SOC levels in the all layers compared to the mature forest site. Comparisons of total SOC between the sites show that there was 219 % more SOC in the

pasture (significant, $p \le 0.0001$) and 25 % less SOC in the regenerating forest (not significant, p=0.5) when compared to the mature forest site.



Figure 4.2. SOC levels from left to right in the pasture, adjacent mature forest and regenerating forest sites for the litter layer and at depths of 0-10 cm, 10-20 cm, 20-30 cm, as well as total SOC including soil and litter for the site. SOC values with the same letter at each soil layer were not significantly different at the p=0.05 level using single factor ANOVA.

Belowground respiration

Belowground respiration was highest in the pasture site for the measured points in June and July, but was the lowest for the majority of August before returning to the highest respiration rates towards the end of the growing season (Figure 4.3). The regenerating forest generally had the lowest belowground respiration throughout the growing season except in August when the pasture site rates dropped and the mature forest site was higher than the regenerating forest site with the exception of one sampling date in August. In the pasture mean belowground respiration for the season was $6.9\pm1 \ \mu mol \ m^{-2} \ s^{-1}$, in the





mature forest it was $5.6\pm0.4 \ \mu\text{mol m}^{-2} \ \text{s}^{-1}$ which was not significantly different (p=0.3) from the pasture, and in the regenerating forest it was $4.3\pm0.2 \ \mu\text{mol m}^{-2} \ \text{s}^{-1}$, which was significantly different from both the pasture (p=0.03) and mature forest (p=0.01) values.

Aboveground and root carbon

The mature forest site had the highest levels of aboveground biomass C and the pasture site had the lowest levels (Figure 4.4). There was no significant difference (p=0.1) found between the grazed (2.4 ± 0.3 Mg C ha⁻¹) and ungrazed (3.5 ± 0.5 Mg C ha⁻¹) pasture biomass samples so they were combined to give a



Figure 4.4. Aboveground biomass and total carbon measured for pasture, and adjacent mature and regenerating forest sites from left to right. Carbon pool values with the same letter for each land type were not significantly different at the p=0.05 level using single factor ANOVA.

mean of 2.7±0.3 Mg C ha⁻¹. Aboveground biomass was 81 % lessin the regenerating forest and 99 % less in the pasture than in the mature forest. The mature forest had the highest root biomass C of 18 ± 6 Mg C ha⁻¹, the regenerating forest contained the second highest root biomass C of 3.7 ± 0.4 Mg C ha⁻¹, and the pasture contained the least root biomass C with 0.18±0.02 Mg C ha⁻¹. Overall the mature forest had the highest total C, while the pasture and regenerating forest contained similar amounts of total C (Figure 4.4). Mature forest total also contained 1.0 ± 0.4 Mg C ha⁻¹ of CWD C.

Discussion

The mature forest site had the highest C content aboveground (Figure 4.4), largely in the form of aboveground biomass with a smaller contribution from CWD. In contrast, the pasture exhibited the highest SOC concentrations (Figure 4.2) due to increased organic matter inputs from the fibrous root systems present in grass species despite losses of surface organic layers due to burning and incorporation into the mineral soil after clearing and cultivation. The SOC lost when a site is burned during site preparation before cultivation can result in a permanent loss of C in surface organic layers for the site (Ellert and Gregorich, 1996; Fearnside and Barbosa, 1998). Grass root systems represent a major input of SOC for grassland and pasture ecosystems and it has previously been observed that conversion of forests to pasture can result in increased SOC (van Ryswyk et al., 1966; Sparling et al., 1994).

In the pasture site, the predominance of grass species that have increased root turnover and density at the soil surface, could explain the increased belowground respiration (Figure 4.3) (Ben-Asher et al., 1994), but the inputs by roots into the system offset these losses resulting in an overall gain in SOC. High belowground respiration in pasture systems when compared to forests has been observed in other areas (Sparling et al., 1994). Relatively higher amounts of substrate from root inputs in the pasture area compared to the forested area could be the cause of increased belowground respiration observed due to greater activity of soil microbes in response to substrate availability (Paul and Clark, 1996). In the pasture, the forest floor was incorporated into the mineral soil during cultivation, undoubtedly contributing in part to the higher levels of SOC from 0 to 30 cm, but there was no litter layer present for comparison with the forest floor in the forested sites.

