ENERGY, WATER, AND CARBON BUDGETS OF YOUNG POST-FIRE BOREAL FORESTS IN CENTRAL SASKATCHEWAN

BY

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ABSTRACT

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Models and previous studies have shown that climate change will bring increased global temperatures. The major cause of climate change is the emission of greenhouse gases into the atmosphere from anthropogenic and natural sources. Carbon dioxide is one of the major greenhouse gases being cycled by forests. Mature boreal forests are often carbon sinks, however, following disturbance, forests can become carbon sources, releasing carbon into the atmosphere. As temperatures increase with climate change, it is predicted that the frequency of forest fires in the boreal forest will increase. Fire will impact the energy and water budgets of the boreal forests which will also impact the carbon budget. Three previously burned forest sites in central Saskatchewan were studied for a minimum of three years using eddy covariance to obtain measurements of the energy, carbon, and water budget. Sites were burned in 1977 (F77 site), 1989 (F89 site), and 1998 (F98 site). Net radiation was similar at all sites for the duration of the study with maximum summer net radiation levels near 13 MJ m^{-2} d^{-1}. Sensible heat flux density and soil heat flux density were similar at F77 and F89 by 2005. Difference in vegetation characteristics were best illustrated by latent heat flux. Latent heat flux density increased gradually over time at F98. Latent heat flux densities were similar at F77 and F89 throughout the study period. Soil heat flux density was similar at all sites in 2004 and 2005 with maximum soil heat flux values of approximately 1 MJ m^{-2} d^{-1}.

Weekly net ecosystem production values at F89 were similar to the F77 site in 2004 and 2005 while F98 remained lower than the mature sites. F77 lost 40 g C m^{-2} and 79 g C m^{-2} in 2004 and 2005, respectively. F89 gained 177 g C m^{-2}, 113 g C m^{-2}, and 88 g C m^{-2} in 2003, 2004, and 2005, respectively. F98 lost 5 g C m^{-2}, 17 g C m^{-2}, and 52 g C m^{-2} during these three years, respectively. Understanding the recovery of boreal forests following fire furthers understanding of how climate change and disturbance in boreal forests could impact local, regional, and global climates in the future.

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1. INTRODUCTION

1.1 Boreal Forests, Climate Change, and Disturbance

Climate change is caused by major greenhouse gases that result in longwave radiation being trapped in the lower levels of the atmosphere. A rapid increase in greenhouse gases has been observed in the last decades (IPCC 2001). The major sources of greenhouse gases are fossil fuel emissions and changes in land use of terrestrial ecosystems (Sarmiento and Gruber 2002). The annual greenhouse gases from fossil fuel emissions are an order of magnitude less than from terrestrial ecosystems (Coursolle et al. 2006). The greatest radiative forcing is caused by carbon dioxide (CO_2), which is especially important in forests. Boreal forests are important carbon sinks that absorb some of this CO_2 through the process of photosynthesis. The Canadian forests and peatlands cover 40% of Canada's land (Kurz & Apps 1999). Because boreal forests cover such a large portion of land, they have the potential to strongly impact the Earth's carbon cycle. Changes in the energy, water and carbon budgets in the boreal forests could impact the world's climate.

One of the most important factors that affect the carbon and energy dynamics in a boreal forest is disturbance. This includes events such as insect infestations, harvest, and fire. These processes greatly impact the age and species composition of the ecosystem. Changes in these characteristics will then impact the water, surface energy, and carbon budgets of the forest as it recovers from the disturbance.

Fire is one of the most important disturbance events in boreal forests. Because boreal forests have unique understory and canopy characteristics and are located in areas that experience dry conditions during summer, fire is frequent. Forest fires often occur as a result of lightning strikes which ignite the dry understory and dry, dead wood. These fires can reach extremely hot temperatures and can be severe enough to burn the surface layers of soil. Because of the large expanse of the boreal forest in Canada, these fires also tend to cover large areas at a given time. It is predicted that increased warm and dry conditions that will accompany climate change will increase the frequency and severity of forest fires in Canada (Flannigan et al. 2005).

1.2 Flux Tower Networks

Studying the impacts that terrestrial ecosystems may have on global climate change poses a difficulty to scientists due to the diversity of species composition, ecosystem structure, and environmental condition on large spatial scales (Margolis et al. 2006). To help understand this issue, networks of flux towers have been developed around the world in many environments to measure and observe CO_2 exchange and energy processes over natural ecosystems. Large networks in Europe, Australia, Canada, and other locations have been created to observe the possible impacts of climate change on greenhouse gas emissions and how carbon, water, and energy exchange change under different circumstances such as disturbance (Margolis et al. 2006).

The international Fluxnet project consists of about 400 flux towers located on five continents with a latitudinal range from 70 degrees north to 30 degrees south. This collaboration started in 1994 to create a network where studies of CO_2, water vapour, and energy could be shared, managed, and archived on a large, international scale. Fluxnet is made up of many regional networks around the world such as Ameriflux, Ozflux, and CarboEurope (Fluxnet 2001).

The Fluxnet-Canada Research Network (FCRN) is one such network that studies the impacts that management practices, disturbance, and climate have on the water, energy, and greenhouse gas fluxes in peatlands, grasslands, and forests across Canada. Established in 2002, the FCRN is a network of 22 sites operated by government agencies and universities throughout Canada. Studies of surface-atmosphere exchanges on a continuous measurement basis using a series of meteorological towers and the eddy covariance method of measuring gas fluxes and chamber measurements of soil-atmosphere interactions are done at each site. Studies of belowground carbon processes and ecosystem respiration, ecosystem structure and function, energy balances and local and regional meteorology are also objectives of the FCRN. This network also studies the impact that disturbances such as insects, harvest, and fire may have on the surface-atmosphere exchanges (Margolis et al. 2006).

The Boreal Ecosystem-Atmosphere Study (BOREAS) was a network that studied boreal forests in Canada using several years of intensive field campaigns, aircraft measurements, and continuous meteorological measurements from 1994 to 1996. This study focused on a 1000 x 1000 km area in Saskatchewan and Manitoba. Mostly, mature sites were studied (Sellers et al. 1997).

As a continuation of BOREAS, in 1996, the Boreal Ecosystem Research Monitoring Sites (BERMS) was formed. These sites include three mature sites (aspen, black spruce, and jack pine) in Saskatchewan ranging from 50 to 150 years of age that were also part of BOREAS. Younger sites including three young post-fire sites of different ages, two harvested jack pine sites, a clear cut jack pine site, and a fen. The three sites used in this project are the three post-fire sites that are part of BERMS (BERMS 2003).

1.3 Previous Fire Chronosequence Studies

The focus of this project is to study the impacts that fire disturbance has on the carbon and energy budgets of boreal forests in Saskatchewan. The sites used for this study are the only fire-disturbed sites that are part of the FCRN. Observations at these sites can then be used by others in the network to make comparisons between the impacts of other disturbances such as harvest on similar sites as well as comparisons to mature sites. These results may also be used to compare to other types of environments including peatlands, grasslands, and other boreal forests. Observations for this project were made over a minimum of three years. Recent studies have observed the energy, water, and carbon budgets at two of the same sites studied here (Amiro, Barr et al. 2006; Amiro et al. 2003; Amiro, Orchansky et al. 2006) or at similar sites (Bond-Lamberty et al. 2004; Chambers et al. 2005; Goulden et al. 2006; Litvak et al. 2003; Liu et al. 2005; Randerson et al. 2006) however, few have based their results on more than one complete year of data.

Amiro, Barr et al. (2006) compared carbon, water and energy budgets of two young post-fire boreal forest sites, a young post-harvest site, and several mature sites located in central Saskatchewan as part of BERMS during the summer of 2001 and 2002. They found that the site burned in 1998 and the site harvested in 1994 were both carbon sources. The site burned in 1989 was a carbon sink, as was the mature jack pine and the mature aspen site. Evapotranspiration levels showed similar results with the site burned in 1989 having the highest evapotranspiration levels followed by the mature aspen site. This was attributed to the large amount of deciduous trees at these two sites.

Chambers & Chapin (2003) studied growing season energy balances of six previously burned black spruce forests in Alaska ranging from 0 – 14 years of age and compared them to the energy balances of unburned sites. Their study showed that albedo initially decreased following fire and then quickly increased as vegetation recovered. Ground heat flux nearly doubled as a portion of net radiation, net radiation decreased following fire, and Bowen ratios increased. They predicted that the changes in energy exchange following fire may have regional climate impacts.

Amiro, Orchansky et al. (2006) studied the energy and water budgets of a collection of mature and recently disturbed boreal sites in Alaska, Saskatchewan, and Manitoba. Summertime Bowen ratios increased following fire and then decreased as the forest aged to approximately 15 years of age. Net radiation during the summer was lower at recently disturbed sites than at mature sites, but was higher than mature sites within one to three years following fire. They predict that if the frequency of forest fire increases within the next decades, radiation and energy balances could result in a cooling effect which could possibly offset the warming caused by climate change. This concept was further developed by Randerson et al. (2006). Randerson et al. (2006) predict that increases in albedo over several decades will have a larger impact on climate than greenhouse gases emitted from forest fire causing a net cooling. Therefore, increases in forest fires in the boreal regions may not increase climate warming (Randerson et al. 2006).

Bond-Lamberty et al. (2004) studied net primary production and net ecosystem production across a chronosequence of black spruce boreal forest in central Manitoba. The sites varied in age from 3 years to 151 years. One component that set this study apart from others is that in addition to studying the carbon gas fluxes, they also took an inventory of the terrestrial carbon pools where carbon can be stored. All the components of carbon were measured for three years. Interannual variability in net primary production was largest in youngest stands and in poorly drained sites. They found that the youngest stands were carbon sources, the intermediate aged stands were strong carbon sinks, and the mature stands were carbon neutral.

Litvak et al. (2003) studied carbon exchange of five boreal forest sites in Manitoba for two growing seasons. Sites ranged in age from 11 years to 70 years. They observed that ecosystem respiration increased with stand age. The youngest stands were found to be weak carbon sinks and the oldest sites were moderate to strong carbon sinks.

1.4 Objectives

The first chapter of this thesis focuses on the energy and water budgets of three boreal forest sites located in central Saskatchewan. The sites have been monitored for a minimum of three years. Each site was burned at a different time (1977, 1989, and 1998). Sites are located 40 km apart from one another and experience similar climate and weather conditions. Because of their close proximity, these sites are ideal for the chronosequence approach to studying the impacts of disturbance on energy and carbon budgets and how they change with time following fire. The energy and water budgets will in turn affect the carbon budget of the sites as well.

The second chapter examines the changes in the carbon budgets at the sites described above. The time that it takes for the CO_2 exchange of a forest to recover after fire gives an indication of how future disturbance could impact the carbon budget and climates on local, regional, and global scales.

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2. SURFACE ENERGY BUDGETS OF THREE POST-FIRE BOREAL FORESTS IN CENTRAL SASKATCHEWAN

2.1 Abstract

The most important stand renewing agent in boreal forests is fire. Changes in surface characteristics, species composition, and ecosystem function following fire will have important impacts on the surface energy and the water budgets of the forest as it recovers and progresses through different stages of succession. Three sites located in central Saskatchewan, Canada were selected for study. Sites were burned in 1977 (F77 site), 1989 (F89 site), and 1998 (F98 site). Sites were located within 40 km of one another and therefore experienced similar climate and weather. Eddy covariance towers were used to measure net radiation (R_n), sensible heat flux (H), latent heat flux (LE). Soil heat flux (G), soil temperature, soil moisture, and supporting meteorological conditions were also measured. Sites were monitored from spring of 2001 (F89 and F98) or 2003 (F77) until the winter of 2005. Measurements were taken throughout the entire year. R_n and H were similar at all three sites by 2003. LE best described the changes that occurred in forests with time following fire. The LE values at the more mature sites were approximately 2 MJ m^{-2} d^{-1} higher than at F98 in mid-summer. This is likely due to the abundance and maturity of the vegetation at the older sites in comparison to F98 where the vegetation is small and sparse. LE also displayed decreases at F77 and F89 that took place during drought years. F98 had the highest soil heat flux in both the summer and winter months of most years. Most energy was partitioned into sensible heat flux which was illustrated by the large number of positive Bowen ratio values. It is important to

understand how time following fire changes the energy balance to predict the impacts that possible increases in forest fire frequency that is expected to accompany climate change may have on local, regional, and global climates.

2.2 Introduction

Forest fires are major stand renewing agents in boreal forests. Forests in high latitudes experience dry conditions during the growing season and have vegetation and understory characteristics that create ideal conditions for forest fires to occur. With the onset of climate change, it is expected that the frequency of forest fires will increase as climate change will enhance the warm, dry conditions in the boreal forest biome (Flannigan et al. 2005). Because the boreal forest covers such a large land area, it has a great potential to impact regional and global climate.

Forest fire has a significant impact on the surface energy budgets of forests as fire changes surface characteristics, species composition, and ecosystem structure (Liu et al. 2005). As the forest progresses through different stages of succession, the vegetation will change, and thus the energy budget will be affected. Chapin et al. (2000) propose that without vegetation changes that follow fire, warming caused by climate change could result in decreased moisture availability, increased sensible heat flux and could therefore lead to an increase in the depth of the planetary boundary layer which would further increase warming. Disturbances in boreal forests change the energy budget and can cause significant climatic impacts (Chambers & Chapin 2003). The removal of vegetation by forest fire will have considerable impacts on the water budget by affecting water runoff, infiltration of water into the soil, and transpiration rates (Valeo et al. 2003). The absence of vegetation immediately following fire will also greatly decrease the amount of precipitation interception (Valeo et al. 2003). Water availability will determine when stomatal closure and opening occurs. Stomata activity will determine the energy and water budgets by controlling water exchange and surface temperatures of vegetation. Stomata activity will also determine plant transpiration rates. Chambers and Chapin (2003) found that after 12 years following fire in Alaska, evapotranspiration rates were higher than at older mature sites, likely due to the increase in deciduous vegetation which has a deeper rooting system and transpires more rapidly than coniferous species. This is evidence that fire will have large impacts on the energy budget.

Fire also changes the albedo of the earth's surface (Amiro, Orchansky et al. 2006; Chambers & Chapin 2003). Chambers and Chapin (2003) found that sites greater than five years old had albedo greater than those of unburned sites. The energy budget has a significant impact on other cycles in a forest ecosystem including the carbon cycle. Plant canopy characteristics such as albedo that affect the energy budget will also affect the carbon budget by controlling the environment and rates of photosynthesis. As surface temperatures increase, an increase in biomass production is expected to occur (Van Kooten & Arthur 1989) and will thus change the carbon budget by increasing the amount of carbon that is present in vegetation. Boreal forests are important carbon sinks (Dixon et al. 1994). However, following fire, forests become carbon sources (Kurz & Apps 1999). The forests' role in carbon sequestration will become more important as we try to reduce the impacts of climate change. If the incidence of forest fire increases, we will see an increase in carbon sources which could potentially amplify the effects of climate change. In the future, it is predicted the energy budgets will work with the changing carbon cycle by altering disturbance frequency, temperature and precipitation on a regional scale (Chapin et al. 2000).

Energy fluxes are also good indicators of what is happening to the factors that affect them such as climate and vegetation. Recently, researchers have begun to study the energy budgets of boreal forests outside of the growing season (e.g., Amiro et al. 2003; Amiro, Orchansky et al. 2006; Chambers et al. 2005). Amiro, Orchansky et al. (2006) studied a variety of forest sites in Saskatchewan, Manitoba, and Alaska of varying ages. The study focused on daily totals of summer energy fluxes over two years. They found that net radiation measured in the summer was lower in successional, disturbed forests than in mature forests but was higher one to three years after fire. They also found that daily Bowen ratio at mature sites was much lower than at sites immediately following fire. Few studies examine the energy budget over a period of multiple years or have focused on forests of intermediate age during the earlier stages of succession (Chambers & Chapin 2003). It is important to study changes that take place over time as forests recover from fire. Outside the growing season, important changes in albedo and in interception of radiation and precipitation will have significant impacts on the energy budget. Liu et al. (2005) found that annual sensible heat fluxes were much lower at young sites compared to an 80-year site. They also found that fire reduced annual evapotranspiration immediately following fire. It is also critical to study how these forests respond to different climate conditions such as drought or wet years.

The objective of this study was to observe the changes in energy fluxes and the energy budget of three boreal forest sites of different ages in central Saskatchewan over several consecutive years. The comparison across this chronosequence will give an indication of how forests recover from fire and how succession will impact the energy budget. Data from the entire year were used in order to obtain a good understanding of the dynamics of young post-fire forests on an annual time-scale. Using a combination of the eddy covariance technique and other field measurements, the energy budget for each site was quantified.

2.3 Methods and Materials

2.3.1 Site Descriptions

Three sites located in central Saskatchewan in the southern boreal forest were selected for study. All three sites were selected for their relatively uniform topography and for their proximately to one another. The sites are located within approximately 40 km from each other. The sites were burned in 1977, 1989, and 1998, respectively. All three sites have a Dystric brunisol soil type with a sandy loam texture.

The fire that occurred in 1977 (F77 site) was the result of a lightning strike that burned approximately 8,000 ha. Following the fire, the area was allowed to regenerate naturally. The vegetation present before the fire consisted of black spruce (*Picea mariana*) and jack pine (*Pinus banksiana*) trees. After the fire, the new vegetation consisted of a variety of tree species with the dominant species being jack pine and black spruce. The dominant understory species is small black spruce trees. There is also a small amount of trembling aspen (*Populus tremuloides*).

The fire near Montreal Lake (F89 site) resulted from a human-caused fire in May, 1989 which covered approximately 13,500 ha. In 1990, parts of the site were aerially seeded with jack pine but the majority of the site was left to regenerate naturally. Prior to the fire, parts of the site had been logged, and the slash residues left were burned in the fire. The dominant vegetation before the fire was jack pine and black spruce. The vegetation now consists of jack pine, black spruce, and trembling aspen. Most of the dead trees have fallen. There is more trembling aspen present at this site than at the other sites.

The youngest site (F98 site) in this study was burned in July, 1998 by a severe fire resulting from a lightning strike. The fire covered an area of 1,700 ha, killing all trees and burning the top layer of soil. Prior to the fire, the dominant tree species was jack pine and black spruce. Today, the dominant tree and understory species is jack pine, although black spruce is also abundant but growing more slowly. There are also patches of actively growing trembling aspen. Many dead trees are still standing and of the dead trees that have fallen, many are suspended above the ground and are decomposing slowly.

2.3.2 Meteorological Measurements

Supporting measurements at the top of each tower included temperature, wind direction and wind velocity measured using a CSAT3 sonic anemometer (Campbell Scientific Inc., Logan, UT, U.S.A.). Incoming photosynthetically active radiation (PAR) was measured using quantum sensors (LICOR Inc., LI190, Lincoln, NE, U.S.A.). Wind direction and speed were also measured with a wind monitor (RM Young, Model 05103, Traverse City, MI, U.S.A.).

Supporting meteorological measurements were recorded using Campbell Scientific (Logan, Edmonton, AB, Canada) 23X and 10X dataloggers at a frequency of between 1 and 10 Hz at all sites depending on the sensor. High frequency measurements were averaged over a half-hour period. Data were downloaded from a remote location using a cellular phone connected to the datalogger.

Rainfall was measured using a tipping bucket rain gauge (Model TE525M, Campbell Scientific Inc) located near the tower at a height of 1.5 m. Snow depth was measured using a SR50 sonic ranging instrument (Campbell Scientific Inc). A snow survey was also taken in 2003, 2004, and 2005 during November and December. Volumetric soil moisture for depths 0 cm to 30 cm were measured using three CS616 soil moisture probes (Campbell Scientific Inc) inserted vertically into the soil.