The values of belowground respiration for the mature forest are at the lower end of the range of 4.4 to 12.7 μ mol m⁻² s⁻¹ found in Boreal forests by Billings et al. (1998), but are higher than the mean belowground respirations of 2.7 μ mol m⁻² s⁻¹ found by Kelliher et al. (1999) and 2.8 μ mol m⁻² s⁻¹ found by Rayment and Jarvis (2000). Belowground respiration values for the pasture were generally within the range observed by LeCain et al. (2002) of 4.9 to 7.6 μ mol m⁻² s⁻¹ on a lightly grazed pasture and the values for the regenerating forest were within the range of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forests of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forest of 2 to 10 μ mol m⁻² s⁻¹ observed in regenerating forest were

The mature forest had aboveground C pools of 305 Mg ha⁻¹ similar to the value of 241 Mg ha⁻¹ found in Boreal forests by Lee et al. (2002), but the belowground carbon in this study of 53 Mg ha⁻¹ was closer to the 62 Mg ha⁻¹ C found belowground in temperate forests than the 409 Mg ha⁻¹ found in Boreal forests by Malhi et al. (1999). These differences were most likely due to differences in species composition and climatic factors between our study area and theirs, located in northern Saskatchewan. Our value for aboveground carbon in the pasture was 2.6 Mg ha⁻¹, which was lower compared to the value of 10 Mg ha⁻¹ found by Houghton and Hackler (2000), but their value included data from sagebrush areas as well as grass dominated ecosystems. The belowground carbon in the pasture was higher than that found by Sparling et al. (1994) for fertilized and unfertilized pastures in New Zealand. The belowground carbon in the 18-year-old regenerating clearcut was lower at 66 Mg ha⁻¹ compared to 90 Mg ha⁻¹ in a five-year-old regenerating clearcut (Lee et al., 2002). This could possibly be due to repeated soil disturbance and removal of leafy vegetation by grazing cattle in the regenerating clearcut area.

Total C pools for the sites showed that the mature forest site had the highest amount of C (Figure 4.4) due to the large amounts of C contained in the aboveground biomass and CWD the values found of 412 Mg ha⁻¹ are lower than those found by Malhi et al. (1999) of 458 Mg ha⁻¹ in Boreal forests in northern Saskatchewan, but higher than those found by Price et al. (1999) of 122 Mg ha⁻¹

from locations across northern Canada. The regenerating forest site total C pools were the lowest (128 Mg ha⁻¹) due to lower aboveground biomass compared to the mature forest, and lower belowground C inputs due to lower levels of litter inputs from aboveground biomass and fewer inputs directly into the soil from grass roots. The values we found in the regenerating forest were higher than those estimated by Brown (2002) of 90 Mg ha⁻¹ using an estimated 5 Mg ha⁻¹ y⁻¹ for regenerating softwood forests. The pasture site had the second highest total C pools of 193 Mg ha⁻¹ with the largest portion from belowground C due to high belowground inputs, which is lower than the IPCC (2000) had estimated for temperate grasslands of 243 Mg ha⁻¹, possibly due to the effects of grazing on our study site.

These results indicate that there is an overall loss of C from ecosystems after clearing of forests regardless of whether the land is allowed to regenerate a forest community or is converted to pasture. Although there is an increase in SOC in the mineral soil of the pasture ecosystem, there is an overall loss of C due to losses of aboveground plant biomass and CWD. Other studies have shown that clearing of land for pasture use has resulted in an increase in SOC compared to the original forest, but this did not make up for the loss of biomass C in the ecosystem (Fearnside and Barbosa, 1998; Conant et al., 2001). Our results differ from a study of temperate ecosystems by Ellert and Gregorich (1996) where SOC in cultivated ecosystems was less than that of adjacent forest

ecosystems, most likely because of more intense use of land relative to our study.