2.3.3 Energy Budget Flux Measurements

Towers were erected at each site with heights of the towers dependent on canopy height. Monitoring began on July 4, 2001 and May 11, 2001 at F89 and F98 sites, respectively. Flux measurements began at F77 on July 16, 2003. Sites had at least 1 km of fetch in all directions with the exception of F98. At F98, fluxes obtained between 0 and 90 degrees from north were excluded because of harvesting that took place following the fire at distances greater than about 200 m. All sites were powered using batteries charged by solar panels. Measurements were recorded by a 23X Campbell Scientific data logger located at the base of the towers. Further site-specific details are given in Table 2.1.

Table 2.1 Site characteristics of three post-fire boreal forest sites in central Saskatchewan located near Prince Albert National Park.

Site	**F77**	**F89**	**F98**
GPS location	N54.48503° W105.81757°	N54.25392° W105.87775°	N54.09156° W106.00526°
Elevation (m)	563	540	548
Year Burned	1977	1989	1998
Measurement Period	2003 - 2005	2001 - 2005 (Oct)	2000 – 2005
Measurement Height (m)	7.4	5.2	(7.7*) 20
Dominant Tree Species	Jack pine	Jack pine	Jack pine, Black spruce
Dominant Understory Species	Black spruce	Black spruce	Jack pine
Average tree height (m)	6	5	18 (dead), 1 (live) **
Average shrub height (m)	2.4	1.6	1.1
Soil Texture	Sandy loam	Sandy loam	Sandy loam
Soil Classification	Dystric Brunisol	Dystric Brunisol	Dystric Brunisol
Soil Organic layer depth (cm)	2.7	4	1
Soil Horizon A depth (cm)	6.7	5	6
Regeneration	Natural regeneration	Jack pine aerially seeded in 1990 Most was natural regeneration	Natural regeneration
Leaf Area Index (TRAC)	2.8***	3**	1.3***

* Measurements were taken at height of 7.7m until August 15, 2002. As of August 16, 2002, measurements were taken at a height of 20 m.

** Measurements taken in 2001 from Amiro, Barr et al. 2006

*** Measurements also reported in Chen et al. 2006

2.3.3.1 Radiation

Net radiation (R_n) was measured at all sites using a four-component net radiometer (Kipp and Zonen CNR1, Delft, The Netherlands). Radiometers were located at the top of the tower at each site. Prior to August 16, 2002 at F98, a two-component radiometer was used (Middleton CN1, Melbourne, Australia). Albedo was calculated using daily totals (including both night and day measurements) of incoming and outgoing shortwave radiation to calculate four-week averages. The four-week average of outgoing shortwave radiation was then divided by the four-week average of incoming shortwave radiation.

During 2001 and the beginning of 2002, flux measurements were taken at F98 at a height of 7.7 m. This means that measurements were taken above the new growth canopy, but they were taken from below the canopy formed by the standing dead trees. Partial shadowing gives net radiation above the new growth only for this period.

2.3.3.2 Sensible and latent heat flux measurements

Sensible and latent heat flux were measured using eddy covariance. Gas fluxes were measured using an open-path LI7500 (LICOR Inc. Lincoln, NE, U.S.A.) infrared-gas analyser (IRGA). Three-dimensional wind velocities and air temperature were measured using a CSAT3 (Campbell Scientific Inc.) sonic anemometer-thermometer. Flux instruments at F98 were placed on a 30 cm triangular tower until August 16, 2002 at a height of 7.7 m. During this time, a closed-path LI6262 IRGA (LICOR Inc.) was used to measure latent heat flux density (LE). This included an 8 mm Bevaline IV tube within the axis of the sonic anemometer. Air was then pulled through 3 m of tubing by a pump at a rate of 5 L min^{-1} and then pushed through a 1 μm particle filter and then into the closed-path analyzer. The gas analyzer was calibrated at least monthly. Corrections in measurements were made for temperature and pressure changes, any lag that resulted from the tubing, and for frequency response (Amiro, Barr et al. 2006). The CSAT3 and gas analyzer signals were sampled at a rate of 10 samples s^{-1}, and the cross-products were stored every 30 minutes. Covariances were calculated from these cross-products. Wind velocities were rotated so the mean vertical velocity was zero over the 30-minute period (Tanner & Thurtell 1969). Due to the flat topography of these sites, the rotation was usually minor.

On August 18, 2002, a scaffold tower was erected at F98 to replace the triangular tower and the flux instruments were moved to a height of 20 m. The closed-path LI6262 IRGA was replaced with an open-path LI7500 IRGA at this time.

2.3.3.3 Soil heat flux measurements

Soil temperatures were measured using chromal-constantan thermocouples at depths of 0.02 m, 0.05 m, 0.1 m, 0.2 m, and 0.5 m with three replicates at each depth. Soil heat flux plates (Thornthwaite Model 610, Pittsgrove, NJ, U.S.A.) were used to measure soil heat flux (G) at a depth of 0.02 m. There were three soil heat flux plates at each site. Soil heat flux at F89 and F98 were calculated by taking an average of the three soil heat flux plates. At F77, only two soil heat flux plates were operational at a given time and the final average was only taken from two of the three plates. On a daily basis, heat storage in the upper 0.02 m of soil is very small and was therefore not included in the estimates of G.

2.3.3 Data Analysis and Quality Control

2.3.3.1 Calculations of fluxes

Virtual sensible heat flux measured by the CSAT3 was changed to actual sensible heat flux by using water vapour measurements from the LI7500. Density correction was used for H to account for differences in density due to air temperature and water content. Corrections from Webb et al. (1980) were used in the calculation of LE. This correction makes allowances for changes in the density of parcels of air that may affect latent heat flux measurements (Webb et al. 1980). Storage terms were not included in the analysis of H and LE over daily time periods because storage is small at these time scales. Fluxes were analyzed using 4-week averages of daily totals. Data analysis was done using MATLAB computing language (Version 7.0.0.19920(R14) The MathWorks Inc., Natick, MA, U.S.A.).

2.3.3.2 Quality control

Data were filtered to reject flux points obtained during periods when measurements would not be reliable. Rain and snowfall events occasionally created conditions when instruments would not perform properly. Data were filtered during these events by calculating the number of instances where the CSAT3 readings were greater than 1000. Friction velocity (u_*) was used as a threshold to only include fluxes where sufficient turbulence was present for measurement. The u_* threshold was calculated by finding nighttime measurements when PAR was less than 0 μmol m^{-2} s^{-1}. The remaining data were then bin-averaged in 10 even bins with equal sample numbers. The threshold was then calculated to be 80% of the average CO_2 flux of the last three bins. The threshold was then rounded up to the nearest 0.05 m/s value (Amiro, Barr et al. 2006). When sites were examined on an individual basis one year at a time, a range of u_* thresholds between 0.2 m s^{-1} and 0.3 m s^{-1} were calculated. For analysis and comparison purposes, a common threshold of 0.25 m s^{-1} was used at all sites for all years as our best estimate. When u_* was below this threshold, energy fluxes were rejected because of insufficient turbulence. Threshold values for energy fluxes were also chosen to reject data that was beyond normal observations. Thresholds for sensible heat flux were -150 W m^{-2} for the lower limit and 500 W m^{-2} for the upper limit. Latent heat flux thresholds were -15 W m^{-2} and 500 W m^{-2}. Thresholds of -1000 W m^{-2} and 1000 W m^{-2} were used for R_n values. Eddy flux data at F98 obtained from 0 - 90° north were rejected because the footprint was in a harvested sector.

2.3.3.3 Gap filling

Small gaps in H and LE (less than four half-hour periods) were filled using linear interpolation. To fill larger gaps in H, a linear regression was made between H and R_n-G to fill gaps based on a 240-point moving window (Amiro, Barr et al. 2006). The window was moved at increments of 48 points at a time.

Nighttime and winter values of LE obtained during periods where u_* was below the threshold value were set to zero for calculating daily totals. Gaps in LE that occurred during the growing season during the day were filled using the same technique as for H. Outside of the growing season, an average for the same half-hour on the previous five days and the following five days was calculated. This was used outside the growing season because during this time, LE is not well correlated with R_n-G (Amiro, Barr et al. 2006).

2.4 Results and Discussion

The maximum amount of data that required gap filling at F77 was 36% in 2003 for LE to 35% for H in 2004. The maximum percentage of data that was gap filled at F89 was 40% for H and LE in 2003 and the minimum percentage was 34% for H in 2004. At F98, a range from 32% of the LE data in 2003 was gap filled to a maximum of 42% which was observed for H in 2004. Both H and LE had similar levels of gap filling at all the sites with no flux having consistently more gap filling used. Gap filled data accounted for less than 10% of G at all sites for all years.

2.4.1 Precipitation

2.4.1.1 Rainfall

Precipitation measurements at all sites were compared to the annual total rainfall measurements taken at Waskesiu Lake weather station (Table 2.2). Both the Waskesiu Lake weather station and measurements from our sites showed that the years 2001 to 2003 were the driest years during the duration of this study compared to 2004 and 2005. The wettest years of the study were 2004 and 2005 at all sites. F77 was the wettest in 2005. With the exception of 2005, total yearly rainfall was higher at F89 than F98 for all years.

2.4.1.2 Snowdepth

Snow surveys were conducted at all sites at the end of 2004 and the beginning of 2005 (Neumann et al. 2006). Snow depths measured using the Sonic Range SR50 were taken near the tower. Comparisons of snow depth measurements are given in Table 2.3. The deepest snow accumulations were observed at F77 during the winter of 2005 when average snow depths were calculated for the months of January and February. This was also observed in the snow survey results. Snow depths obtained using snow survey measurements were at least 5 cm greater at F77 than at the two younger sites. This is somewhat unexpected because the vegetation at F77 is the most mature and the canopy will therefore intercept more snow. It was noted that significant snow interception took place at F77 where only minor interception occurred at the other sites (Neumann et al. 2006). Fixed point measurements taken with the sonic ranger SR50 showed good agreement with the snow survey results at all sites (Neumann et al. 2006).

Table 2.2 Annual precipitation and annual evapotranspiration values for three post-fire boreal forest sites in central Saskatchewan. Measurements and 30-year normals are from Waskesiu Lake were obtained from Environment Canada online archives.

	Site	30-Year Normal	2001	2002	2003	2004	2005
Average Annual Rainfall (mm) Waskesiu Lake		336.1	214.2	318.4	284.8	476.3	450.4
Total Annual Precipitation (mm) Waskesiu Lake		467.3	282.5	446.7	329.6	612.8	528.4*
Total Annual Rainfall (mm)	F77					440.6	559.2
	F89				236.6		490.7
	F98				276.7	502.1	423.5
Annual Evapotranspiration (mm)	F77					372.9	438.3
	F89				325.4	367.5	408.4
	F98				288.4	262.2	345.9
Maximum Snow Depth (Sonic Ranger)(cm)	F77					66.0	68.0
	F89			75.0	75.0	43.0	49.0
	F98			22	37	46	44
Average Snow Depth January and February (cm)**	F77					41.0	55
	F89		N/A	N/A	27.2	29	42
	F98		N/A	N/A	27.4	30.0	41

* Data for January and February 2005 unavailable
** From Sonic Ranger

Table 2.3 Comparison of snow depth measurements using a SR50 Sonic Range instrument and a snow survey taken three times per winter at three boreal forest sites in central Saskatchewan. Survey results are from Neumann et al. (2006) and are based on 10 depth measurements for each sampling visit. Five water equivalent measurements were taken at each visit.

Site	Date	Mean Snow Depth (cm)*	Sonic Ranger Depth (cm)	SD Snow Depth (cm)*	Mean Measured Water Equivalent (mm)*	Mean Snow Density (g cm^{-3})*	Mean Water Equivalent Using Mean Density (mm)*
F77	15-Dec-04	25.0	26.0	6.0	53.5	0.204	51
	15-Feb-05	50.3	56.0	7.0	121.6	0.237	119.3
	16-Mar-05	53.4	61.4	6.7	119.6	0.212	113.3
F89	15-Dec-04	19.7	19.0	6.4	43.3	0.177	34.9
	15-Feb-05	45.6	43.0	7.5	90.5	0.208	94.8
	16-Mar-05	42.8	42.6	9.8	99.6	0.235	100.6
F98	15-Dec-04	20.9	18.4	3.3	39.6	0.177	36.9
	15-Feb-05	45.6	42.3	3.5	92.1	0.202	92.4
	16-Mar-05	40.9	37.5	5.8	91.9	0.22	90

SD: Standard deviation
* from Neumann et al. 2006

2.4.2 Radiation

2.4.2.1 Albedo

The albedo showed obvious differences among sites (Figure 2.1a). F89 had a higher albedo during the summer compared to the other sites, especially during 2003. Winter albedo was greatest at F98 due to less tree canopy so that the snow-covered ground surface was more easily seen. Albedo at all sites reached maximums ranging from approximately 0.3 to 0.4. These values are higher than findings by Arain et al. (2003) who found that maximum winter albedo in a mature conifer forest was near 0.25. Albedo decreased drastically in summer to values near 0.1.

Charred, black surfaces present following severe fires can cause lower albedo than that of an unburned forest (Amiro et al. 1999; Chambers & Chapin 2003; Liu et al. 2005). Due to the abundance of standing dead trees that are covered in black carbon at F98, we can expect the albedo of this forest to be lower than the other sites where the dead trees have fallen. Mature forests also have a more complete canopy than the younger sites and would therefore have more vegetation to intercept snow which would result in higher winter albedo. Although it snowed more at F77 (Neumann et al. 2006) more exposed snow would be present at F98, explaining the high albedo.

F77 is almost entirely coniferous with fewer deciduous trees than F89. Higher summer albedo at F89 can likely be explained by the abundance of deciduous trees at this site compared to the other two sites that are dominantly coniferous. Coniferous forests often have much lower albedo during the summer than deciduous forests (Betts & Ball 1997; Chambers & Chapin 2003; Liu et al. 2005). Because of the albedo differences, species mix dictates the albedo of a site. The more coniferous trees that are present, the lower the albedo of the site.

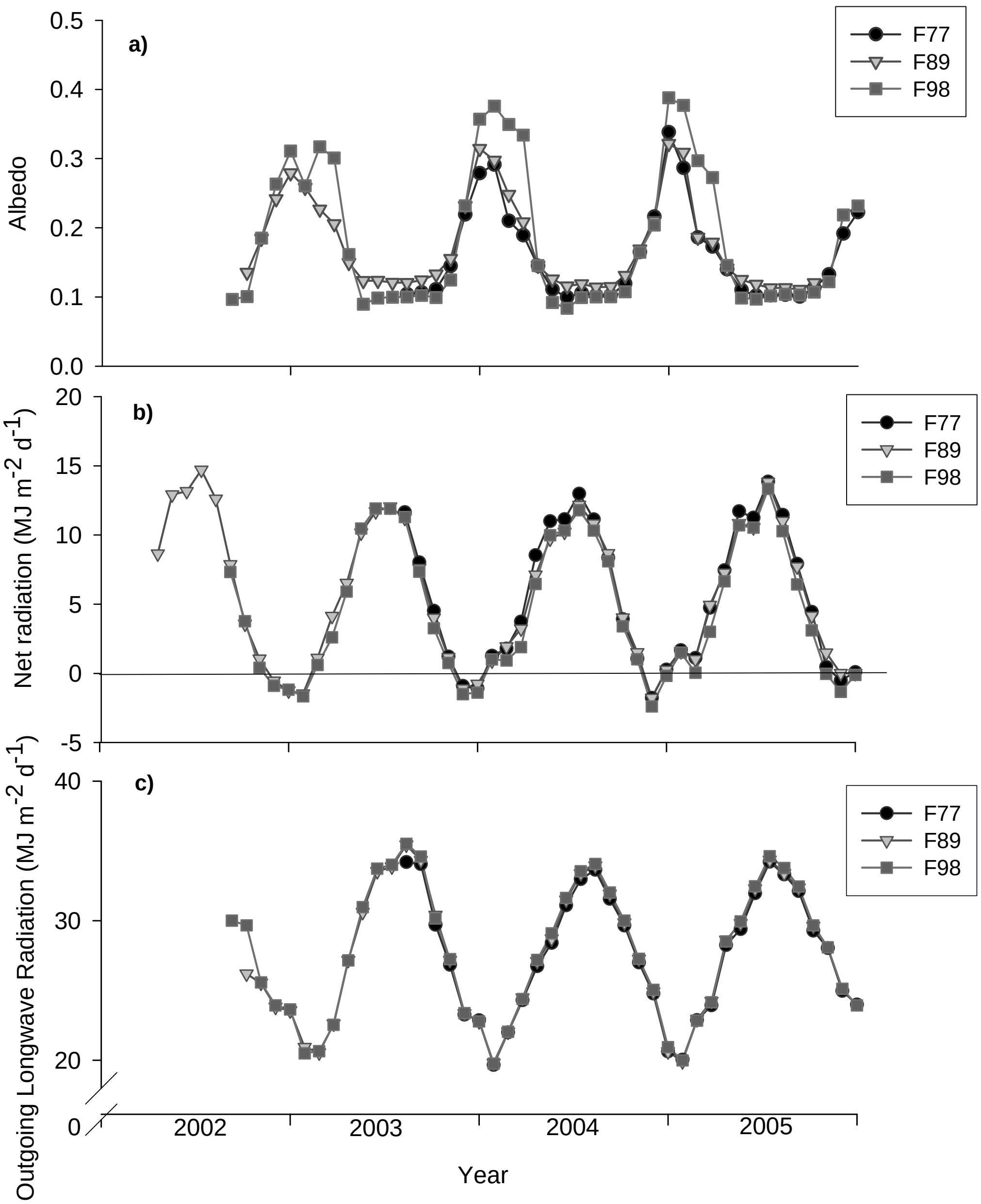

Figure 2.1 a, b, c Albedo, net radiation, and outgoing longwave radiation of three post-fire boreal forest sites in central Saskatchewan burned in 1977 (F77), 1989 (F89), and 1998 (F98). Measurements taken from 2002 to 2005. Points represent 4-week averages of daily totals for radiation and an average for albedo.

2.4.2.2 Net radiation

Net radiation varied little among sites (Figure 2.1b). There was slight separation between sites during the summer of 2004 with F77 having the highest R_n values, but even then, it was only slightly higher than those of F89 and F98. When sites were compared using daily totals, slopes of the regression line in a comparison between F89 and F98 showed that F89 was 0.93 ($r^2 = 0.82$) to 0.95 ($r^2 = 0.95$) of F98 with the largest slope occurring in 2004. Comparisons between F77 and the younger sites yielded slopes close to 1 during 2004 and 2005. Slopes near one indicate that R_n is not different among sites.

When R_n was separated into its components, it was unclear which component was responsible for the similarities in R_n. Incoming shortwave radiation was similar at all sites. Outgoing shortwave radiation was the only radiation component which varied among sites. R_n was higher in 2005 than in 2003 and 2004 at all sites compared to previous years. This can be explained by the dry conditions experienced in 2003. Drought-like conditions may have caused water stress for the vegetation which would result in stomata closure in an attempt by the vegetation to conserve water. The stomata closure could then lead to an increase in the surface temperature of the plants and would have also increased longwave radiation (Figure 2.1c). However, in forests where dead trees remain standing, there is not a large increase in solar radiation reaching the forest floor. Without shortwave radiation to heat the soil surface, there would not be a large increase in outgoing longwave radiation. Amiro et al. (1999) found there was not a significant change in R_n following fire in a forest stand where dead trees were still standing. This would explain why there is little difference in R_n at F98 where many dead trees are standing compared to F77 and F89 where most of the standing trees are living.

These results do not agree with other research that found that fire increased R_n. For example, annual R_n at a forest site in Alaska showed that fire decreased R_n as a result of increased snow cover in spring (Liu et al. 2005). The increases in R_n during the spring may be explained by a difference in snow melt during the spring. Following fire, there will be more snow cover due to the lack of vegetation to intercept it. This will cause a decrease in annual R_n and will decrease the shortwave radiation being absorbed by the ground surface (Liu et al. 2005). Our four-week averaging may not show these spring-time differences.

2.4.2.3 Longwave radiation

Incoming longwave radiation varied very little among sites or among years. Outgoing longwave radiation (Figure 2.1c) was very similar among sites during the summer months. The outgoing longwave radiation during the winter varied little from year to year. However, over a shorter time period of three days, outgoing longwave radiation varied more from site to site. Figure 2.2 shows outgoing longwave radiation over three days in mid-July. Outgoing longwave radiation is greatest at midday and is often the greatest at F98. Outgoing longwave radiation was often the lowest at F77. When studied over longer time periods, these differences become less apparent. Outgoing longwave radiation is driven by surface temperatures. Surface temperatures are expected to be higher at F98 due to the amount of charred black debris compared to the other sites. Higher surface temperatures at F98 may also occur as a result of more radiation penetration to the surface because of less canopy cover and the dead tree canopy reduces wind near the surface so there is less convective cooling.

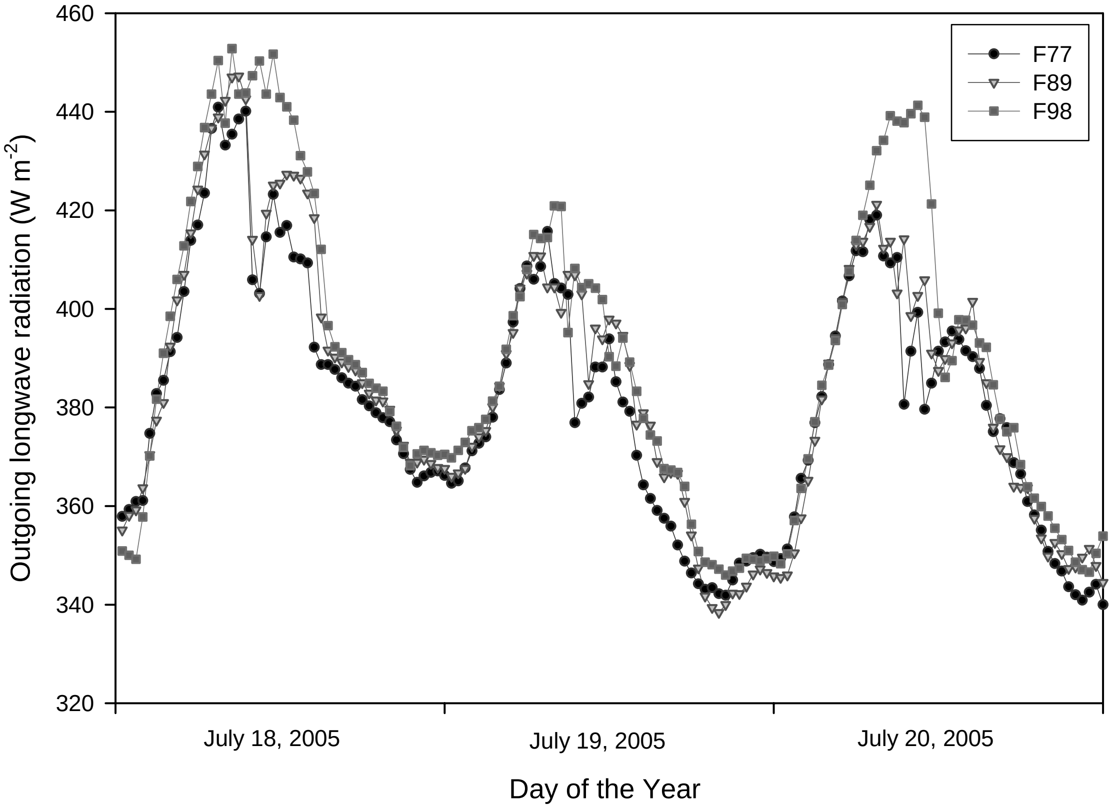

Figure 2.2 Outgoing longwave radiation at three post-fire boreal forest sites burned in 1977(F77), 1989 (F89), and 1998 (F98) in central Saskatchewan. Points represent half-hourly values from July 18 - July 20, 2005.

The long-term observations indicate that there is very little variation in longwave radiation among sites. This is somewhat unexpected. One would expect that differences in surface vegetation would impact the warming of the surface and would thus affect the longwave radiation being emitted from the vegetation. This was not observed over the long term. This is also surprising because one would expect that drought conditions experienced in 2003 would yield warmer surface temperatures and therefore increase the outgoing longwave radiation. The lack of variation may also be due to compensation between night and day over long term measurements. Over longer time periods, the differences in longwave radiation during the day and night may cancel each other out.

2.4.3 Sensible Heat Flux Density

Sensible heat flux density (Figure 2.3) at F89 during 2002 shows a high value of H at the peak point in the summer based on 4-week averages. The maximum observed H was in the summer of 2002 at F89 when H was approximately 6.8 MJ m^{-2} d^{-1}. Peak H values then decreased at all sites in 2004. In 2004 which was the year that all three sites experienced the lowest peak H, F77 and F89 had similar peak H values around 7 MJ m^{-2} d^{-1}. Peak H at F98 was slightly lower in 2004 with a maximum value of 5.8 MJ m^{-2} d^{-1}. In 2005, peak H increased slightly at F77 and F89 to levels near 7.5 MJ m^{-2} d^{-1}. Peak sensible heat flux density at F98 decreased in 2005 compared to the previous years to levels near 5.5 MJ m^{-2} d^{-1}. Comparisons of H between F98 and F89 showed that the similarity between sites increased with time after fire with F89 being consistently higher than F98. In 2003 and 2004, regression between F89 and F98 using daily totals was similar with slopes of 0.85 (r^2 = 0.67) and 0.95 (r^2 = 0.67), respectively. Comparison between F77 and F98 showed reasonable linear regression with slopes ranging from 0.86 (r^2 = 0.78) to 1.06 (r^2 = 0.62) with F77 being larger than F98. Similar values were observed with the comparison between F89 and F77 with F77 values being larger than F89.

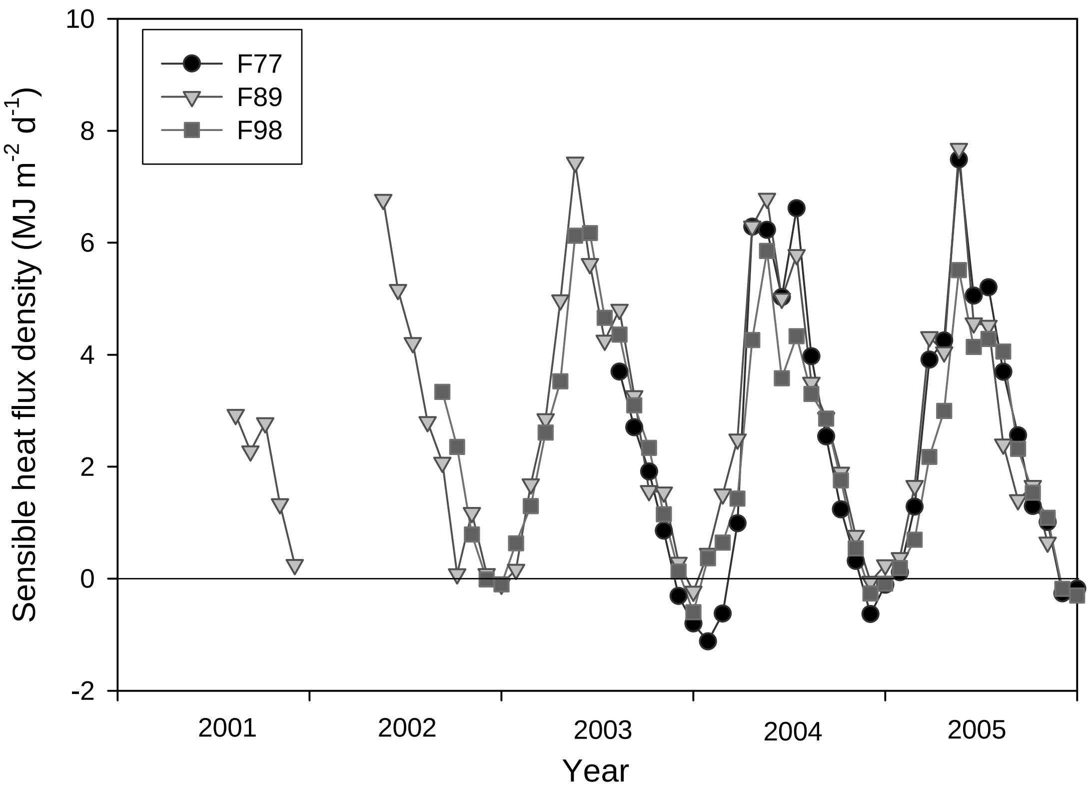

Figure 2.3 Sensible heat flux density at three post-fire boreal forest sites burned in 1977 (F77), 1989 (F89), and 1998 (F98) in central Saskatchewan from 2001 to 2005. Points represent 4-week averages of daily totals.

Peak sensible heat flux densities at F89 prior to 2004 are high which would suggest that during these drought years relatively, more energy is put into warming the atmosphere. This is an expected result because water would be limited in this situation and therefore, transpiration and evaporation would be reduced. During the summer of 2004 when all sites experienced wetter conditions, the sensible heat flux decreased. As more water becomes available, more energy is partitioned into evapotranspiration and less into sensible heat fluxes, therefore H declines as a result. H values at all three sites are very similar during 2003, 2004, and 2005. F89 had a slightly higher H during 2003 but became almost equal to F77 during the next two years.

2.4.4 Latent Heat Flux Density

The peak summer latent heat flux density at F77 decreased slightly between 2003 and 2004 and increased between 2004 and 2005 (Figure 2.4). At F89, peak LE fluxes decreased slightly between 2001 and 2002 and decreased by 2 MJ m^{-2} d^{-1} between 2002 and 2003. There was also a small decrease between 2003 and 2004 followed by a large increase of approximately 2 MJ m^{-2} d^{-1} in 2005 at F89. LE at F98 increased every year from 2001 through to 2003, decreased slightly in 2004 by about 1 MJ m^{-2} d^{-1} and then increased again in 2005. The largest increases at F98 took place between 2002 and 2003 and between 2004 and 2005.

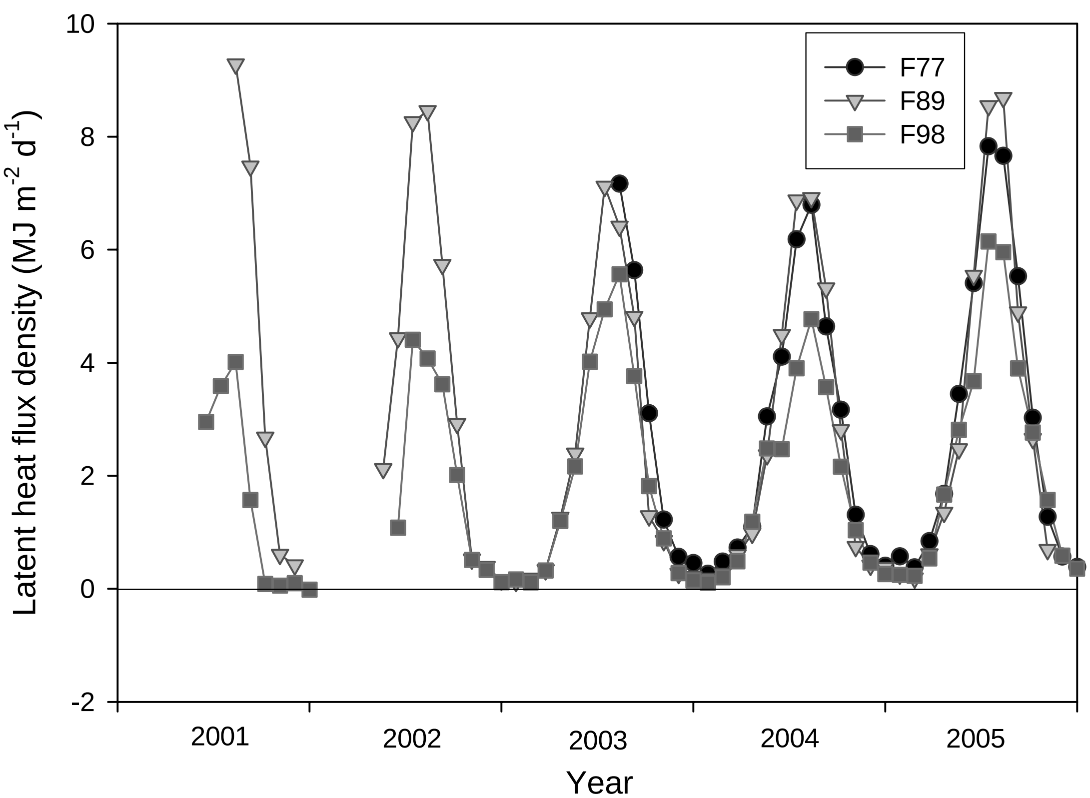

Figure 2.4 Latent heat flux density at three post-fire boreal forest sites burned in 1977 (F77), 1989 (F89), and 1998 (F98) in central Saskatchewan. Points represent 4-week averages of daily totals.

LE at F98 was almost half of the LE at F89 during the summers of both 2001 and 2002. For the years 2003 to 2005, F77 and F89 had LE fluxes that were almost exactly the same. During 2003 and 2004, F77 and F89 had maximum LE values that were between 1 and 2 MJ m^{-2} d^{-1} higher than those observed at F98. Maximum LE values in 2005 at F77 and F89 exceeded those at F98 by almost 2 MJ m^{-2} d^{-1}. Annual evapotranspiration values (Table 2.2) were consistently higher at F77 for all years. Annual evapotranspiration values were also highest at all sites in 2005, demonstrating the wet conditions. F98 had the highest volumetric soil water content (SWC) (Figure 2.5) in 2003 with a maximum value of approximately 0.25 m^{-3} water m^{-3} soil which is illustrated in Figure 2.5. F77 had the highest soil moisture in 2004 and 2005 with levels reaching approximately 0.3 m^{-3} water m^{-3} soil. Slopes for the comparison of LE between F98 and F89 were greater than unity for all years with F98 being less than F89. The same trend was observed with the comparison between F98 and F77. However, the regression line for the comparison showed F77 was greater than F89 as indicated by the smaller slopes with values between 0.83 ($r^2 = 0.85$) and 1.0 ($r^2 = 0.86$). These values hovering close to one indicate that LE is very similar at all sites.

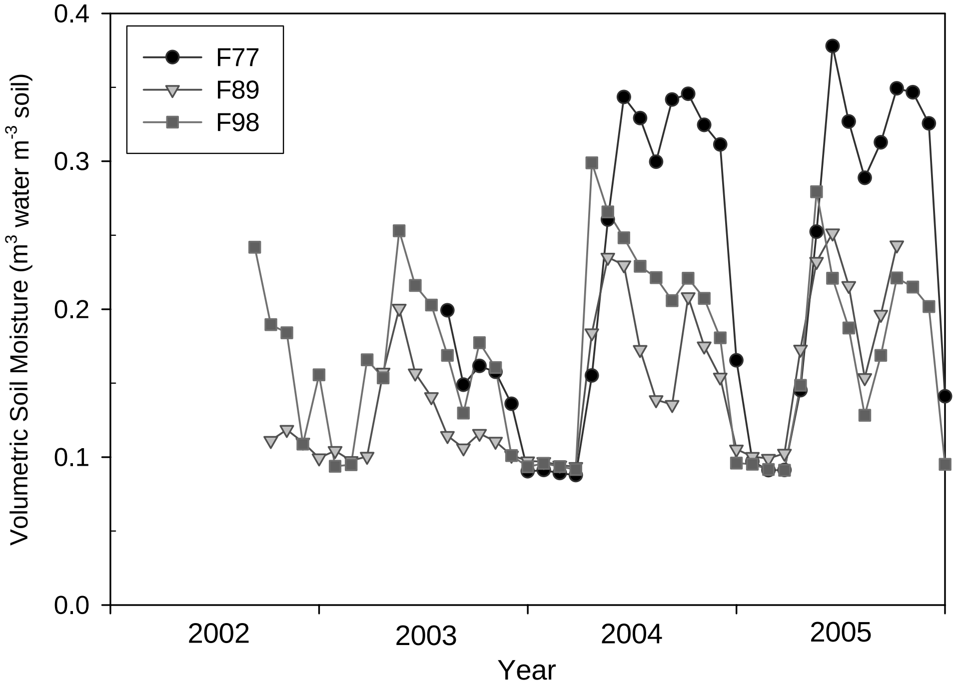

Figure 2.5 Volumetric soil moisture content at three post-fire boreal forest sites burned in 1977 (F77), 1989 (F89), and 1998 (F98) located in central Saskatchewan from 2002 to 2005. Points represent 4-week averages of daily totals.

The annual trends of LE at these sites illustrates the impacts that drought can have on the LE of recovering forests. The decreases observed at F89 in 2003 and 2004 are due to the drought that occurred in 2003. Although there was a drought, F98 still experienced increases in LE while still maintaining LE lower than the other sites. This is because the vegetation at this site was still growing quickly and would continue to transpire somewhat, regardless of the drought. Wetter conditions were experienced at all sites in 2004 and 2005. These conditions were apparent when studying SWC. SWC increased drastically at F77 from the year 2003 to 2004 and 2005. SWC at F89 and F98 also increased during 2004 and 2005 however, not to the same extent that F77 increased. This is likely due to the amount of snow at F77 compared to the other two sites. F77 received more snow than F89 and F98 during 2004 and 2005. During 2003, SWC was much lower at all sites. This increase in available water was not observed in LE fluxes in 2004, likely because the water-stressed vegetation had not yet recovered from the previous year. The increases observed at all sites in 2005 show that as available water increases, evapotranspiration will also increase provided that the vegetation has recovered.

Differences in LE among sites can likely be explained by the maturity and abundance of the vegetation at each site. When compared to more mature sites, F89 had the highest LE values for 2001 and 2002 (Amiro, Orchansky et al. 2006). F98 has vegetation that is short and sparse in comparison to the vegetation at F89. This short vegetation transpires less during the growing season than the older and more abundant vegetation that is present at F89. Short vegetation will also allow more sunlight to penetrate deeper through the canopy which will allow for more soil evaporation to occur. Heijmans et al. (2004) found that latent heat fluxes were strongly affected by light conditions and that at sites where more sunlight reached the surface, more evaporation took place. The amount of deciduous trees at a given site will likely cause a dramatic increase in transpiration at the peak point of the growing season, during and after leaf out. This was supported by work by Arain et al. (2003) which demonstrated that peak LE values were observed in July as a result of high vapor pressure deficits observed during this time. It is also likely that a dramatic decrease in transpiration will occur during the fall when the leaves begin to senesce. At a 15-year-old site in Alaska, LE increased by more than 50% following leaf emergence, but had lower evapotranspiration values than a mature forest during the spring and fall due to the lack of leaves on deciduous trees (Liu et al. 2005).

Soil moisture can have major implications on the LE flux. It was found that there was no relationship between soil moisture and LE along a transect running from a tundra to a forest area (Beringer et al. 2005). Increases in LAI will reduce the amount of evaporation from the soil surface while increasing the amount of water transpired from the vegetation (Beringer et al. 2005). This is visible at F89 which has the highest LAI (Table 2.1). This site also persistently has the highest levels of LE. F98, which has the lowest LAI, has the lowest LE levels. Because these sites are young, transpiration from vegetation plays a more important role in total evapotranspiration than soil evaporation. At F77, high levels of volumetric soil moisture indicate that soil evaporation may play a larger role in overall evapotranspiration than at the other sites.