The differences in C pools have implications for land managers in sub-Boreal regions when considering clearing of forested land if management goals include the sequestration of C because clearing results in an overall loss of C. There is the potential to increase C pools in previously deforested ecosystems using management practices to reduce the net loss of C from clearing and increase SOC (Batjes, 1998; Robertson et al., 2002; Uri, 2000). In sub-Boreal ecosystems, a combination of management practices are needed that both permit harvesting of forests for timber and conversion to non-forest use and minimize losses of C from these ecosystems. Possible alternatives to clearcutting of forests and conversion to pasture or tree plantations include partial cutting of forests and agroforestry, as well as a movement towards uses that allow multiple uses of the land base in order to minimize C losses from system (Batjes, 1998), though in order to have the same total C more land would need to be altered. No matter the intended use of the land, the impacts of changes in land use need to be considered in terms of the C balance of landscapes in order to reduce inputs of C to the atmosphere and maximize C uptake and storage in terrestrial ecosystems.

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Appendix B

Soil Organic Matter Calculation (adapted from Ellert et al., 2001):

Calculation of soil mass and C mass where M_{soil} is the mass of soil per unit area (Mg ha⁻¹), M_c is the mass of C per unit area (Mg ha⁻¹), D_b is the bulk density of the soil (Mg m⁻³), T is the thickness of the soil layer (m), and conC is the concentration of C from lab analysis (kg Mg⁻¹):

 $M_{soil} = D_b * T * 10,000 m^2 ha^{-1}$

 $M_{soil} = 1.3 \text{ Mg m}^{-3} * 0.1 \text{ m} * 10,000 \text{ m}^2 \text{ ha}^{-1} = 1296 \text{ Mg ha}^{-1}$

 $M_c = (conC/1000) *D_b * T * 10,000 m^2 ha^{-1}$

 $M_c = (51.94/1000) *1.3Mg m^{-3} * 0.1 * 10,000 m^2 ha^{-1} = 67.32 Mg ha^{-1}$

The equivalent mass of C needed to compare soils on an equivalent mass basis was determined by adding additional thickness of the soil layer (T_{add} , m), where $M_{soil, equiv}$ is the mass of the heaviest layer (Mg ha⁻¹), $M_{soil, surf}$ is the sum of soil mass in surface layers:

$$T_{add} = (M_{soil, equiv} - M_{soil, surf}) * 0.0001 \text{ ha m}^2$$

Db

$$T_{add} = (1518 \text{ Mg ha}^{-1} - 1296 \text{ Mg ha}^{-1}) * 0.0001 \text{ ha m}^{-2} = 0.02 \text{ m}$$

1.3 Mg m⁻³

The amount of C contained in the additional layer to add was based on the C concentration of the layer being added to due to how the data was sampled where $M_{c, add}$ is the mass of C in the additional layer (Mg ha⁻¹):

 $M_{c, add}$ = (conC/1000) *D_b * T_{add} * 10,000 m² ha⁻¹ $M_{c, add}$ = 11.53 Mg ha⁻¹ The C in the additional layer ($M_{c, add}$) was then added to the original layer ($M_{c, surf}$) to determine the mass of carbon in an equivalent soil mass ($M_{c, equiv}$), all in Mg ha⁻¹:

 $M_{c, equiv} = M_{c, surf} + M_{c, add}$

 $M_{c, equiv} = 67.32 \text{ Mg ha}^{-1} + 11.53 \text{ Mg ha}^{-1} = 78.85 \text{ Mg ha}^{-1}$

Aboveground Biomass:

Biomass of the representative tree (B, kg) was calculated using the measured values of tree diameter at 1.3 m (D, m), and tree height (H, m). Species specific constants to determine biomass (a and b) from Standish et al. (1985) were used in the formula:

B =
$$a + b * D^2 * H$$

B = $82.6 + 272.3 * (0.108)^2 * 9.35 = 112 \text{ kg for a lodgepole}$
pine

This value was converted to kg C based on coniferous trees containing 42 % C resulting in 47 kg C in the example tree. This value was extrapolated to an area based measurement using a measured mean stand density of 0.13 trees m⁻² resulting in 6.2 kg C m⁻² for the example.