2.4.5 Soil Heat Flux Density

Soil heat flux density (G) at F98 was smallest during the winter of 2001, 2002, and 2003 compared to the other sites (Figure 2.6). F98 also had the greatest G during the growing seasons of the same years. In 2004 and 2005, the summer G values are very similar at all sites likely due to the vegetation similarities that would be present by this stage, even at F98. Regression between F98 and F89 showed a slope of 0.95 ($r^2 = 0.85$) and 1.09 ($r^2 = 0.86$) in 2004 and 2005 respectively with F98 being higher than F89 in 2004. Slopes for the comparison of F98 and F77 had slopes ranging from 0.9 ($r^2 = 0.89$) with F98 being higher than F77 to 1.4 ($r^2 = 0.89$) with F77 being higher than F98.

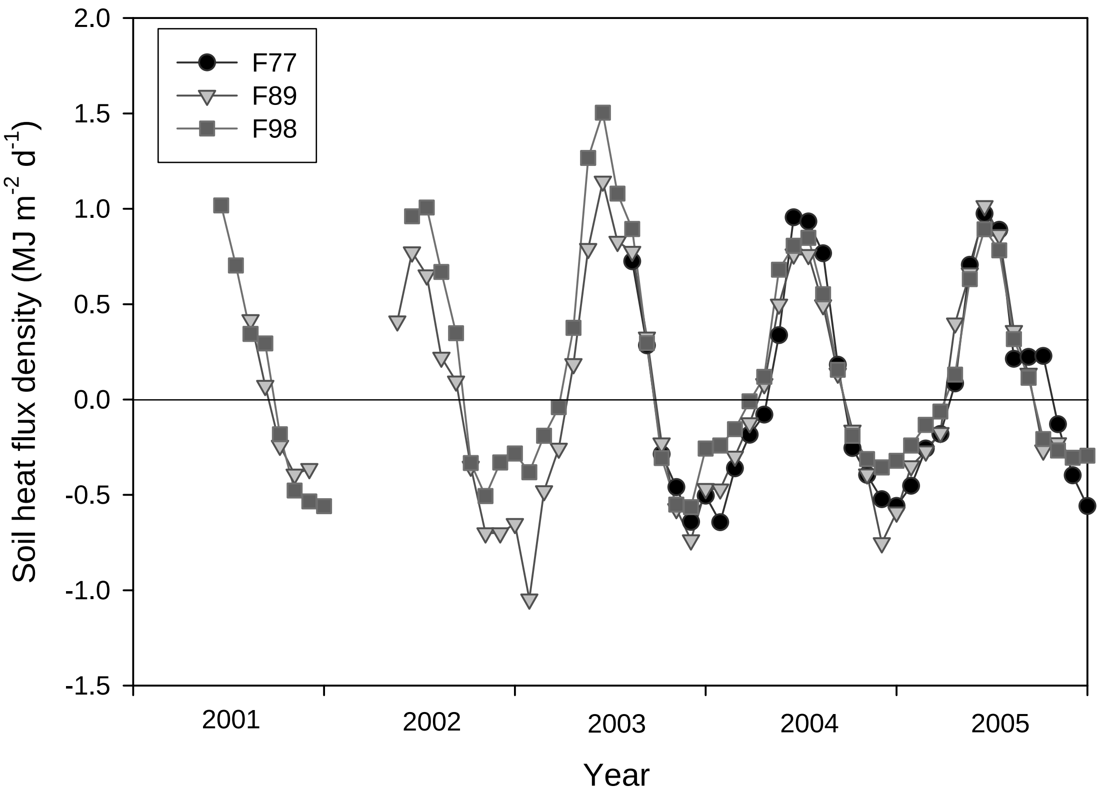

Figure 2.6 Soil heat flux density at three post-fire boreal forest sites burned in 1977 (F77), 1989 (F89), and 1998 (F98) located in central Saskatchewan for 2001 to 2005. Points represent 4-week averages of daily totals.

Soil temperatures (Figure 2.7) at a depth of 0.02 m at all sites were highest in the summer of 2003 with temperatures ranging between approximately 14.5°C and 17°C. Maximum soil temperatures dropped slightly during the summers of 2004 and 2005. During the winters, the lowest soil temperatures were observed in 2003 and the highest winter temperatures were observed between 2004 and 2005.

Soil temperatures are slightly higher at F98 than the other sites during the summer for all years. During the winter, F89 had the lowest soil temperatures but only by a very small margin.

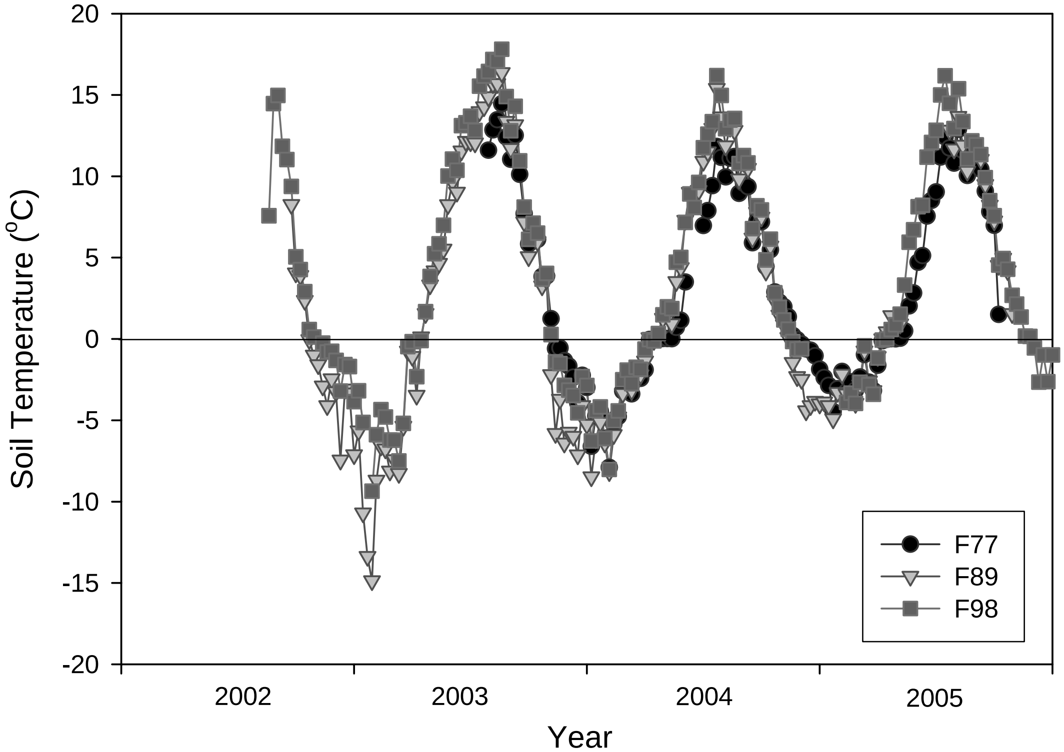

Figure 2.7 Soil temperatures at 0.02 m depth at three post-fire boreal forest sites burned in 1977 (F77), 1989 (F89), and 1998 (F98) located in central Saskatchewan for 2002 to 2005. Points represent weekly averages calculated from daily totals.

Annual trends in G can be attributed to differences in available water. During the drought of 2003, soil heat flux is expected to increase as LE became lower. More net radiation would be put into the soil heating because less energy is being put into evaporating water. During the summer of 2003, 13% of R_n was soil heat flux at F89. The dry conditions could also likely increase the soil temperatures which would also lead to higher soil heat fluxes. In 2004 and 2005 when conditions were wetter, G levels returned to levels similar to those prior to the drought. However, this becomes more complicated when soil conductivity is considered. Wet soils will have a greater conductivity than dry soils which could lead to larger G at times.

The higher G observed at F98 can be attributed to the amount of bare soil that is present at F98. Bare soil will be darker colored and will thus have a lower albedo than soil covered or shaded by vegetation. The low albedo will allow the soil surface to absorb more energy from incoming solar radiation. During the winter, snow is likely to accumulate more on the soil surface in areas with less vegetation. The snow will provide an insulating layer above the soil and will prevent the loss of energy through the surface. Deeper snow accumulation will also impact the soil heat flux by supplying a thicker insulating layer above the soil surface to prevent energy losses. Less negative soil heat fluxes in winter at F98 may be explained by the increase in albedo as a result of the snow cover. Because more bare snow is exposed at F98 due to this site's lower LAI, it will likely reflect more solar radiation. The older sites will have similar G because of their similar vegetation characteristics and therefore similar levels of snow interception. Although F77 had more snow accumulation according to the snow surveys and sonic ranger measurements, this was not reflected in the soil heat flux measurements. We would expect that a thicker layer of snow above the soil surface would provide more insulation, however this may not have been the case at this site in particular.

Soil heat flux is also dependent on the thickness of soil organic layer present on the soil surface. The severity of the forest fire will determine how much soil organic layer remains following the fire (Amiro, Orchansky et al. 2006). Age of the forest stand will also determine the thickness of the litter layer (Amiro, Orchansky et al. 2006). A thick soil organic layer and litter layer will provide an insulating layer above the soil that will decrease the amount of energy being lost from the soil surface (Baldocchi et al. 2000) and will likely result in decreases in G as the forest ages (Amiro, Orchansky et al.

2006). The soil organic layer and litter layers are thicker at F77 and F89 than at F98 because they are more mature and further decomposition of the litter has occurred. The soil organic layer and litter layer will also decrease the amount of sunlight reaching the soil surface and will therefore increase albedo (Baldocchi et al. 2000). Increased pore space will increase diffusion of heat and water through the litter layer as well which will impact the thermal characteristics of the soil surface below (Baldocchi et al. 2000).

2.4.6 Bowen Ratio

The Bowen ratio (ratio of H/LE based on daily totals) indicates that, for much of the monitoring period at all sites, most energy was partitioned into sensible heat (Figure 2.8). This is shown when the absolute value of the Bowen ratios is greater than one. Generally, Bowen ratios are high in spring before leaves have emerged because H is high, but LE is very low due to the lack of leaves present for transpiration. Bowen ratios are highest in the springs of 2004 and 2005 when LE was the highest at all three sites. The spring of 2003 had very high Bowen ratios showing the impacts of the drought experienced that year. An area of interest is in the winter at the beginning of 2004 when the Bowen ratio at F89 and F77 have low negative values whereas F98 has high positive values.

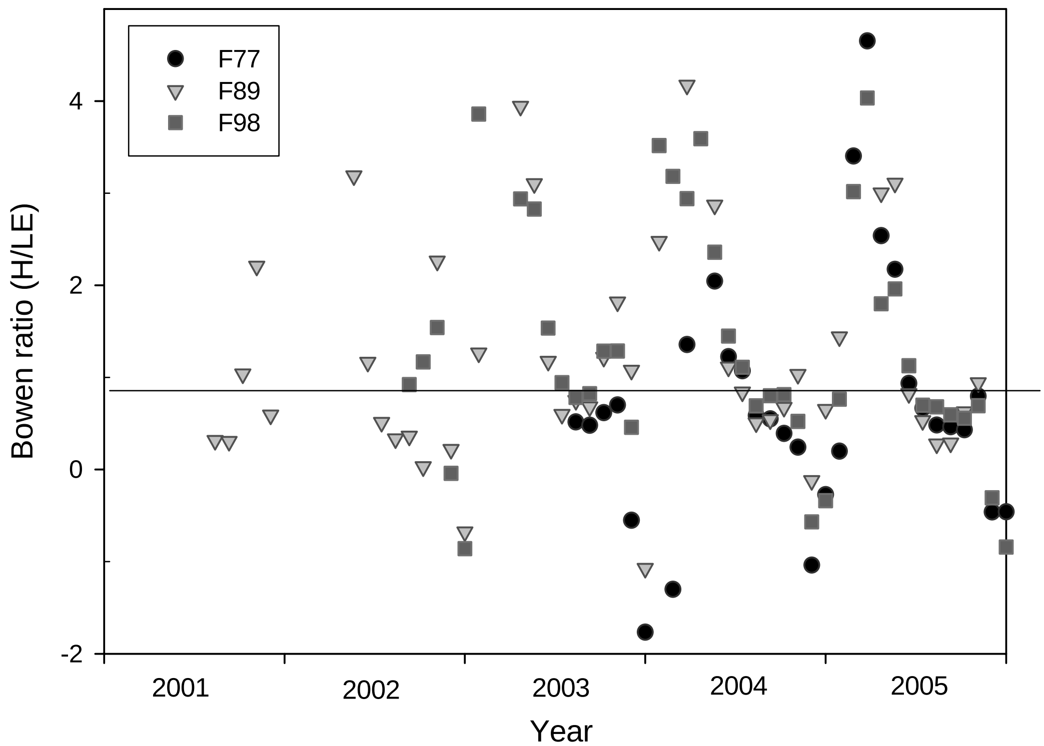

Figure 2.8 Bowen ratios (H/LE) at three post-fire boreal forest sites burned in 1977 (F77), 1989 (F89), and 1998 (F98) located in central Saskatchewan for 2001 to 2005. Points represent 4-week averages of daily totals. The horizontal line marks H/LE = 1. Points greater than 5 or less than -2 were excluded.

Bowen ratios were greatest (both positive and negative) during 2004 and 2005 because LE values approach zero but do not become negative, where H values can be less than zero causing a negative Bowen ratio. This is somewhat unexpected because during these years, conditions were wetter than average so we would expect that more energy would be going towards evaporating water than to warming the air. The dry conditions would increase stress on the vegetation, causing them to conserve water and therefore, evapotranspiration would decrease. This would cause the ratio between H and LE to increase. We would predict that Bowen ratios at a young forested site would be much higher than in a mature conifer forest due to the low levels of LE and the higher levels of

H due to the small and sparse canopy. Jarvis et al. (1976) found that the mature conifer forests they studied had Bowen ratios ranging from 1.3 to 1.9 which was higher than expected. Bowen ratios found in this study show that H was much greater than LE in the spring of all years. Arain et al. (2003) also found that H flux densities were much greater than LE during spring and early growing season. Amiro, Orchansky et al. (2006) also observed high Bowen ratios at young sites ranging from 1.5 to 3. They also observed high evapotranspiration values at F89 which was responsible for the low Bowen ratios at that site.

Immediately following a fire in a conifer forest in Alaska, fire increased H and decreased LE. This led to a high Bowen ratio of approximately 3.5 (Chambers & Chapin 2003). A four-year-old post-fire site in the same region had a lower H than unburned stands, and the ratio of LE/R_n that was higher than unburned stands. This resulted in a lower Bowen ratio of 1.3 (Chambers & Chapin 2003). This suggests that within four years following fire, drastic changes can take place in the energy budget of a boreal forest. This agrees with our results which show that Bowen ratios are often lower at the more mature sites and that time following fire leads to an increase in LE and thus a decrease in Bowen ratio.

2.4.7 Energy Budget Closure to Calculate LE

Energy budget closure was calculated at all sites through linear regression between H+LE and R_n-G as daily totals. Closure was relatively good for all sites (Table 2.4). Energy budget closure at F77 ranged from 0.95 (r^2 = 0.81) in 2004 to 0.98 (r^2 = 0.51) in 2005. At F89, mean energy budget closure was 0.96 with a range of 0.94 (r^2 = 0.75) to 0.99 (r^2 = 0.77) for the years 2003 to 2005. Energy budget closure at F98 ranged between 0.71 (r^2 = 0.51) and 0.84 (r^2 = 0.63). Overall, the closure values observed in this study agree well with closure values reported by other researchers. Liu et al. (2005) reported closure with slopes ranging from 0.77 to 0.83 for a 15-year old site in Alaska. Although there are several reasons that lack of energy budget closure could occur, it is still unclear what the actual source of poor closure is (Wilson et al. 2002).

Table 2.4 Energy budget closure regression line values of H + LE vs. R_n-G of three post-fire boreal forest sites in central Saskatchewan burned in 1977 (F77), 1989 (F89), and 1998 (F98).

Year	Site	Slope	Intercept	r^2
2003	F77	0.96	0.6	0.93
	F89	0.99	0.31	0.77
	F98	0.84	1.08	0.63
2004	F77	0.95	0.19	0.81
	F89	0.96	0.77	0.87
	F98	0.71	0.98	0.51
2005	F77	0.98	0.31	0.84
	F89	0.94	0.6	0.75
	F98	0.79	1.07	0.58

We wanted to explore the possibility of using the energy budget to calculate LE as a substitute for measuring LE. To do this, measured LE values were plotted against LE values calculated from the energy budget equation (LE = R_n-G-H) using measurements taken from April 15 to October 15. Regressions were forced through zero to remove any residuals. In Table 2.5, sites showed regression lines with slopes ranging from 0.89 to 1.07. R-square values ranged from 0.22 to 0.85. Very low fluxes at F98 resulted in proportionally more scatter which caused poor regression relationships between calculated and measured LE with r^2 values less than 0.21 with a substantial offset. The slope observed at F77 for 2003 may be an artifact due to the small data set available for that year when monitoring only commenced in the summer. When we assumed that energy budget closure was 0.9, calculated values of LE would be very close to the measured LE observed in the field.

Table 2.5 Slopes of regression of comparison of measured LE vs. calculated LE from the energy balance for two sites in central Saskatchewan for the years 2003 to 2005. Numbers in brackets are r^2 values. Regressions were forced through zero.

	Site	
Year	**F77**	**F89**
2003	0.95 (0.85)	0.89 (0.45)
2004	0.96 (0.22)	0.89 (0.64)
2005	0.92 (0.48)	0.92 (0.48)

2.4.8 Ecological Implications

Changes in the energy budget and the feedbacks of the energy balance to the atmosphere will have important impacts on the forest ecosystem. Changes in H and LE following fire could have major implications on how the forest will recover from fire and how it will continue to develop over time.

Increases in H that were observed could result in an overall increase in air temperatures over the forest. An increase in air temperature could lead to an increase in the speed of recovery by vegetation following fire. Air temperature increases could also impact other components of the energy budget including evapotranspiration and soil heat flux. Higher temperatures may also lead to an increase in drought conditions which could potentially lead to an increase the area burned by forest fires (Chapin et al. 2000).

Drought has a significant impact on the energy budget of boreal forests. Drought caused increases in H, decreases in LE, lower soil moisture levels, and increases in G. These impacts will in turn have implications on the ecosystem. If the frequency of drought conditions and therefore forest fire increases with climate change, the boreal forest will likely experience an increased abundance of young deciduous species common in early successional stands. The increase in deciduous forests could then lead to increased transpiration and decreased H (Chapin et al. 2000). An decrease in H would have a significantly negative feedback on climate warming (Chapin et al. 2000). Drought caused decreases in LE and increases in H at the sites studied here. Future droughts will likely cause similar effects. It is predicted that drought effects will increase with climate change which could lead to a more permanent change in the magnitude of the energy fluxes.

LE also increases over time as the forest ages because transpiration increases. The water budget will change as vegetation becomes taller and more mature, precipitation interception will increase and less moisture will reach the forest floor. This will lead to a decrease in soil moisture levels. Increases in the density and size of vegetation will also lead to less snow accumulation on the forest floor.

There were no apparent differences in G and R_n by 2004 among sites. This may change as each site matures but how each flux will change remains uncertain.

After approximately five years following fire, all of the energy fluxes at F98 began to more closely resemble the fluxes at F77 and F89 but remained slightly lower than the other sites. Further observation is required at F98 to investigate when the fluxes will be similar to the other sites and whether they will continue to increase to a certain level and then plateau.

2.5 Conclusions

Research here demonstrates that there are slight differences in the energy budget among sites of varying age following fire. By using 4-week averages in this analysis, we illustrated longer–scale temporal patterns and differences. However, using this method may also result in missing some important events that may help to describe the differences among sites.

The energy budgets shown here illustrate that only slight differences are present in the energy budget by five years following fire. The youngest site came to resemble the 27-year site by 2004 and 2005 in H and G. All fluxes at F77 and F89 were almost the same by 2004 and 2005. The energy budget will likely have an impact on how the carbon budget of these sites compares as well. Similar observations were made by Chambers & Chapin (2003) who observed that young forests came to resemble mature forests by five years following fire.

Fire removes the overlying vegetation which will decrease albedo immediately following the fire by exposing more bare soil and charred black surfaces burned by the fire. More shortwave radiation will reach the surface, and thus will increase surface temperature. Increases in surface temperature will lead to an increase in the longwave radiation emitted by the surface and will hence lead to lower levels of R_n. Increased surface temperature will also increase the sensible heat flux following fire. The lack of vegetation following fire will result in much lower levels of transpiration and the LE is expected to be low. Chambers and Chapin (2003) give evidence that fire will increase albedo and decrease H. These decreases in H suggest that changes in stand age within the boreal forest can have comparable impacts on the atmosphere (Eugster et al. 2000).

Climate change will likely increase the rates of nitrogen and carbon cycling as well as increase temperatures. These impacts may result in a significant increase in biomass production (Van Kooten & Arthur 1989). This would mean that in the future, forests may recover much faster following fires and their energy budgets could come to resemble those of mature forests sooner at a younger age. Increased sensible heat flux will likely have a significant impact on regional climates leading to a warming of the region.

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3. CARBON BUDGETS AT THREE YOUNG BOREAL FOREST SITES IN CENTRAL SASKATCHEWAN FOLLOWING FIRE

3.1 Abstract

Climate change is an important problem facing the world today. The increase of greenhouse gases in the atmosphere has caused a warming of the Earth's atmosphere. It is expected that the frequency of forest fires will increase with climate change. Boreal forests are one of the most important carbon sinks. They are responsible for storing large amounts of carbon within their vegetation and soil. Following disturbance such as fire, boreal forests can become carbon sources. Here, three young post-fire boreal forest sites in central Saskatchewan were measured using the eddy covariance technique to observe carbon fluxes of forests following fire. Sites were burned in 1977 (F77 site), 1989 (F89 site), and 1998 (F98 site). Eddy covariance towers were constructed at each site and measurements were taken for a minimum of three consecutive years for the entire year. Net ecosystem exchange (NEE) was measured as well as other supporting meteorological measurements and energy fluxes. Gross ecosystem production (GEP), ecosystem respiration (R), and net ecosystem production (NEP) were calculated from R and NEE measurements. Carbon stock measurements were taken in 2001 and 2005 to measure the changes that occured in the different carbon pools with time after fire. NEP at the youngest site recovered to levels similar to those of more mature sites within a period of approximately seven years. Both R and GEP levels became more similar among F77 and F89 by 2005 however, F98 still maintained somewhat lower values. Carbon stock measurements showed that at all sites, vegetation, woody debris, and organic soil carbon were the most dominant carbon pools in young boreal forests. Understanding the impacts

of fire on the carbon budgets of boreal forests is important for future predictions of the possible impacts that climate change may have on local, regional, and global carbon budgets and climates.

3.2 Introduction

Anthropogenic climate change is the result of additional greenhouse gases entering the atmosphere which change the energy balance cycle. Greenhouse gases are emitted from a variety of sources including agriculture, wetlands, and human activities. Carbon dioxide (CO_2) is the most common greenhouse gas of concern for cycling in the boreal forest. As climate change has become an issue, it has become more important to study how carbon dioxide enters the atmosphere and how carbon can be sequestered instead of being released.

Fire has a large impact on the carbon cycle in the boreal forest because it is one of the main stand renewing agents (Amiro 2001). Boreal forests cover a large portion of the earth's surface. Mature boreal forests are known carbon sinks (Dixon et al. 1994) and can store carbon in a variety of different forms and in several major pools including vegetation, coarse woody debris, organic soil horizons and mineral soil (Pregitzer & Euskirchen 2004). However, following fire, forests become carbon sources (Kurz & Apps 1999). In addition to releasing large quantities of CO_2 and other greenhouse gases into the atmosphere during the fire, the vegetation changes that take place following a fire also result in the release of CO_2 by changing dynamics of processes including photosynthesis, respiration, and decomposition.

It is predicted that climate change will result in increases in surface and air temperatures in the boreal region. Increases in surface temperatures will likely result in an increase in biomass production (Van Kooten & Arthur 1989) and could potentially alter the carbon budget by increasing the amount of carbon that is present in vegetation. These forests' role in carbon sequestration will become more important as we try to reduce the impacts of climate change. If the incidence of forest fire increases, we will see an increase in carbon sources which could potentially amplify the effects of climate change.

Carbon dioxide levels are often measured in terms of net ecosystem productivity (NEP). NEP is the net exchange or difference of gross ecosystem productivity (GEP) and ecosystem respiration (R) (e.g., Dunn et al. 2006). Gross ecosystem productivity includes any process that results in carbon dioxide being stored or absorbed, including photosynthesis. Ecosystem respiration used to calculate NEP includes both heterotrophic and autotrophic respiration at all levels of the forest stand including soil and canopy respiration.

Air and soil temperature affect CO_2 exchange. Air temperature is an important factor in determining overall ecosystem carbon exchange because it impacts photosynthesis and respiration. Several studies have illustrated that decreasing temperature is a major factor contributing to the decrease in GEP and R during the fall (Bergeron et al. 2007; Goulden et al. 1997; Vogg et al. 1998).

Respiration is an important contributor to the net ecosystem carbon exchange. Soil respiration is highly influenced by soil temperature (Gaumont-Guay et al. 2006). This includes autotrophic and heterotrophic respiration. Bergeron et al. (2007) observed increases in GEP and R when there were increases in annual air temperature.

Leaf area index (LAI) is a site characteristic that impacts the carbon exchange of a boreal forest. Both GEP and R increase when leaves on deciduous tree species emerge in the spring and become photosynthetically active. Deciduous species will also behave differently in terms of photosynthesis than coniferous trees. Deciduous forests that have low LAI can have similar gas exchange rates to mature coniferous forests that have higher LAI due to their different rates of photosynthesis (conifers photosynthesize more slowly) (Goulden et al. 2006). Fire greatly decreases LAI in forests. As forests recover from fire, LAI increases and species composition changes from deciduous shrubs and grasses, to larger deciduous trees and coniferous trees. The changes in LAI that occur as forests progress through different stages of succession will have significant impacts on the carbon budget.

Albedo and radiation impact carbon exchange by determining the temperature of many surfaces in a forest site. In Chapter 2 of this study, it was demonstrated that albedo among three post-fire boreal forests of varying ages was different. These sites had different vegetation cover and different amounts of bare ground and charred surfaces which impacted the albedo. Darker surfaces will reflect less radiation and will warm to higher temperatures than lighter surfaces. Higher temperatures will then impact respiration and photosynthesis rates.

Moisture levels also play a role in carbon exchange. Several studies have investigated the impacts of drought on carbon exchange in a variety of forest types. Soil moisture strongly impacts respiration rates at the soil level. The sites studied here showed a large increase in volumetric soil water content during 2003 and 2004 (Chapter 2). Moisture levels will also influence respiration and photosynthesis above ground.

Periods of drought which was observed in 2002 and 2003 at the study sites will likely have a large impact on the carbon budget.

Several studies in recent years have measured carbon exchange in boreal forests during different stages of succession following disturbance (Amiro, Barr et al. 2006; Bergeron et al. 2007; Bond-Lamberty et al. 2004; Goulden et al. 2006; Litvak et al. 2003; Pregitzer & Euskirchen 2004; Randerson et al. 2006). However, few have studied post-fire sites for more than two years (Amiro, Barr et al. 2006; Goulden et al. 2006) or for periods outside of the growing season. A similar study was conducted by Litvak et al. (2003) where they measured the carbon exchange along a chronosequence of five post-fire sites in northern Manitoba. They found that most biomass accumulation takes place in sites ranging from 20 to 70 years following fire with the highest accumulation taking place around 30 years following fire. However, this study was only conducted for two months during the growing season and annual carbon fluxes were not available. Amiro, Barr et al. (2006) conducted a study where the energy and carbon budgets of two post-fire sites (the same as are studied here) were compared to several other sites including mature stands and harvest stands. This study yielded results that suggest that carbon budgets of disturbed stands are net carbon sources for a period of at least seven years. This study only used two years (2001, 2002) of data for their observations.

Although there has been work done studying fire effects on carbon budgets of recovering forests, there is still some uncertainty in the impacts that fire will have on recovering forests over time. Models have shown that there are net carbon losses in boreal forests for many years following fire (Kurz and Apps 1999). It is our objective to obtain data to help validate these models. The present study focuses on the changes in the components of the carbon cycle with time following fire in three boreal forest sites located in central Saskatchewan. Carbon fluxes were measured at three sites of varying ages near Prince Albert, Saskatchewan using the eddy covariance technique. A chronosequence of forest ages was created by studying forests that were burned in 1977 (F77), 1989 (F89), and 1998 (F98). The sites were in close proximity to one another so they experience similar climate and weather conditions. Sites F89 and F98 were monitored from the spring of 2001 to the winter of 2005. Measurements at F77 were taken from summer 2003 to winter 2005. Carbon stock measurements were taken at the sites in 2001 and 2005 to make a comparison of how carbon storage within the different pools changed over time. The impacts of available moisture will also be examined.

3.3 Methods and Materials

3.3.1 Site Description

Details and site characteristics of the three sites used in this study are described in Section 2.2.1.1.

3.3.2 Carbon Stock Measurements

Forest carbon stocks were measured for above-ground and below-ground components. Above-ground carbon pools include trees, shrubs, ground vegetation, coarse woody debris, and litter. Below-ground components include soil organic carbon and roots. Measurements of these carbon stocks were taken in 2001 at F89 and F98, and again in 2005 at all three sites. A comparison of these stocks will give an indication of how the distribution of carbon in forest ecosystems can change over time following fire.

3.3.2.1 Tree characterization measurements

Plots with dimensions 10 m x 10 m were constructed at each site. These plots were used for tree measurements. One plot was measured at F77, three plots were measured at F89, and four plots were measured at F98. The same plots used in 2001 at F89 and F98 were measured in 2005. F77 was not measured in 2001.

All trees in the plot with diameter at breast height (DBH) greater than 5 cm were measured for height and diameter. Heights were measured using clinometers. Diameters were measured at breast height (approximately 1.5 m) using calipers. Species of living and dead trees were recorded. Heights of dead trees were not measured. The amount of carbon present in above-ground biomass was calculated using equations from Gower et al. (1997).

3.3.2.2 Shrub characterization measurements

Within each large tree plot, at least one 3 m x 3 m plot was constructed in the corner of the larger plot for shrub measurements. Any woody vegetation that was greater than 15 cm in height and had a diameter of less than 5 cm was considered a shrub. Shrub heights were measured using meter sticks. For carbon biomass calibration purposes (biomass vs. height), selected shrub species of typical heights were cut and placed in separate bags for later analysis. At F77, four shrub plots (one large plot) were measured. A total of 12 shrub plots (three large plots) were measured at F89 and four shrub plots were measured at F98. For biomass calibration, shrubs were dried at 40°C for a minimum of 48 hrs. Samples were then weighed and the number of stems per sample was recorded to obtain a mass per stem calibration with height.

3.3.2.3 Clip plots (biomass) characterization measurements

To obtain samples of understory vegetation, 1 m x 1 m plots were constructed in the corners of the large tree plots (four clip plots for each large plot). All woody vegetation with heights less than 15 cm and all herbaceous vegetation was clipped and placed in bags. Samples were then oven-dried at a temperature of 80 °C for at least 24 hours and weighed to determine the oven dry mass.

3.3.2.4 Litter and organic soil layer measurements

Litter and soil organic layer samples were taken from a 10 cm x10 cm plot located within the clip plots. Litter was scraped off the top layer of soil and placed in a plastic ziplock bag. The organic layer was then cut off the top level of mineral soil and placed in a separate plastic bag. These samples were then oven-dried at a temperature of 80 °C for 48 hrs and weighed to obtain the oven-dry mass. These 10 cm x 10 cm samples were used to calculate the organic soil carbon for F89 only in both 2001 and 2005. To calculate the amount of carbon per m^2 using organic samples, a carbon content of 18.3% was used at all sites.

3.3.2.5 Woody debris measurements

A fuel triangle (10 m x 10 m x 10 m transect) was constructed within the large tree plots using one side of the tree plot as one side of the triangle. A string was used to outline the triangle and was marked at 5 m intervals. Along the 5 m intervals, diameters of the debris crossing the string were measured. Every five meters the smallest diameter class was dropped. Diameter categories were as follows: 5 mm to 10 mm, 10 mm to 20 mm, 20 mm to 30 mm, 30 mm to 50 mm, 50 mm to 70 mm, and greater than 70 mm.

Trees that were still rooted, dead branches attached to living trees, stumps, and exposed roots of standing trees were not measured. Diameters were measured using small calipers.

3.3.2.6 Soil Samples

Soil cores were taken using a tulip planter that had a diameter of 5 cm and a depth of 15 cm. Cores were then cut into slices 5 cm thick and placed in separate bags. Two cores were taken to a total depth of 30 cm when possible. A total of four samples were taken in each large tree plot. Samples were then divided into two parts based on weight. Half of the sample was used for root analysis, and half was used for combustible carbon content.

Soil samples of the organic soil layer were used to calculate the amount of carbon at F77 and F98. Carbon content was assumed to be 18.3% carbon per m^2. The carbon content was calculated by taking the average carbon contents of samples analyzed from at F89 and F98 in 2001.

Soil samples used for combustible carbon content in mineral soil were oven-dried at 70 °C for 48 hrs and 5 g of dried soil was placed in a ceramic crucible. Crucibles were then placed in a muffle furnace at 375 °C for approximately 16 hrs. Samples were then removed and placed in a desiccator until the temperature had fallen to approximately 150 °C. Samples were then weighed. Carbon was calculated by subtracting the final sample weight from the initial oven-dried soil weight. This gives the amount of carbon that was combusted in the muffle furnace.

3.3.2.7 Root Samples

Soil samples were dried at 70 °C for 48 hrs. Dry samples were then weighed. Dry samples were then soaked in water for approximately one hour in order to "re-hydrate" the fine roots, making them less brittle. The samples were then run through a root washer device to remove the majority of soil and large debris from the roots. The samples were then filtered though a 40 micron 55 mm filter paper in a Buchner funnel attached to a suction vacuum to remove the water. The filter paper containing the roots was then removed and roots were separated from the paper. In instances where roots were not adequately filtered, root samples were picked through by hand. The roots were then dried in an oven at 70°C for 48 hrs and then weighed to four decimal places. Carbon content of roots is estimated to be 50% of the dry weight of the roots. This equation was applied to the dry root samples to obtain the mass of carbon contained in the roots per unit area. Coarse root biomass was estimated using tree measurements and the equation used by Li et al (2003). This equation was also used to estimate the carbon present in the coarse roots.

3.3.3 Leaf Area Index (LAI) Measurements

Leaf area index was measured to obtain a description of the canopy at each site. Clip plots were done at F98 to characterize LAI of understory vegetation. The TRAC method was used at F77 and F98 but was not used at F89.

3.3.3.1 LAI clip plots

Samples of jack pine, black spruce, and trembling aspen were used for LAI clip plots. Clip plots of understory species including fireweed, wild rose, and other species.

Leaves or needles were stripped off the stems and were dried and weighed. Samples of needles and leaves from each species were traced on paper to obtain an estimate of surface area. At F98 shrubs of typical size were cut and dried. Needles/leaves were then removed and weighed. Weight and surface area were then used to calculate a leaf area to weight calibration.

3.3.3.2 Optical methods

Tracing radiation and architecture of canopies (TRAC) (Third Wave Engineering, Ottawa, Canada) is a standard method used to measure LAI of tall canopies. TRAC measurements were done by constructing transects 100 m long with markers placed at 10 m increments. The TRAC was held at a constant height and the transect was walked at a consistent speed of one meter per three seconds (Leblanc et al. 2002). This was done on clear, sunny days at approximately 10:00 am. Further details on TRAC methods are described in Chen et al. (2006).

3.3.4 Meteorological Measurements

Towers were constructed at each site. Tower height at F77 and F89 were 7.4 m and 5.2 m respectively. From 2001 until August 16, 2002, a triangular tower with a height of 7.7 m was used at F98. On August 17, 2002, the triangular tower was replaced with a 20 m tall scaffold tower. Most supporting meteorological measurements and flux measurements were taken at the top of the towers at all sites. The supporting meteorological methods are described in detail in Section 2.2.2.

3.3.5 Carbon Flux Measurements

CSAT3 (Campbell Scientific Inc.) sonic anemometers were used to measure air temperature and three-dimensional wind velocities at all three sites. Carbon dioxide measurements were made using an open-path LI7500 (LICOR Inc, Lincoln, NE) infrared gas analyzer (IRGA) at all sites. The LI7500 was calibrated using a known concentration of CO_2 and a zero point several times a year. The calibration showed little drift in the measurements so more frequent calibration was not required. The CSAT3 and gas analyzer signals were sampled at a rate of 10 samples per second, and the cross-products were stored every 30 minutes. Covariances were calculated from these cross-products. The wind velocities were rotated so the mean vertical velocity was zero over the 30-minute period (Tanner & Thurtell 1969). This rotation was normally minor because of the flat topography.

At F98 prior to August 16, 2002, a closed-path LI6262 IRGA (LICOR Inc.) located on a 30 cm triangular tower at a height of 7.7 m was used to measure CO_2 and latent heat flux (LE). The closed-path IRGA included 3 m of 8 mm Bevaline IV tube located in the axis of the sonic anemometer. Air was pulled through the tubing at a rate of 5 L min^{-1} by a pump and pushed through a 1 μm particle filter into the closed path analyzer. At least once a month, the gas analyzer was calibrated. On August 16, 2002, a scaffold tower was erected at F98 and the gas analyzers were relocated at a height of 20 m. Corrections to the data were made to account for temperature and pressure changes, any lag that resulted from the tubing, and for frequency response (Amiro, Barr et al. 2006). On August 18, 2002, the closed-path LI6262 IRGA was replaced with an open-path LI7500 IRGA at a height of 20 m. NEP (net ecosystem production) was calculated by adding the measured eddy flux to the estimated storage term that was used to account for storage within the canopy. NEP was calculated as the negative of net ecosystem exchange (-NEE).

Flux measurements of net radiation, sensible and latent heat fluxes, and soil heat flux were also made. These measurements are described in Sections 2.2.3.1, 2.2.3.2, and 2.2.3.3, respectively.

3.3.6 Data Analysis and Quality Control

3.3.6.1 Quality control

Data were filtered to reject flux data obtained during periods such as rain and snowfall events when instruments would malfunction. Friction velocity (u_*) thresholds were used as a criterion to determine when sufficient turbulence was present for accurate flux measurement. During times of low turbulent mixing, the eddy covariance technique underestimates fluxes (Wohlfahrt et al. 2005). The u_* thresholds were calculated by finding nighttime measurements (when photosynthetically active radiation was below a threshold value of zero) and removing outlying data points. The remaining data were then bin-averaged according to a u_* in 10 bins with an equal number of points. A graph of NEE vs. u_* was constructed using these data. The area when the relationship begins to plateau is the u_* threshold, defined as 80% of the average NEE of the last three bins. The threshold was then rounded up to the nearest 0.05 m s^{-1} value (Amiro, Barr et al. 2006). When u_* was below this threshold, carbon fluxes were rejected because of inadequate turbulence. When determined by site and by individual years, u_* thresholds ranging from 0.2 m s^{-1} to 0.3 m s^{-1} were observed. To maintain consistency for analysis among sites and throughout the entire study period, thresholds for u_* were set to 0.25 m s^{-1} at all sites and all years. NEE values that were less than -30 μmol m^{-2} s^{-1} or greater than 30 μmol m^{-2} s^{-1} were also rejected from analysis to remove outliers.