Coarse Woody Debris (CWD):

CWD volume (V, $m^3 ha^{-1}$) was calculated using the measured diameter of CWD pieces (d, cm), the length of the transect used (L, m) and the angle that the CWD piece was on relative to the ground (a, °) and the formula from Van Wagner (1982):

 $V = (1.234/L) * (d^2 * sec(a)) m^3 ha - 1$

 $V = (1.234/24 \text{ m}) * ((25 \text{ cm})^2 * \text{sec}(0)) = 32 \text{ m}^3 \text{ ha}^{-1}$

The volume (v, m³) of the wood for each CWD piece was then measured and after oven drying the mass (M, g) was measured and this was used to determine the C content (M_c, g) of each piece based on CWD containing 49 % C (Laiho and Prescott, 1999), the C density (C, g C m⁻³) of the CWD was then determined. This value was then multiplied by the volume of CWD per hectare (V) and converted to Mg C ha⁻¹ of CWD-C with 1.42×10^{-6} Mg C ha⁻¹ for the example.

 $M_{c} = M * 0.49$

_{Mc} = 75.87 g * 0.49 = 37.17 g C

 $C = M_c / v$

 $C = 37.17 \text{ g C} / 840 \text{ m}^3 = 0.44 \text{ g C m}^{-3}$

<u>Chapter 5: Land type classification and distribution of carbon pools and</u> <u>belowground fluxes in a sub-Boreal pasture-forest mosaic.</u>

Introduction

Assessment and monitoring of environmental properties that influence carbon (C) dynamics is becoming increasingly important as atmospheric concentrations of carbon dioxide increase. Quantification of C pools and fluxes across all terrestrial ecosystems is necessary to determine the magnitude and location of terrestrial C sinks, as well as to determine any areas that have high potential for change as global temperatures increase (Tans et al., 1990; Schimel, 1995). High latitude forests, including Boreal forests, have already been recognized as one of the areas with both large pools of C and high potential for change in these pools (Woodwell et al., 1983; Dixon et al., 1994; Falge et al., 2002).

Along with the potential changes within these ecosystems due to climate change, there are also changes associated with land use management in high latitude forests. Clearing of aboveground biomass for timber production or conversion to other uses, such as agriculture, represents a major C loss from the tree biomass and can result in changes in belowground C pools as well (Woodwell et al., 1983; Johnson et al., 1991). After a forested site is cleared changes in soil properties such as soil temperature, soil moisture and organic matter inputs to the soil often result in a large release of C from the soil (Burke et al., 1989; Ellert and Bettany, 1995; Batjes, 1998). This flush of C has been

attributed to the creation of a more favourable microclimate for decomposer organisms through soil disturbance (Voroney et al., 1981; Paul and Clark, 1996; Mielnick and Dugas, 2000). Once soil is disturbed it can take from 30 to 60 years before it reaches an equilibrium with the environment and begins to accumulate C again (Mann, 1986; Ellert and Gregorich, 1996; Robert, 2001; Rosell et al., 2001).

Aboveground biomass that is removed during clearing of a forest can eventually be replaced by regenerating trees. Although the lost biomass can be replaced and C sequestered through vegetation growth, it occurs over long time frames and may not result in a comparable forest to the original (Harmon et al., 1990). In some cases the regeneration of forest vegetative biomass does not occur. If forested areas are converted to non-forest use, such as pastures, the vegetation regenerating on the site consists mainly of forage species and represents only a small fraction of the vegetation originally present on the site. In pastures and other agricultural ecosystems there is the possibility that the loss of aboveground C could be compensated for through increases in soil C, but this does not always occur (Sparling et al., 1994; Homann et al, 1995; Fearnside and Barbosa, 1996; Read et al., 2001).

Because of the differences between the carbon pools distributed on different land types it is important to determine the distribution of these over a larger scale. A greater understanding of C dynamics at a landscape scale is

needed to more clearly see how ecophysiology, natural ecosystem dynamics, land use management and natural variability interact to determine carbon balance (Malhi et al., 1999). The spatial distribution of ecosystem types and how they change with land management is also of importance to the terrestrial C balance and should be measured in order to analyze any changes that could affect C dynamics (Schmiel, 1995; Read et al., 2001).