3.3.4.2 Gap filling

During the growing season, gap filling methods used by Amiro, Barr et al. (2006) were used to fill gaps resulting from low u_* conditions, instrument malfunction, or rain. Small gaps of less than 4 half-hour periods in NEP were filled using linear interpolation.

Respiration (R) was estimated as –NEP when GEP (gross ecosystem productivity) was known to be zero (during night and days when air temperatures were below 0°C). To fill large gaps in R, a relationship between R and soil temperature (Ts) was developed. An additional parameter was then added to the empirical relationship between R and Ts. The new parameter was estimated using a linear regression model and was permitted to change with time. The model with the new parameter was then used to estimate R during the day and was used at night to fill gaps. GEP was calculated as NEP + R during the day. At night, GEP was set to zero. An empirical model using the relationship between GEP and PAR with an additional parameter (similar to the R and Ts relationship model) was used to fill gaps in GEP. Gaps in NEP were filled using modeled GEP – R. Further explanation and equations used for gap filling of carbon fluxes are given in Appendix I.

Using an open path IRGA can result in unreliable measurements during cold periods. Sensors can heat air surrounding the instrument which can alter measurements. Often during the winter, measurements that indicate a carbon uptake are observed, when we expect that only a small release of carbon should be taking place. It is impossible for such events to occur when snow is covering the ground and photosynthesis is not taking place (Amiro, Orchansky et al. 2006). For this reason, winter respiration and NEP data were filled using the procedure outlined in Appendix II.

Gap filling procedures that were used to fill gaps in H and LE are described in detail in Section 2.2.3.3 of the previous chapter.

3.4 Results and Discussion

The main components that make up the net ecosystem exchange are GEP and R. The balance of these two components determines NEP which is used to determine the overall ecosystem exchange of carbon. A maximum of 40% of NEP data was gap-filled at F89 in 2004. The percentage of NEP data gap-filled at F98 ranged from 33% in 2003 to 49% in 2004. The percentage of gap-filled NEP data at F77 was 33% in 2004 and 38% in 2005.

3.4.1 Gross Ecosystem Production

GEP gives a clear illustration of the trends that occur at sites of different ages (Figure 3.1). GEP is the amount of photosynthesis that takes place at a given site. It is the carbon absorbed by the vegetation via photosynthesis.

F89 had the highest peak GEP levels for all years. Maximum GEP at F89 was approximately 9 g C m^{-2} d^{-1} in 2003. GEP at F98 was much lower with maximum levels around 4 g C m^{-2} d^{-1}. In 2004, maximum levels of GEP at F89 were slightly higher than in 2003 and were more than twice that of the maximum GEP at F98. F77 had peak GEP levels between 6 g C m^{-2} d^{-1} and 7 g C m^{-2} d^{-1}. During 2005, GEP at F98 was higher than previous years. Maximum GEP at F98 was between 5 g C m^{-2} d^{-1} and 6 g C m^{-2} d^{-1}. GEP values at F77 and F98 show progressive increases over the entire course of the study period. At F98, increased GEP is likely due to the increase in vegetation at this site as it grows and matures over time.

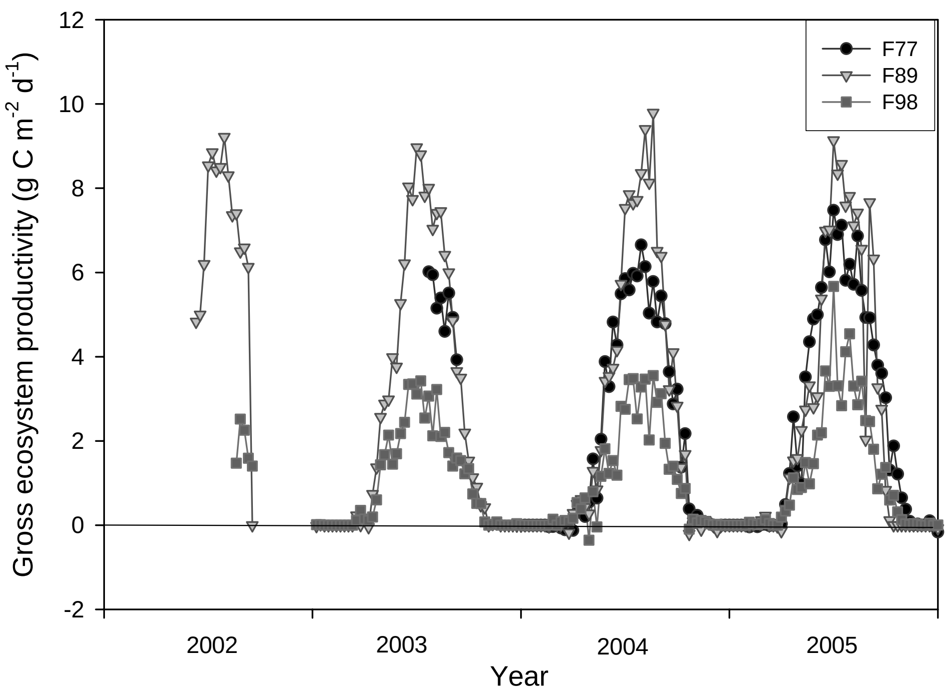

Figure 3.1 Gross ecosystem production of three post-fire boreal forests burned in 1977 (F77), 1989 (F98), and 1998 (F98) in central Saskatchewan from 2002 to 2005. Points represent weekly averages of daily totals.

At all sites, there was an increase in GEP in spring with the maximum GEP being reached in mid-summer. GEP then began to decline as fall approached. The seasonal trends observed at the three post-fire sites here agree with past studies. Several studies have observed that in spring, rapid increases in air temperature without increases in soil temperature led to maximum gains of carbon during that period. The temperature increase caused GEP to increase at a faster rate than R which lead to the gains of carbon (Bergeron et al. 2007; Dunn et al. 2006; Falge et al. 2002). In fall, decreasing radiation causes GEP to decline (Bergeron et al. 2007; Goulden et al. 1997; Vogg et al. 1998).

During this time GEP contributes to annual carbon gains only minimally (Bergeron et al. 2007; Goulden et al. 1997).

Age has an obvious effect on the GEP of a forest. The youngest site consistently had the lowest GEP whereas the two more mature sites had GEP values that were almost double that of the young site. The intermediate site (F89) had higher GEP than the oldest site for all years. This may be due to the slightly larger number of deciduous trees at F89 compared to F77. This may result in higher levels of photosynthesis at F89 than at F77 during the summer.

McCaughey et al. (2006) found that the most important environmental factor controlling photosynthesis was air temperature. An increase in GEP occurred as a result of increased photosynthesis which led to a 74-year old mixedwood forest changing to a strong carbon sink (McCaughey et al. 2006). Total GEP was greater when annual air temperature was higher and when the growing season began early in the year (Bergeron et al. 2007).

3.4.2 Ecosystem Respiration

F89 had the highest maximum R levels in all years (Figure 3.2). The highest R at F89 was achieved in 2004 with a peak of approximately 9 g C m^{-2} d^{-1}. A maximum R of 8 g C m^{-2} d^{-1} was reached at F77 in 2003. Much lower R was observed at F98 with values near 4 g C m^{-2} d^{-1}. In 2004, R at F89 was higher than during 2003. Maximum R at both F77 and F89 was near 8 g C m^{-2} d^{-1} whereas, R at F98 was about half that. In 2005, R levels decreased at F89 but remained similar to previous years at F77. F98 had R levels slightly higher than in 2005 with maximum levels near 4 g C m^{-2} d^{-1}. R at all sites seemed to be more consistent at all sites during the growing season with a longer peak period than in previous years (the maximum levels were longer lived).

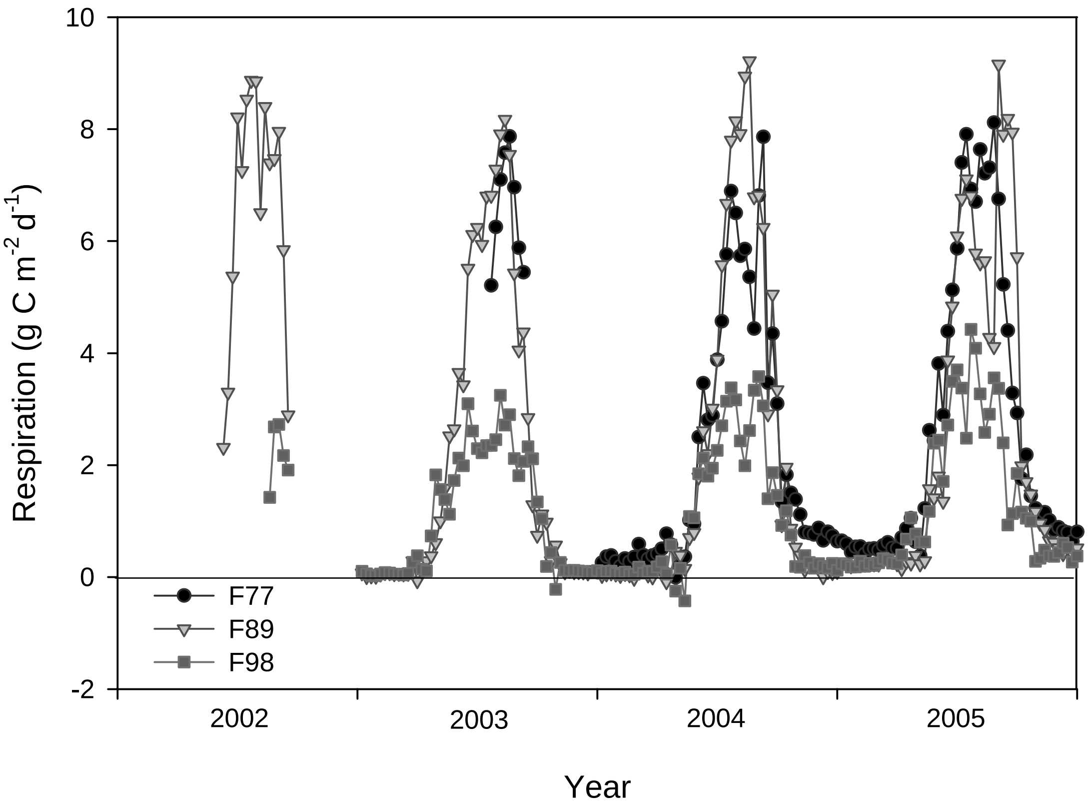

Figure 3.2 Respiration fluxes from three post-fire sites burned in 1977 (F77), 1989 (F89), 1998 (F98) in central Saskatchewan from 2002 to 2005. Points represent weekly averages of daily totals.

Respiration gives a good indication of what is happening at sites in terms of vegetation growth. More mature sites will have higher levels of autotrophic respiration due to the abundance and size of the vegetation. A younger site such as F98 will have smaller vegetation that will respire less than more mature sites.

Temperatures will also have a large impact on respiration rates. For example, Goulden et al. (2006) found that more respiration and more photosynthesis took place during the spring when temperatures were increasing at a site burned in 1989 in northern Manitoba. Smaller respiration rates were observed during the fall when temperatures dropped. During fall, temperature and photoperiod are the main driving forces behind the decreases in GEP and R (Bergeron et al. 2007; Goulden et al. 1997; Vogg et al. 1998).

3.4.3 Net Ecosystem Production

3.4.3.1 Seasonal Trends

Weekly averages of NEP data show that most often, F89 has the highest NEP levels in the summer (Figure 3.3). During the early summer, all sites are carbon sinks. This is indicated by positive NEP values showing carbon uptake by the vegetation. During the later parts of the summer, all sites become carbon sources which is indicated by negative NEP values.

In 2003, F89 was the strongest carbon sink early in the growing season. Maximum NEP values were just over 3 g C m^{-2} d^{-1} at F89 while maximum values at F98 were near 1 g C m^{-2} d^{-1}. In the late summer, F89 was a weak carbon source (or close to neutral) during 2003 and 2004, but it was a strong carbon source in the fall of 2002 and 2005. During late summer NEP at F77 was near -3 g C m^{-2} d^{-1} and was the strongest carbon source among the three sites. This same trend was observed in all years. F98 was a weak carbon source at the end of summer and then became a stronger source later in autumn. F98 was the weakest carbon source and the weakest carbon sink throughout the fall of 2003 with the lowest NEP values reaching -1 g C m^{-2} d^{-1}.

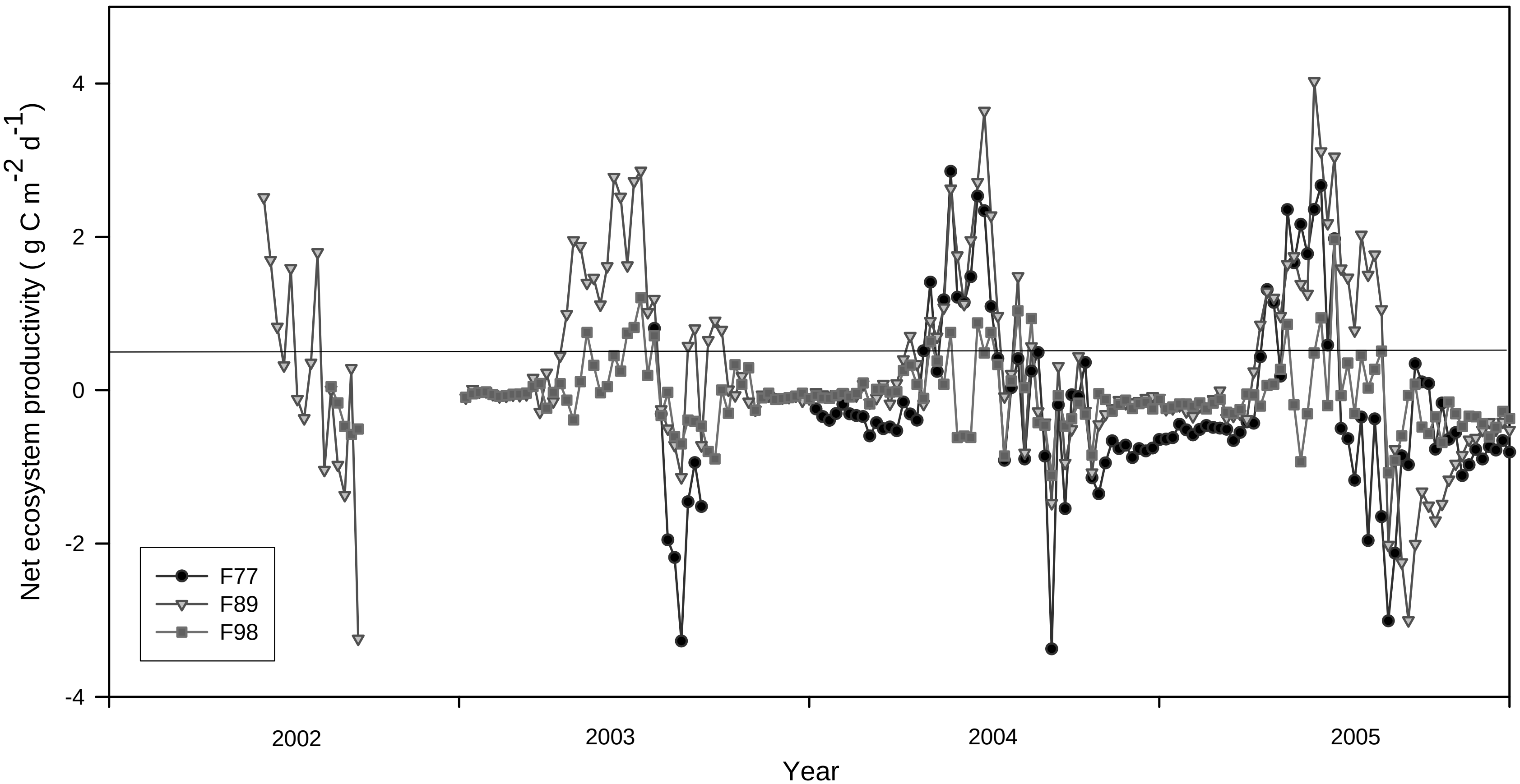

Figure 3.3 Net ecosystem production of three post-fire boreal forests burned in 1977(F77), 1989 (F89), 1998 (F98) in central Saskatchewan from 2002 to 2005. Points represent weekly averages of daily totals. Measurements from April 15 to September 15 inclusive are shown for years with incomplete years of data.

In 2004, the maximum levels of carbon uptake (NEP) observed at F89 were similar to those observed during the spring of 2003. F77 had NEP values that were close to those of F89. Maximum NEP levels were near 3 g C m^{-2} d^{-1} at F77 and F89. F98 was a weak carbon sink during the spring of 2004. NEP levels increased gradually through the spring until the summer when F98 reached its peak NEP at approximately 1 g C m^{-2} d^{-1}. Late in the summer of 2004, F77 and F89 became stronger carbon sources than in 2003. Levels of approximately -3.5 g C m^{-2} d^{-1} were observed at F77 which was the strongest carbon source that year. F98 was a weak carbon source in 2004, however, it released more carbon than in 2003. During the winters, all sites were carbon sources, but F77 was the strongest source.

The most extreme levels of NEP were observed during 2005. Maximum NEP at F98 was near 1 g C m^{-2} d^{-1} similar to previous years. NEP levels at F98 and F77 were similar to previous years. F98 reached its maximum carbon uptake slightly later in the season than the other two sites. During the late summer of 2005, F89 saw a large decrease in NEP and became a stronger carbon sink than the previous years. F89 lost approximately 3.5 g C m^{-2} d^{-1} during this time. F89 showed a gradual increase in source strength during the fall from 2003 to 2005. F77 was also a strong carbon source during the fall of 2005, however the levels of NEP observed at F77 were more similar to those of previous years. F98 was a weaker carbon source in the fall of 2005 than in 2004.

These results agree with other studies that have followed young post-fire boreal sites (Amiro, Barr et al. 2006). The similar values of NEP exhibited by 2005 between F77 and F89 indicate that forest can recover within approximately sixteen years following fire to levels resembling those of more mature forests. Previous studies have shown that carbon exchange in boreal forests can experience exchange rates similar to mature forest within 10 to 30 years after disturbance (Amiro et al. 2003). Goulden et al. (2006) found that midday CO_2 exchange at a burn site began to resemble levels at unburned stands four years following fire. This study did not yield results that were as clear. NEP was still lower at F98 than at F77 and F89 even after seven years following fire. Age of forest stand plays an important role in the strength of a carbon source or sink. A more mature stand will have more abundant vegetation that is well developed. An older forest will also likely have a well developed canopy and understory which would lead to a more diverse and complicated ecosystem. There will be higher levels of respiration and photosynthesis in areas with more mature vegetation compared to areas with smaller, less developed stand.

The species composition of a given site will also have a significant impact on how quickly a forest will recover following fire. For example, a site that has a large abundance of trembling aspen trees may recover more quickly than a site with fewer aspen trees. This is because aspen roots may survive fire and recover more quickly than other deciduous species (Amiro, Barr et al. 2006). However, a site dominated by coniferous species such as jack pine would not recover as fast. Jack pine roots die during the fire and therefore root respiration would cease after the fire. However, the decomposition of these roots would occur and would contribute to respiration (Amiro 2001).