Under the Kyoto Protocol monitoring of changes in C pools and fluxes across all ecosystems is required, including those induced by land use change (Coomes et al., 2002). The political pressure to determine the location and magnitude of C sinks and C pools placed on governments by the Kyoto Protocol has resulted in the need to quantify C dynamics in all land types in order to determine the extent and distribution of the terrestrial C sinks and sources (Cruickshank et al., 2000; Dong et al., 2002). Due to the broad range of ecosystems in which C dynamics are studied and the large amount of data generated from research there is a need for an accurate mapping technique that can be easily applied to any land surface type (Murdiyarso and Wasrin, 1995; Chen et al., 2000; Izaurralde et al., 2001).

The potential of using remotely sensed images such as air photographs as a tool in mapping of land types and C dynamics has been recognized, but has been underutilized in the past (Murdiyarso and Wasrin, 1995; Brown, 2002). The remote sensing techniques currently under development need to be used in

combination with ground based measurements due to the inability of remote techniques to quantify portions of the terrestrial C pool (Coomes et al., 2002). Measurement of C contained in soils and as coarse woody debris (CWD) is difficult in areas covered in vegetation and CWD is often excluded from C pool estimates (Coomes et al., 2002; Dong et al., 2002). There is also a tendency to base ground measurements to be used with remote sensing techniques in undamaged late successional ecosystems, which could potentially result in an incorrect estimate of C pools (Coomes et al., 2002).

A program to utilize data from air photographs in the form of digital orthophotos to classify and analyze the distribution of land types was developed. The program was used with data on C pools and belowground respiration to determine their distribution over a local to regional scale. Our objective was to present a preliminary study of the feasibility and accuracy of using remotely sensed data in the mapping of C dynamics between different land types including mature forests, regenerating forests, and pastures.

Methods

Determination of C pools and belowground respiration

Carbon pools and belowground respiration were measured for mature forests, regenerating forests, and pastures in a sub-Boreal ecosystem as outlined in Chapter 4. Means of the values obtained were taken as representative of those found in the area surrounding the study site near Hixon, BC and used with a mapping program to determine the distribution of C pools and belowground fluxes. Total loss of potential C pools was also determined for the study area based on the assumption that the area would be forested if land clearing had not occurred.

Classification and mapping of land types and associated C pools and fluxes

In order to classify the different land types surrounding the study site, a novel program intended to analyze orthophotos and determine land type was designed. Initially several test programs were coded to test the compatibility of the neural network package (Stuttgart Neural Network Simulator, IPVR, Stuttgart, Germany) and the graphics package (Netpbm, Open Source Development Network, Boston, MA). Once a compatible combination of neural network, graphics package and test program was determined, the inputs of the program (the orthophoto and the mask file) and the outputs (a colour indexed map of the orthophoto area) were determined. Once these were determined, the program was designed, coded in C, to transform the orthophoto to the output map through the neural network and was subsequently tested and modified to decrease any errors in the program's code, and to increase accuracy of the analysis.

The program that resulted from this process, the land type classifier (LTC), classified the orthophoto by analyzing an odd numbered square of pixels through the trained neural network and determining the most probable land type based

on greyscale values. This block of pixels then moves across the orthophoto, partially overlapping the previous section, to determine land types. The program was then tested to determine which parameters of samples needed to train the neural network and size of the pixel block resulted in the most accurate classification. The size of the pixel block tested ranged from 7 to 15, and the number of training samples ranged from 800 to 4000 (Galpin, 2002). From this we determined that small pixel block sizes had difficulty identifying patterns in mature forest due to the mosaic of age classes and gaps in forests, while large pixel block sizes had difficulty identifying smaller features, such as narrow roads and streams. We also determined that a larger number of training samples (above 2000) resulted in more accurate classification of land types.

Using an orthophoto for the mapsheet 93G038, which included the Hixon study site, a mask file of land types was created following the specifications used in Galpin (2002). Five land-surface types were used: pasture, regenerating forest, mature forest, water, and roads/bare soil. The mask file and the 93G038 orthophoto were used with the LTC program (Galpin, 2002) and used to train the neural network in the program using 2000 training samples chosen from the mask file and 11 pixel square grid. Once the neural network had been trained to classify the five different land types, the orthophoto of the 93G038 map sheet was analyzed and a colour-coded map of the land types was created. This map was analyzed using the analysis portion of the LTC to determine the areas of each land type in the orthophoto using a ratio of 1 m² for each pixel in the image.

Results

Analysis of the orthophoto for the Hixon study site (Figure 5.1) using the LTC showed that most of the area surrounding the Hixon site was mature forest (Figure 5.2), representing approximately 66 % of mapsheet 93G038 area (Table 5.1). Of the land types where measurements of C pools and fluxes were made pasture was the least prevalent on the landscape, with approximately 12 % of the area classified as pasture.

Although pasture and regenerating forest are relatively minor components of the entire mapsheet area (12.5 % and 15.6 % respectively), pasture contained higher belowground C pools and higher belowground respiration than regenerating forest because of higher SOC levels in the former. The large proportion of mature forest represented on the mapsheet area resulted in it being the major contributor to both aboveground and belowground C pools, but also having the highest total belowground respiration. The regenerating forest aboveground biomass C pools were several orders of magnitude higher than the pasture, but still much lower than the mature forest.

If land management had not resulted in clearing of portions of the forest and change to pasture or regenerating forest areas there would be a total area of 14326 ha or 94 % of the landscape area. This area would have a total C pool of 5.65×10^6 Mg C representing a 26 % increase with 4.38×10^6 Mg C aboveground

and 1.27x10⁶ Mg C belowground with increases of 26 % aboveground and a 10

% increase belowground. The total belowground respiration for the area would

be 6.91×10^5 Mg C y⁻¹ representing a 2 % increase over the present landscape.

Table 5.1. Proportions of land area of each land type and associated C pools and belowground respiration for the land type and the area represented by the land type in the orthophoto.

Land Type		CWD C (Mg C ha ⁻¹)	Aboveground C (Mg C ha ⁻¹)	Belowground C (Mg C ha ⁻¹)	Total C (Mg C ha ⁻¹)	Belowground Respiration (Mg C ha ⁻¹ y ⁻¹)
Pasture		-	2.66	196	115	57.3
Regenerating Forest		-	58.1	66	121	35.8
Mature Forest		1.0	305	89	359	48.2
	Total Area (ha)	Percent of Area (%)	Aboveground C Pools (Mg C)	Belowground C Pools (Mg C)	Total C Pools (Mg C)	Belowground Respiration (Mg C y ⁻¹)
Pasture	1901	12.5	5.05x10 ³	3.72x10⁵	3.77x10 ⁵	1.09x10⁵
Regenerating Forest	2357	15.6	1.37x10 ⁵	1.56x10 ⁵	2.93x10 ⁵	8.43x10 ⁴
Mature Forest	10068	66.6	3.08x10⁵	8.96x10⁵	3.98x10⁵	4.85x10 ⁵
Water	273	1.8	-	-	-	-
Roads and Bare Soil	522	3.5	-	-	-	-
Total	15121		3.22x10 ⁶	1.42x10 ^⁰	4.46x10⁵	6.78x10⁵

Discussion

In general the LTC correctly classifies the different land types with the neural network only making errors approximately 10 to 15 % of the time. If the neural network in the LTC were trained using a greater number of training samples, such as 4000 instead of the 2000 used, this would reduce the errors, but increase the time needed to train the network. The majority of the errors in classification were found in the identification of small roads and small shadowed

areas in the forest. Using a block of 11 pixels for classification causes small roads or patches of bare soil to be missed and these are generally classified as the same category as the overall patch that they run through. This problem could be reduced by using a smaller block size of 9 or 7 pixels, but would lead to increased problems with shadow patches. In the forested areas small clearings that are in shadow are classified as water instead of forest due to the dark coloration. This problem is of greater concern when using a smaller block of pixels for classification, but is reduced as the block size is increased up to 15 or 17 pixels. Due to the opposing nature of the solutions for the small road and shadowed forest clearings errors are going to occur during classification regardless of the pixel block size; therefore we chose a mid range size to attempt to minimize both problems. Another noticeable problem in the LTC classification is that regenerating clearcuts in the early stages of regeneration tend to be classified as pasture. This is most likely due to the presence of large amounts of deciduous vegetation in these clearcuts, which gives them similar coloration to the pasture vegetation, although the two land types do not share similar properties of C dynamics.