One would expect the NEP of forests containing deciduous species to be highest during the late spring, after the deciduous species have leafed out. As the fall approaches and deciduous leaves senesce, respiration becomes greater than GEP. Goulden et al.

(2006) found that swift recovery in CO_2 uptake in young forests was likely due to the quick growth of deciduous understory species such as fireweed following fire. A large number of deciduous species will also result in a shorter growing season which will lead to a shorter peak time of CO_2 uptake.

3.4.3.2 Annual Totals

Annual totals of GEP, R, and NEP were calculated for years where complete years of data were available (Table 3.1). F89 was a carbon sink for all years. In 2003, F89 was a carbon sink of approximately 177 g C m^{-2} y^{-1}. In 2004 and 2005, F89 was a weaker carbon sink of 113 g C m^{-2} y^{-1} and 88 g C m^{-2} y^{-1}, respectively. Both F77 and F98 were consistent annual carbon sources. F98 was the weakest carbon source in 2003 at -5 g C m^{-2} y^{-1} and was the strongest carbon source in 2001 at -132 g C m^{-2} y^{-1} (Amiro, Barr et al. 2006). It is reasonable that the strength of the carbon emissions would be greatest when the stand was at its youngest age. F98 increased in its source strength in 2004 and 2005 at -17 g C m^{-2} $y^{-1}$and -52 g C m^{-2} y^{-1} respectively. F77 was a carbon source for both 2004 and 2005. It was a strong source in 2005 at -79 g C m^{-2} y^{-1} compared to 2004 where the annual NEP was -40 g C m^{-2} y^{-1}. The change in source strength at F77 may be partially caused by old coarse woody debris that is now decomposing compared to the younger sites where the coarse woody debris may not be decomposing as actively. Greater decomposition would result in higher respiration levels, causing the site to increase in source strength.

Table 3.1 Annual totals (g C m^{-2} y^{-1}) of gross ecosystem production (GEP), ecosystem respiration (R), and net ecosystem production (NEP) at three post-fire boreal forests in central Saskatchewan burned in 1977 (F77), 1989 (F89),and 1998 (F98). Gap filling procedures were used to fill winter data for years with complete data sets (Amiro 2001).

	Site								
		F77			**F89**			**F98**	
Year	**GEP**	**R**	**NEP**	**GEP**	**R**	**NEP**	**GEP**	**R**	**NEP**
2001									-132*
2002						68*			-87*
2003				962	785	177	396	401	-5
2004	746	786	-40	896	782	113	385	402	-17
2005	904	983	-79	936	848	88	455	507	-52

* From Amiro, Barr et al. 2006

Annual GEP levels at F98 were about half that of the other two sites. In 2005, F98 had an annual GEP of 455 g C m^{-2} y^{-1} compared to F77 where the annual GEP was 904 g C m^{-2} y^{-1}. Annual R values showed similar results at F98 compared to F89 and F77. The differences in annual NEP for 2005 between F77 and F89 are due to differences in annual R values. Both sites had almost identical annual GEP values, but R at F77 exceeded R at F89 by approximately 135 g C m^{-2} y^{-1}. The differences in annual NEP between these two sites during 2004 are due to differences in GEP. F89 had annual GEP values that were approximately 150 g C m^{-2} y^{-1} higher than that of F77. This implies that there were likely vegetation differences between these two sites that would cause more photosynthesis at F89 during this year, without causing increases in R.

3.4.5 Evapotranspiration Relationship with NEP

Evapotranspiration has been shown to be correlated with NEP and GEP values (McCaughey et al. 2006). Daily totals of NEP were plotted against evapotranspiration values for the summer of 2004 at all our sites. Regressions had r^2 values of less than 0.06 indicating very weak relationships. We also had poor relationships between evapotranspiration and GEP. This is somewhat surprising because we would expect evapotranspiration to be strongly related to NEP. This is because if stomata are open to transpire water, carbon uptake would occur. During periods of increased levels of evapotranspiration, NEP was positive indicating an uptake of carbon when evapotranspiration was high. At evapotranspiration levels below 1.5 mm d^{-1}, NEP was often negative indicating a carbon loss at times when plant activity for carbon gain was less than respiration losses.

3.4.6 Impacts of Drought

The differences observed from year to year in NEP at these sites may be explained by the drought during 2002 and 2003 at these sites. This may have caused lower carbon uptake than would usually be expected from sites of this age. Very wet conditions were present at all sites in 2005. Amiro, Barr et al. (2006) hypothesized that differences in heterotrophic R may be due to changes in moisture because increased moisture may lead to more decomposition of coarse woody debris (Amiro, Barr et al. 2006).

The impacts of drought on the carbon budget are also important in the boreal region. All aspects of the carbon budget including respiration, gross ecosystem productivity, and vegetation development are all affected by conditions when moisture is limited. This is visible in R in 2002 and 2003 data when R is lowest at both F89 and F98 during 2003. This is likely illustrating the water stress experienced that the vegetation would be experiencing during this time.

One of the main reasons that drought has a large impact on the carbon budget is that it is a major controller of the vegetation development. Dry conditions decrease LAI of deciduous species because these species attempt to conserve water by producing fewer leaves. The reduction in LAI will in turn decrease GEP (McCaughey et al. 2006). Reduced LAI can also have a significant impact on R, usually decreasing it (Gaumont-Guay et al. 2006). Daily R is also affected by soil water content and water table depth (Bergeron et al. 2007).

3.4.7 Carbon Stocks

Carbon stock measurements give a good indication of the changes in vegetation and surface characteristics that take place with time following fire. A comparison of carbon stocks measured in 2001 and in 2005 is given in Figures 3.4 to 3.6. Measurements in mineral soil carbon were drastically different between 2001 and 2005 and are likely an artifact caused by inconsistency in sampling or laboratory techniques between the two years and are not shown in Figures 3.4 to 3.6. In comparison to 2001 measurements at F89 and F98, a clear increase in some carbon pools can be observed by 2005. At F89, a larger amount of carbon is present in the above-ground tree biomass in 2005 than in 2001 by approximately 0.4 kg C m^{-2}. The amount of carbon present in the above-ground shrub biomass also increased from 2001 to 2005 by approximately 0.9 kg C m^{-2}. The organic soil layer carbon was still a prominent carbon pool in 2005, however, it was 2 kg C m^{-2} less than that of 2001. Coarse woody debris was the same in 2001 and 2005. The amount of carbon stored in both coarse and fine roots increased slightly in 2005. The total amount of carbon including mineral soil carbon across all carbon pools was approximately 2.2 kg C m^{-2} higher in 2005 than in 2001. When mineral soil carbon was not included, the difference in the total carbon between 2005 and 2001 was -0.2 indicating a carbon loss.

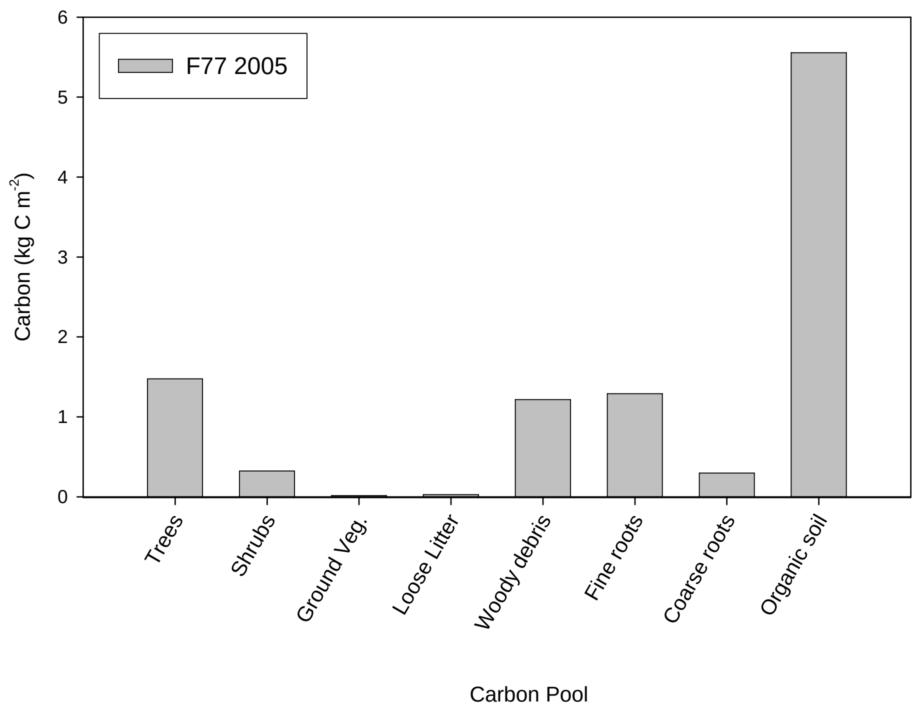

Figure 3.4 Carbon pools measured in 2005 at a boreal forest in central Saskatchewan burned in 1977. Mineral soil carbon measurements are excluded.

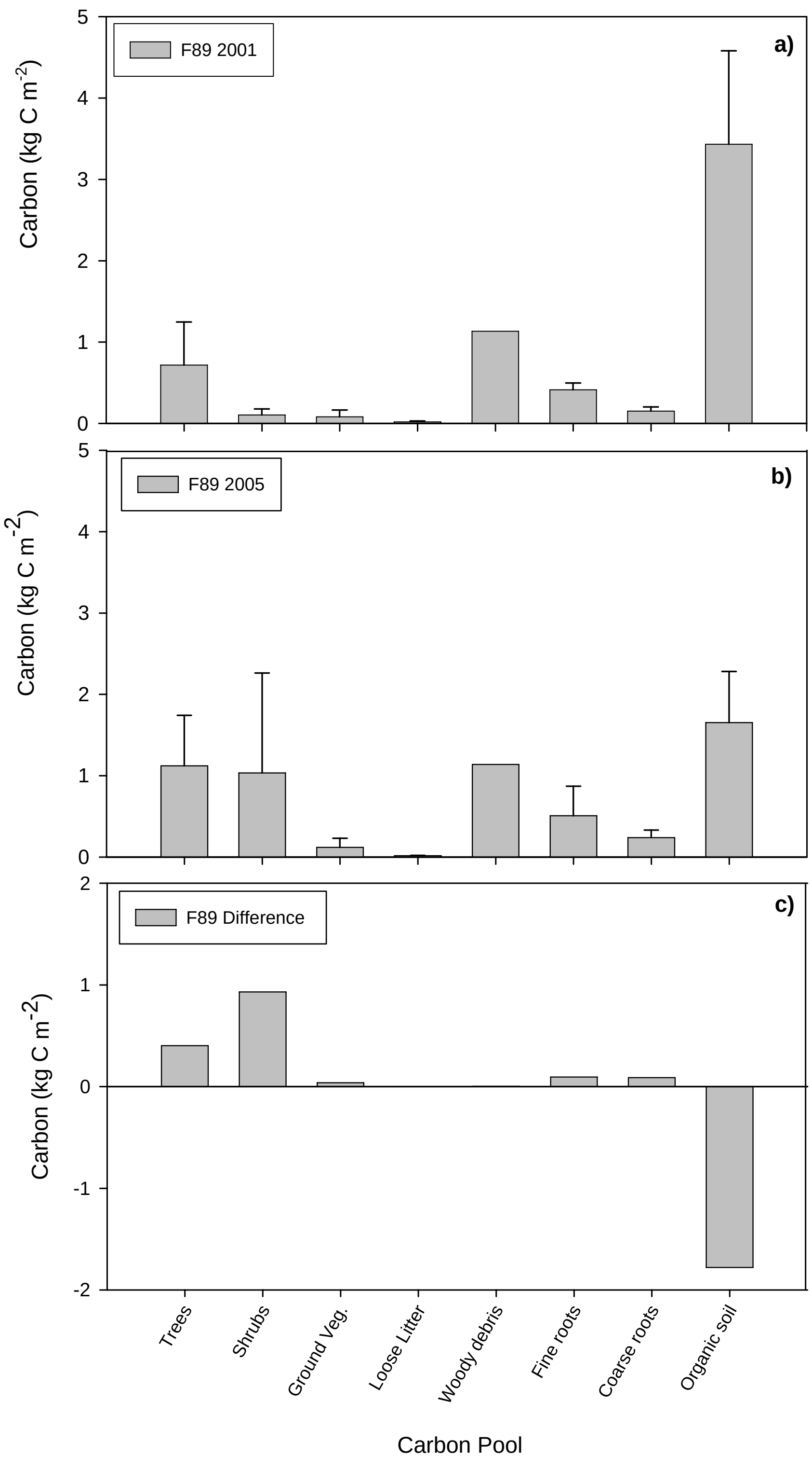

Figure 3.5 a, b, c Carbon pools measured in 2001 (a) and 2005 (b) at a boreal forest in central Saskatchewan burned in 1989. Figure 3.5 c shows the differences observed between measurements taken in 2001 and 2005. Mineral soil carbon measurements are excluded. Error bars represent standard error between plots.

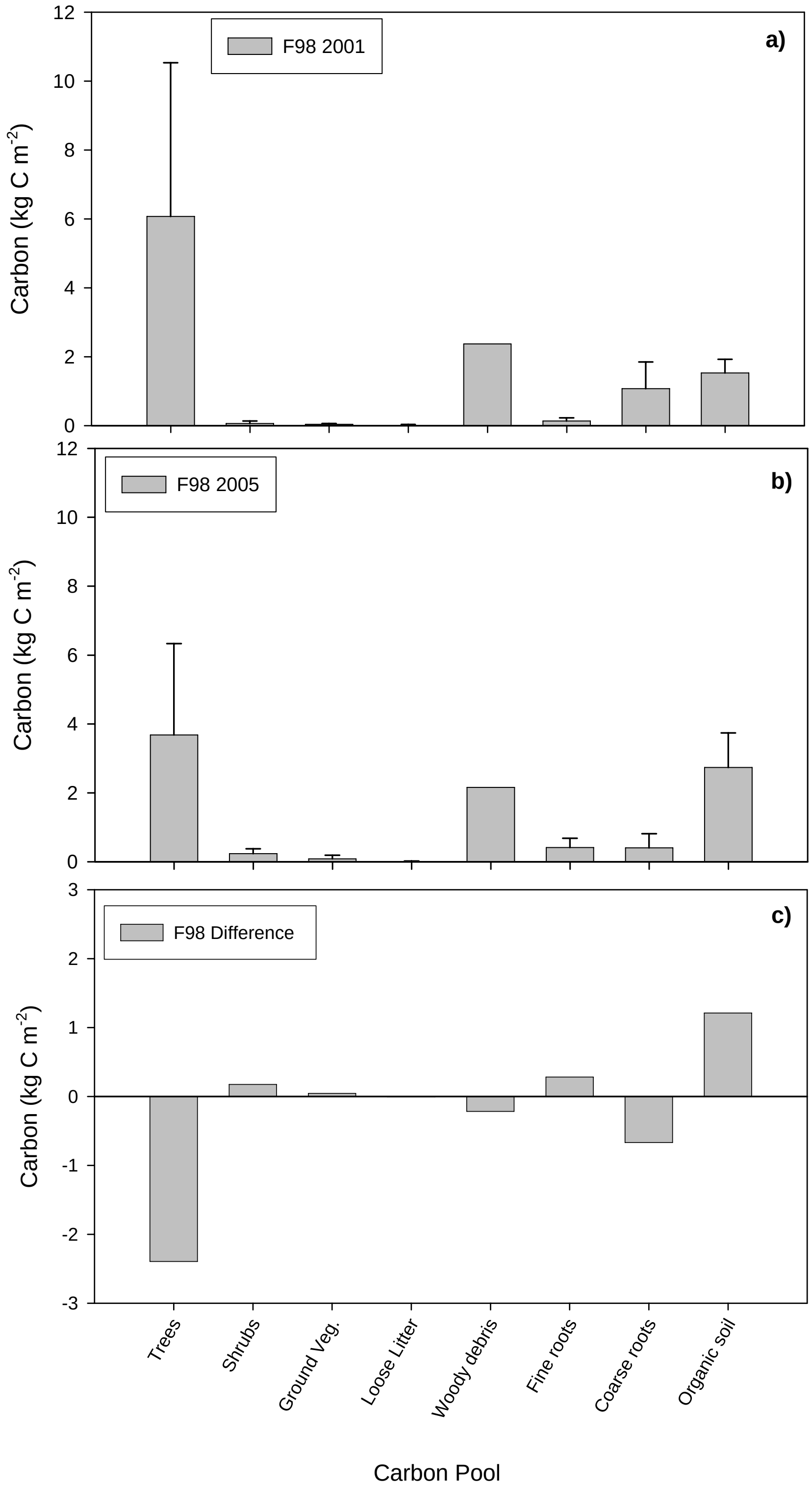

Figure 3.6 a, b,c Carbon pools measured in 2001 (a) and 2005 (b) at a boreal forest in central Saskatchewan burned in 1998. Figure 3.6 c shows the differences observed between measurements taken in 2001 and 2005. Mineral soil carbon measurements are excluded. Error bars represent standard errors between plots.

At F98, above-ground tree biomass was the largest pool in 2001. The total amount of carbon stored as above-ground tree biomass was approximately 2.4 kg C m^{-2} less in 2005 than in 2001. Because most of the trees measured at F98 are dead, it is likely that this decrease in carbon is due to the dead trees falling. Coarse woody debris was almost the same in 2001 and 2005. The amount of carbon present in the organic soil layer was 1.2 kg C m^{-2} higher in 2005 than in 2001. The amount of carbon present as fine roots increased from 2001 to 2005 by approximately 0.3 kg C m^{-2}. This is a realistic increase because, as the forest recovers from fire and as vegetation becomes more abundant and diverse, there would be an increase in fine root biomass as a result of the new vegetation. The amount of carbon present as coarse roots decreased with time by approximately 0.7 kg C m^{-2}. The amount of above-ground shrub biomass also increased over time. An increase of approximately 0.2 kg C m^{-2} was observed from 2001 to 2005. The total amount of carbon in all pools was approximately 0.5 kg C m^{-2} lower in 2005 than in 2001 when mineral soil carbon was included in the calculations of the total. Total carbon in all pools excluding mineral soil carbon was 1.6 kg C m^{-2} lower in 2005 than in 2001. Although we expect the amount of carbon in large trees to decrease slightly because the trees are falling over as time progresses, the loss of C in this pool at this site is much larger than we would expect. This is partially due to the number of trees measured in 2005 compared to the number of trees measured in 2001. In two of the four plots measured, the number of trees measured in 2005 was much lower than 2001, sometimes missing almost half of the trees. Unlike F89, trees at F98 were not tagged so it is uncertain if the trees have fallen over, fallen outside the plot, or were missed during sampling in 2005. This would result in a gross underestimation of the carbon stored in above-ground biomass of trees in 2005.

Unfortunately, no previous carbon stock measurements had been made at F77 in 2001 for comparison although the Canadian Forest Service made measurements in 2003. The largest amount of carbon in measurement at F77 was in the organic soil layer. There was approximately 4.5 kg C m^{-2} of carbon in this pool. The second largest carbon pool at F77 was the above-ground tree matter with 1.5 kg C m^{-2}. Coarse woody debris had carbon values close to those of the trees with a value of 1.2 kg C m^{-2}. The remainder of the carbon at F77 was distributed in mineral soil carbon, coarse roots, fine roots, shrubs, and ground vegetation in order of the greatest amount of carbon to the lowest amount of carbon for a total of 15.4 kg C m^{-2} with mineral soil carbon and 7.8 kg C m^{-2} without mineral soil carbon.

Trees were the most dominant pool of carbon at F98 in comparison to F77 and F89. F98 had approximately 3 kg C m^{-2} more than the other sites. This is expected because the dead standing trees are larger than the trees at the other sites. Woody debris was also largest at F98 due to the large amount of fallen, dead trees. The shrub carbon stock is largest at F89 compared to the other sites. Ground vegetation loose litter pools were similar at all three sites. Fine roots and coarse roots were similar at all sites. Carbon in the soil organic layer was the largest at F77. This is because the vegetation at this site is mature and as a result, there will be more litter falling from the trees and shrubs then at a younger site such as F98 where the vegetation is small and sparse.

Bond-Lamberty et al. (2004) found a consistent increase in most carbon pools including tree foliage, tree wood, understory vegetation, and coarse roots in a chronosequence of well-drained forests near Thompson, MB for 37 years after fire. These pools then saw a decrease in carbon. Fine roots increased in carbon until 71 years following fire and then began to decrease.

Peichl and Arain (2006) found that above-ground tree biomass increased with age at a white pine plantation ranging in age from 2 to 65 years of age. They also found that forest floor carbon increased with age for approximately 15 years and then ceased to show further carbon accumulation in that pool. If the sites used in this study were followed further, we may observe similar results.

3.4.8 Ecological Implications

The ecological impacts of changes in the carbon cycle are numerous. The area burned in forest fires in Canada is expected to increase with climate change (Flannigan et al. 2005). If the area burned increases, the amount of forest acting as carbon sources will also increase. This may have profound impacts on the global carbon budget. A large effort should be made to preserve mature boreal forests so that there will be an increase in the area of forests acting as carbon sinks. However, fire is an important stand renewing agent that is still critical for the regeneration of the site.

An increase in fire frequency may also cause an increase in the amount of deciduous forests which could then lead to increases in transpiration and a decrease in sensible heat flux. The changes in these fluxes could be an overall negative feedback to warming (Chapin et al. 2000). If the abundance of deciduous forests increases, seasonal variations in CO_2 concentration could be amplified (Chapin et al. 2000; Randerson et al. 1999; Zimov et al. 1999).

In comparison to other types of disturbance, such as harvest, fire results in unique stages of succession. Natural regeneration following fire results in a more diverse species composition than harvested stands because aspen suckers recover quickly as do coniferous species such as jack pine and black spruce, which produce serotinous seeds. This will result in a more rich mix of deciduous and coniferous species. As a result, this diversity can cause higher NEP levels than at harvested sites of similar age (Coursolle et al. 2006).

Climate change may also cause the boreal forest to move further north replacing large portions of tundra and could also result in the borders of boreal forest moving northwards as well. This could result in a positive feedback on the atmosphere (Randerson et al. 2006) by increasing the amount of vegetation in the northern regions of the world. The composition of the boreal forest may change as well. There could be a decrease in tree species that are adapted to boreal environments (Barber et al. 2000; Dunn et al. 2006; Wilmking et al. 2004). If the area of boreal forest decreases, so will the area of potential carbon sinks.

3.5 Conclusions

Fire plays an important role in a forest's carbon cycle. Fire releases carbon as CO_2 and other carbon-containing gases during the fire and changes the surface characteristics of the site. As the forest matures, vegetation changes will cause changes in the GEP and R and will have an overall impact on NEP. The size of carbon pools such as trees, shrubs and understory vegetation, soil, and woody debris, will also change with time after fire.

We observed that F98 and F77 were consistent carbon sources, while F89 was a consistent carbon sink. This agrees with studies that found that midday summer NEP fluxes at sites as young as 4 years old were similar to those of unburned, mature sites (Goulden et al. 2006).

Fire results in a forest becoming a carbon source immediately following fire due to the removal of most of the vegetation responsible for storing and exchanging the bulk of carbon in the ecosystem. As vegetation recovers, the carbon exchange slowly decreases its source strength and eventually becomes carbon neutral and then a carbon sink. This has been shown in several studies that have followed mature forests for several years as well as in studies of chronosequences (Amiro, Barr et al. 2006; Bond-Lamberty et al. 2004; Litvak et al. 2003; Pregitzer & Euskirchen 2004; Randerson et al. 2006; Wirth et al. 2002;). Vegetation recovery was illustrated well in the measurements of the carbon pools at the sites discussed here. However, carbon stock measurements did not accurately quantify changes in carbon pools in a short period of only four years. Although the direction of gains or losses of carbon are consistent with NEP results indicating sources or sinks, it is difficult to measure small differences in carbon pools over a short period. An increase in the amount of carbon stored in carbon pools will have an important impact on the overall carbon budget.

Drought played a significant role in the dynamics of the carbon budget. Dry conditions decrease LAI which then decreases levels of photosynthesis and respiration. Warm temperatures experienced during droughts also increase respiration.

When studied with the energy and water budgets, the carbon budget can offer important information on the interactions between the surface and the atmosphere in terms of both gas and energy exchange. Because both the energy and carbon budgets strongly impact one another, it is important to study how disturbance will impact them. The boreal forest is one of the most important carbon sinks in the world and understanding the carbon budget of these boreal forests may shed light on the possible changes that disturbance and climate change may have on a more regional and global scale.

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4. GENERAL SYNTHESIS

Studying the energy, water, and carbon budgets of boreal forests and their recovery after fire is essential to further our understanding of these ecosystems and how they may respond to changes in climate. If fire frequency increases, young, recovering forests will play a large role in determining the regional and global climates in the future. Randerson et al. (2006) studied how impacts of fire in the boreal forest may impact climate change by examining the radiative forcing using a chronosequence located in Alaska and comparing to a mature control site nearby. They used albedo, black carbon on snow and ice, and the effects of greenhouse gases to predict changes that may occur in radiative forcing with climate change. Radiative forcing increased during the first year following fire and decreased over an 80-year fire cycle period. They found that while radiative forcing resulting from increased greenhouse gas emissions has an impact on global temperatures (Ramaswamy and Chen 1997), the impacts resulting from changes in albedo following fire were concentrated in boreal regions (Bonan et al. 1992; Snyder et al. 2004; Brovkin et al. 2004; Ramaswamy and Chen 1997). They predict that fire severity, species composition of recovering post-fire sites, and the duration of snow cover will determine the balance between positive and negative radiative forcing and that fire could even have a negative radiative forcing overall.

Chapin et al. (2000) predict that climate warming could reduce moisture availability and increase surface temperatures and sensible heat flux. These changes in the energy balance could then lead to the depth of the planetary boundary layer increasing resulting in further warming. However, the changes in vegetation and the energy balance that follow fire may combat further warming by increasing moisture availability and decreasing sensible heat flux which could lead to a net cooling (Chapin et al. 2000).

This study has demonstrated that time since fire has an important impact on the energy, water, and carbon budgets of forests following fire. Using eddy correlation methods, we measured energy, water and carbon budgets. With the increased threat of climate change, it has become more important to study the fluxes to help understand how climate change could impact the global climate and the earth's forests and other ecosystems. These flux measurements may also be a powerful tool in helping to predict how to decrease the impacts of climate change and how to slow the process of global warming.

4.1 Energy and Water Budget

Study of the energy budget at the three sites described showed only small differences among sites. The energy budget measurements indicated that a young, post-fire site begins to resemble more mature stands in terms of the energy budget within a decade after fire. The most important factors influencing the energy budget were moisture availability and the changes in albedo and LAI that occur with different stages of succession. Drought has an important impact on both H and LE at the sites studied. LE decreased during times when water availability was low while H increased during these times.

4.2 Carbon Budget

Similar seasonal trends were observed for the carbon fluxes. F89 and F77 had similar weekly NEP fluxes in 2004 and 2005 while F98 had lower values. F89 had high, positive annual NEP indicating a net sink of carbon. F77 and F98 were net annual sources of carbon with F77 being a stronger source than F98. Similar results were observed by (Amiro, Barr et al. 2006). Drought conditions also had impacts on the carbon budget by decreasing the respiration slightly at all three sites studied here as well.

The study of the energy budgets and the carbon budgets of boreal forests are becoming more important as the threat of climate change arises. Because the energy budget has an impact on the carbon budget, it is good that both budgets have been examined here.

It is predicted that the increased air temperatures that will accompany climate change will increase the area burned by forest fire in Canada (Flannigan et al. 2005). An increase in the area burned will make the changes in the energy budgets and the carbon budgets more important on a global scale.

4.3 Future Studies

To expand our knowledge of how boreal forests change over time, studies that observe sites for longer periods of time are required. Ideally, sites would be monitored from the time the fire occurred, for periods of decades. However, this is not always possible. Goulden et al. (2006) found that using the chronosequence approach to studying how forests change following disturbance was an adequate method for observing the broad view of fire recovery after fire but it is difficult in many instances to locate sites close enough to one another so that they are similar in most characteristics other than age (such as soil type, weather, elevation, and drainage). For more accurate comparisons, more site characteristics should be monitored including site hydrology which could strongly influence the energy and water budgets of a site.

This work will allow for future comparisons to the energy and carbon dynamics of other boreal forests that are mature or have been disturbed by other mechanisms. Coursolle et al. (2006) compared the August diurnal patterns of carbon fluxes in mature, intermediate-aged, and young conifer stands, mature deciduous stands, disturbed forests including the three studied here, fens, and a bog. This comparison showed that young post-fire sites had the highest ecosystem respiration. They also found that intermediate-aged coniferous forests had the highest levels of diurnal NEP and GEP. This study is a good example of comparisons that need to be addressed in the future.

Comparisons using these results can be used to compare to energy and carbon studies in recently harvested stands. Because forests recover differently from fire than from harvest, both the energy and carbon budgets will recover differently. One of the main determinants of differences in energy and carbon budgets between post-fire and post-harvest sites is that fire often leaves large debris behind which is left to decompose, whereas harvest removes most of the above-ground large biomass (Amiro 2001).

Further study of the impacts of insect disturbance would also be a valuable addition to the knowledge of the dynamics of boreal ecosystems following fire. Because forests recover differently from fire than from harvest or insects, it is important to study all the various types of disturbance. Studying insect impacts is difficult due to the unpredictability of when an insect outbreak may occur. The increase in temperatures that may accompany climate change may increase the number and severity of insect pest explosions in the boreal region.

4.4 Contribution to Fluxnet-Canada

As a part of the Fluxnet-Canada research network, this study will be a valuable addition to Canada's effort in investigating the energy and carbon fluxes of boreal forests, grasslands, and peatlands. This study includes the only young post-fire sites in the network. The data used in this study have been added to the FCRN Data Information System and is made available to other members of the network, as well as the public on the FCRN website (www.fluxnet-canada.ca).

Study of the energy, water, and carbon budgets in the boreal forest is an important part of examining the carbon dynamics of the earth. Combined with study of the carbon uptake from water bodies and oceans, the study of the release of carbon from fossil fuels, carbon dynamics in agriculture landscapes and tundra ecosystems, we can create an illustration of the global carbon budget. With this knowledge, we can begin to find ways to maximize carbon sequestration and slow the effects of climate change.

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APPENDICES

Appendix I. Gap filling procedures

We followed the Fluxnet-Canada Research Network gap-filling procedures. The following has been copied *verbatim* from Appendix B of Amiro, Barr et al. (2006) with permission from the publisher Elsevier and from the lead author, B.D. Amiro. This section is provided for information only.

I.1 Gap Filling NEP

The procedure first derived GEP and R from half-hourly NEP values, and then filled gaps in half-hourly GEP, R, and NEP using simple empirical models that were constrained by the measured data. The gap-filling procedure made use of two simple annual empirical relationships determined from measured data. One was between R and soil temperature (T_s) at a shallow depth (usually 5 cm) and the other was between GEP and photosynthetically active radiation (PAR) above the stand. In the first step, parameters were obtained for the annual relationships. In the second step, one additional parameter for each relationship was introduced. In order to account for changes in other environmental variables (e.g., soil moisture, vapour pressure deficit) or phenological stage over a short period of time, this parameter was allowed to vary in time with the other parameters held constant. The time-varying parameter was determined within a moving window as the slope of the linear regression (forced through the origin) between estimates of R (and GEP) obtained from the annual relationships and R (and GEP) from the measurements. The implementation of the moving window used a fixed number of acceptable data points (30-min measurements) rather than a fixed period of time. The window was 100 data points wide, moved in an increment of 20 points at a time. For each window, the obtained value of the time-varying parameter was assigned to the mean time of the 100 data points (i.e. usually near the centre of the window). The values of the time-varying parameters for each individual half-hour period were then estimated by linear interpolation. These adjusted relationships were then used to fill the data gaps. The sensitivity of R and GEP to T_s and PAR, respectively, was largely determined by the annual relationships, but was modified to some extent by the time-varying parameters within each window period.

The processing steps to estimate NEP, GEP and R were:

a) NEE was estimated as the sum of the measured eddy (F_c) and storage fluxes (S_c).

b) NEP was estimated as –NEE after excluding low-u_* data at night. In applying the low-u_* data rejection to night-time periods, S_c was included in the estimate of NEP.

c) Small gaps in NEP (four half-hour periods or less) were filled by linear interpolation before larger gaps were filled using the procedure outlined below.

d) Measured R is estimated as R = –NEP during periods when GEP was known to be zero, i.e., during night and during night and day in the cold season (periods when both air temperature (T_a) and T_s are less than 0 ˚C).

e) An empirical logistic R = *f*(T_s) relationship was fit to the measured R values from (d):

$$R = r_1 / (1 + \exp[r_2(r_3 - T_s)]) \tag{I.1}$$

where r_1, r_2, and r_3 are empirical constants.

f) An additional empirical parameter, $r_w(t)$, was introduced into Eq. (I.1) resulting in

$$R = f(T_s, t) = r_w(t) r_1 / (1 + \exp[r_2 (r_3 - T_s)]) \tag{I.2}$$

and $r_w(t)$ was allowed to vary in time (t). It was estimated using a linear regression of the modelled R estimates from (e) versus the R measurements from (d) within a 100-point moving window.

g) The R = $f(T_s,t)$ model (Eq. 2) was used to estimate R during the day and to fill gaps in R at night.

h) GEP was estimated as NEP + R (daytime) or zero (nighttime and during periods when both T_a and T_s were less than 0˚ C).

i) An empirical GEP = f(PAR) model was fit to the non-zero GEP data from (h):

$$GEP = \alpha Q P_x / (\alpha Q + P_x) \tag{I.3}$$

where α is the quantum yield, Q = downwelling PAR, and P_x is the photosynthetic capacity (GEP at light saturation). Both parameters (α and P_x) were treated as constants.

j) An additional empirical parameter, $p_w(t)$, was introduced into Eq. (I.3), resulting in

$$GEP = f(Q, t) = p_w(t) \alpha Q P_x / (\alpha Q + P_x) \tag{I.4}$$

and $p_w(t)$ was allowed to vary in time and was derived using a linear regression of the modelled non-zero GEP estimates from (i) versus the non-zero GEP measurements from (h) within a 100-point moving window.

k) The GEP = f(Q,t) model (Eq. 4) was used to fill gaps in GEP.

l) Gaps in NEP were filled using modelled GEP - R.

I.2 Gap Filling H and LE

We assume that all convective measurements made by the eddy covariance technique were underestimated during low u_* conditions. Hence both H and LE also needed to be corrected using the same threshold values for u_* used for NEP. This keeps a constant Bowen ratio. Gaps of less than two hours (i.e., four half-hour periods) were interpolated.

For H:

A regression was developed between H and R_n-G for good data periods, based on a 240-point (half-hour period) moving window, moved in an increment of 48 points at a time. The missing H data were then filled with the regression equation output.

For LE:

Night values of LE. Most gaps in LE occur at night. Although night-time LE can be slightly positive or negative, for the purposes of daily totals, we set the LE gap values to 0.

Growing season daytime values of LE. We used the same technique as used for H, based on regression between LE and R_n-G using a 240-point moving window of good daytime measurements.

Non-growing season daytime values of LE. LE was not well correlated with R_n-G in winter because it approaches zero. For the non-growing season (T_a and T_s are both <0° C), we gap-filled using the average value for that half-hour period based on the previous 5 days and following 5 days (i.e., 10 points for the same half-hour period). This is appropriate since there are no real physiological controls over LE in winter.

Appendix II. Calculation Of Annual Net Ecosystem Production

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January 19, 2007.

II.1 Introduction

One of the desired outputs of continuous measurements of net ecosystem production (NEP) is the summation of half-hourly periods to calculate the annual total NEP. This is a measure of carbon gain or loss by the ecosystem for the year. The challenge in this calculation is related to the methods used to fill gaps in the data. In the case where flux data are continuous, without instrument malfunction, gaps are mostly caused by the exclusion of data when turbulence is insufficient to measure the vertical flux. We characterize this by excluding data during conditions when the friction velocity (u_*) is below a value known as the u_* threshold. However, gaps are also caused by instrument malfunction and periods of rain and are filled according to the algorithms given in Appendix I.

In addition to these normal gap-filling challenges, we have created large gaps in the data because of concerns over the ability of the open-path infrared gas analyser to give reliable carbon flux data during cold conditions. Some of these features are given by Amiro, Orchansky, Sass (2006), showing that apparent downward fluxes of CO_2 can be measured during cold conditions where we do not have a known sink for CO_2. We still do not know the cause for this phenomenon, nor the true contributing factors. However, we recognize that we need to find some way to calculate annual NEP. Previously, we assumed that respiration at the fire sites during cold temperatures could be estimated by knowledge of respiration at the mature sites in the same region (Old Aspen, Old Jack Pine, Old Black Spruce) (Amiro, Barr et al. 2006). Here, we offer an alternative method to achieve an estimate of annual NEP.

In the following analysis, we arbitrarily exclude all NEP data when the air temperature is less than 0 °C. We recognize that some photosynthesis can still occur below this air temperature because of radiation heating of leaves, but assume that this is small. We then assume that the only exchange process operating below this temperature is respiration, and model this to fill gaps in the cold periods to estimate annual NEP.

II.2 Method

All NEP data (without storage) were excluded when the air temperature at a nominal height of 1.5 m was less than 0 °C. These gaps were then filled using an algorithm for respiration. The algorithm was specific for each site and for each year to capture differences that could be caused by changes in forest vegetation or soil moisture. The algorithm was developed as a 2-parameter exponential relationship between measured NEP at night (when conditions exceeded the u_* threshold) and soil temperature at a depth of 2 cm (mean of 3 locations) such that:

$$R = a\,e^{bT} \tag{II.1}$$

where R = ecosystem respiration measured, T is the soil temperature, and a and b are fitted parameters.

Equation II.1 was developed using data when the air temperature was between 0 and 10 °C to bias the relationship to cooler temperatures. The regression was based on data classed into 1-degree bins, and then a curve fit was done using Sigmaplot (Version 8.02a, Systat Software Inc., 2004). The gaps were then filled using the regression equation for the below 0 °C air temperatures.

We also calculated annual Gross Ecosystem Production (GEP) and ecosystem respiration (R). The gap-filled NEP and GEP were continuous and were summed for the year. Note that GEP was calculated as equal to 0 for below freezing temperatures. The ER was then calculated as R = GEP –NEP to achieve carbon balance.

II.3 Results and Discussion

The regression parameters for Equation II.1 are given in Table II.1. Note that all regression coefficients were reasonably high, with the best relationships shown for F77.

Table II.1 Parameters for exponential curve fit (Equation II.1)

Site and Year	r^2	a	b
F98 2005	0.79	0.4835	0.1711
F98 2004	0.91	0.2766	0.2329
F98 2003	0.75	0.2355	0.2488
F89 2005	0.80	0.5907	0.2228
F89 2004	0.92	0.3316	0.2966
F89 2003	0.69	0.3436	0.2240
F77 2005	0.95	0.980	0.1764
F77 2004	0.93	0.917	0.2005

Using the parameters from Table II.1, we calculated annual NEP, GEP and ER

II.4 References

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