Due to the large proportion of mature forest over the landscape analyzed and the large C pools compared to the smaller belowground respiration values associated with the mature forests (Table 5.1), it is likely that the landscape both stores large amounts of C and acts as a net sink for C. The mature forest land type in general stores large amounts of C and conversion of mature forests to

other land types reduces the C pools and can cause the land to become a net C source (Fearnside and Barbosa, 1996; Pennock and van Kessel, 1997; Thornley and Cannell, 2000). This indicates that for maximum C sequestration in this area a reduction or change in clearing practices is necessary to preserve the mature forests, which contribute the most to increasing C pools.

When the values for mature forest C pools and fluxes were extrapolated over the cleared land area the overall C pools on the landscape increased (Table 5.1). The increased C would be a result of much higher aboveground biomass in mature forest compared to the pastures and regenerating forests. Also higher belowground C in the mature forest from decomposing woody root systems and incorporation of litter into the soil would increase total C.

Management practices that incorporate maintenance of the high aboveground C pools found in mature forests, while still allowing other activities such as timber harvesting and agricultural activities are needed in order to maximize C pools over the landscape and minimize changes that could negatively impact users of the land. One of the possible solutions to this problem is combining agriculture and forestry into a sustainable agroforestry system. Systems where forage species or crops have been grown with trees have been shown to increase C pools (de Jong et al., 2000; Padney, 2002). This type of management strategy can also decrease the amount of timber that needs to be harvested from undisturbed forests, with harvesting occurring on agroforestry

lands instead resulting in less C being lost from the system (Ford-Robertson et al., 1999; Padney, 2002). Management strategies that allow continued use of the land for economic benefits as well as preservation or increasing C pools are needed to mitigate atmospheric CO_2 increases.

When scaling of the values obtained for C pools and belowground respiration from the site to landscape level there are several concerns over the accuracy of this process. The sites chosen for the scaling must be representative of that land type over the whole landscape where scaling will occur. Sites must have similar climatic, landform, topographic and edaphic features as well as similar soil and vegetation types in order to be representative of the whole landscape (Izaurralde et al., 2001). If the study site chosen for scaling is not representative of the landscape data could be misinterpreted and the overall results skewed. There is also the problem of high variability within some of the site properties, such as soil type. The sites chosen as representative were based on general site characteristics, but due to the highly variable nature of soils, it is possible that there are pockets of other soil types both on the representative site and on the landscape which could skew the results when scaling up.

Overall the use of the LTC for landscape classification is a relatively simple and reasonably accurate means of determining the distribution of various land types. In the case of our study it has shown that the large amounts of mature forest on the landscape represent a major C pool and possibly a C sink, but clearing of these areas on a large scale could alter the C balance of the landscape. The ability to assess land use patterns over large areas and design management strategies from this information that gives the best potential use of the land base is essential for sustainable development and management of C pools. Use of small scale studies over a large variety of ecosystems including different land types in combination with analysis programs like the LTC represent a potential tool in decision making for land managers and policy makers in the future. If mitigation strategies for increasing atmospheric CO₂ are to be implemented, a system where all ecosystem types are included in C cycle measurements to determine their potential for C sequestration and the design of management strategies that maximize C sequestration both aboveground and belowground, while minimizing C sources is needed.

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Figure 5.1. Orthophoto image used for the analysis with the LTC program. The Hixon study site is outlined.

97 1 km



1 km 98

Chapter 6: Summary

It was determined that both pastures and regenerating forests acted as small net sinks for carbon (C) of -0.03 g C m⁻² and -0.05 g C m⁻² over the 2001 and 2002 growing seasons, respectively. Inclusion of C fluxes for the non-growing season in the pasture showed that the site was a net source for CO₂ of 0.90 g C m⁻². This result disagrees with what had previously been found on most agricultural sites, but many other studies did not include the effects of grazing or include full year measurements. If measurements were made over the full year there is also the potential that it would change the sign of the C flux to become a net source of CO₂, due to the small size of the net sink observed in the regenerating forest. A comparison of the regenerating forest site measurements with those from the same period in previous years showed that there is a large amount of interannual variation on the site with measurements of both net sinks and net sources occurring on the same site. The variation that was observed was most likely due to a combination of environmental factors that contributed to both the photosynthetic capacity of the site and the belowground respiration.

Further studies to determine the variability of C fluxes in pastures and regenerating forests, both between years and in different age classes, is needed to better determine the magnitude of the C flux in these systems. In any potential study, incorporation of year round data is essential to fully quantify the C flux from a site. Inclusion of other greenhouse gases such as methane and nitrous
oxide in these studies would also increase our knowledge of how these ecosystems contribute to the carbon cycle and to climate change.

Along with the need to further examine the differences in C fluxes between land types, there is also a need for a better understanding of the controls of CO₂ fluxes and how they affect C sequestration. In examining some of the controls of belowground respiration it was found that soil temperature and soil nitrogen content exerted the most control over belowground respiration, and both showed positive linear relationships. Soil moisture showed a weak linear relationship with belowground respiration, but, when examined using a non-linear relationship, it was found that there was a stronger correlation between soil moisture and belowground respiration with maximum respiration occurring at approximately 45% H₂O. Soil organic carbon also showed a linear relationship with belowground respiration, but was not as strong as that seen with soil nitrogen. The C to N ratio of the soil showed a negative relationship with belowground respiration, while root surface area showed no relationship to belowground respiration. Although our results did not show that any one environmental or biological parameter was significant in explaining the variation observed in belowground respiration, the combined effects of temperature, soil moisture, soil microbial biomass, soil carbon, soil nitrogen, carbon to nitrogen ratio, root surface area and the interaction between them explained a significant portion of the variation observed in belowground respiration. The remaining portion of unexplained variation needs to be explored in future studies using the

100

above variables as well as others such as the chemical composition and structure of soil organic matter, aggregation of soil, species of vegetative cover and any other potentially significant variables.

The effects of land use management and the clearing of forests were also clearly demonstrated by this study. Net losses of C occurred after clearing of forested lands, despite increases in soil organic carbon (SOC) observed in pasture areas. The mature forest sampled was shown to have a significantly higher total C pool when compared to both regenerating forest and pasture. The high total C was, for the most part, due to the high biomass C values, both above- and belowground, and in spite of the lowest SOC of the land types sampled. The pasture and regenerating forest had similar total C pools, due to high SOC in the pasture and relatively high biomass C in the regenerating forest. The pasture area also had the highest loss of C from belowground respiration, while the regenerating forest had the lowest respiration, most likely due to differences in available C for decomposition and belowground biomass respiration. Over time, the C losses from clearing of mature forest may be compensated for by increases in aboveground biomass in regenerating forests and belowground C inputs in pastures, but the length of time for this to occur, if it does occur, is uncertain.

The differences in magnitude of both C pools and C fluxes between land types makes it necessary to assess the distribution of different land types at a

101

larger scale. The land type classifier (LTC) program that was developed is a potential tool for land managers to use in this task. The LTC program allows assessment of existing land types and can be used in combination with existing data on C pools and fluxes to estimate these over a landscape. There is also the possibility that the LTC program could be used to determine what the effects of land management are in terms of C pools, using area based estimates. The combination of C pool measurement, C flux measurements, and the LTC program with knowledge of C cycling and its controls in terrestrial systems demonstrates potential for use as a tool to improve management of C at a landscape level.

Recommendations

Due to the uncertainty in estimates of existing C pools and fluxes, it is important to assess multiple age classes for C accumulation and to determine the distribution of mature forests, regenerating forests, pastures, and other cleared areas. Repeated assessment of C pools on the landscape over longer time intervals is needed to better assess the loss or accumulation of C over a larger scale. Land management decisions need to be based on current assessments of the landscape so that the best potential uses of the land base are implemented. Management strategies that include multiple uses of land, such as agroforestry, and management practices that increase C sequestration, such as conservation tillage, need to be implemented and monitored to ensure

102

that the maximum value of an area is obtained, in terms of both environmental concerns and economic benefits.

In the future, political pressure from the Kyoto Protocol or another similar agreement will demand monitoring of C fluxes and pools over a much larger area than is currently being studied. Simple methods of assessing C pools and fluxes and a better understanding of C dynamics are needed to meet this challenge. Research projects similar to those presented here need to be implemented in ecosystems representing all land types over the long term if mitigation of atmospheric CO_2 increases through terrestrial C sinks is to be feasible.