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When do replanted sub-boreal clearcuts become net sinks for CO_2?

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Abstract

After forest harvesting, sites are initially sources of CO_2, but eventually become sinks for CO_2 after some period of years following reforestation. This period for boreal forests has been assumed to be 10 years, but this has not been validated empirically for most forest types including sub-boreal spruce-dominated forests of central British Columbia, Canada. Therefore, we sought to determine the timing of the source to sink transition for a sub-boreal clearcut. Clearcuts such as the one documented in this study occurring on glaciolacustrine deposits with relatively poor drainage represent about 20% of the 1.5 million ha in the Prince George area. Net ecosystem CO_2 exchange (NEE) for a clearcut was measured over four growing seasons in years 5, 6, 8 and 10 after harvest. A Bowen ratio approach in combination with a bottom-up modeled NEE based on ecosystem component CO_2-flux measurements was used for years 5 and 6. In years 8 and 10, growing season NEE was measured using an open-path eddy covariance system. A cross comparison of Bowen ratio and eddy covariance systems was performed and measurements agreed relatively well ($r^2 = 0.58$). The results demonstrated that while this clearcut was still a source for C (NEE of +336 to +487 g C m^{-2}) after 6 years, it was most likely a sink for C between 8 (NEE of −189 to −52 g C m^{-2}) and 10 (NEE of −185 to −48 g C m^{-2}) years following harvest.

Keywords: Clearcuts; Sub-boreal; Eddy covariance; Bowen ratio; Carbon

1. Introduction

In many of Canada's conifer forests, particularly those in wetter regions with long fire return intervals and/or with long-standing fire suppression activity, forest harvesting is now becoming the dominant disturbance agent. For example, conifer forests of British Columbia now provide nearly 40% of the total timber harvest for Canada (NRCAN, 2004), 70% of this from the province's interior, nearly one quarter of which is from the sub-boreal Prince George Forest District alone (British Columbia Ministry of Forests, 2004). As a consequence, what was once a predominance of older age-classes (MacKinnon and Vold, 1998) is in many locations now undergoing a rapid conversion to younger planted second growth stands (Burton et al., 1999).

Boreal forests contain more carbon (C) than any other global forest type (559 Gt C) and more than 22% of all terrestrial vegetation and soil C stocks (Watson et al., 2000). At the same time, deforestation has contributed approximately one third of the total increase in atmospheric CO_2 since preindustrial times (i.e. pre-1850), with 20% of the increase occurring within the last decade (Watson et al., 2000). Many studies now link forest harvesting to further increases in CO_2 levels through depletion of C-rich old-growth stands and their replacement with young C-poor plantations (Thornley and Cannell, 2000; Harmon and Marks, 2002) and through shortening of forest rotation intervals (Price et al., 1997; Liski et al., 2001). Rising CO_2 in the atmosphere, in combination with contributions from other greenhouse gases, is currently believed to be the primary cause of the current global warming trend (Watson et al., 2000; Prentice et al., 2001).

Carbon models of northern conifer-dominated forests have used 10 years as the point at which secondary succession forest stands begin to be net sinks for C after harvesting or disturbance (Kurz and Apps, 1994), with shorter intervals for boreal deciduous than conifer stands (Amiro, 2001), but empirical data to support this assumption are scarce. In a chronosequence of boreal Jack pine (*Pinus banksiana* Lamb.) stands in central Saskatchewan, Monte Carlo simulations of net ecosystem productivity based on belowground CO_2 fluxes and carbon

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stocks suggested that 0 and 5 years old stands are sources for CO_2 (net loss of CO_2 to the atmosphere) whereas 10-year old stands are a slight C-sink (Howard et al., 2004). Similarly, a small C-sink was observed for northern boreal stands in Manitoba 11-year post disturbance (Litvak et al., 2003) and consistent with the slightly negative annual net ecosystem CO_2 exchange (NEE) observed for a 12-year-old boreal Scots pine stand in southern Finland (Kolari et al., 2004). In British Columbia, coastal stands of young Douglas-fir plantations are still net sources of CO_2 after 3 years (Humphreys et al., 2005) but the year of transition to sink status has yet to be confirmed.

A long-term CO_2-flux and carbon stocks study was established in 1999 at the Aleza Lake Research Forest to determine when replanted sub-boreal spruce and fir forest clearcuts of central British Columbia become net sinks for CO_2. Growing season Bowen-ratio ecosystem CO_2-fluxes as well as plant and soil CO_2-exchange measurements combined with previous measurements of winter CO_2 fluxes showed that in years 5 and 6, the clearcut was still a net source for CO_2 (Pypker and Fredeen, 2002a,b). In this paper, we have added to this study with measurements of NEE for the same clearcut in years 8 and 10 after harvesting with the intent of documenting its transition from C source to sink.

2. Methods and materials

In this study we examined ecosystem level fluxes from an 85 ha cutblock in the Aleza Lake Research Forest (ALRF) located approximately 60 km east of Prince George (54°03′11″N, 122°03′40″W) which was clearcut harvested in the winter of 1994. The clearcut, in the conifer-dominated intermontane sub-boreal region of central British Columbia, was planted largely to 1- to 2-year-old interior hybrid spruce (*Picea glauca* (Moench) Voss × *engelmannii* Parry ex Engelm.) seedlings in the summer of 1995 at a density of approximately 1200 seedlings ha^{-1}. The original primary forest was dominated by subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) and interior hybrid spruce with lesser inclusions of veteran Rocky Mountain Douglas-fir (*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco) on upper landscape positions and frequent white birch (*Betula papyrifera* Marsh.) in old-growth canopy gaps. The climate is characterized by having a mean annual temperature of 3 °C and mean annual precipitation of approximately 900 mm, with approximately one third of this as snow (Murphy, 1996). Soils across the site were classified predominantly as a complex of imperfectly drained Gleyed Gray Luvisols and poorly drained Orthic Luvic Gleysols with silty clay loam textures (P. Sanborn, UNBC, unpublished), a common complex over much of the ALRF (Arocena and Sanborn, 1999).

In 1999, a 1 ha plot was established within a relatively flat spruce-planted area with a fetch of at least 300 m in all directions in the eastern end of the 85 ha cutblock. At 5 (1999) and 6 (2000) years from harvest, a Bowen-ratio energy balance (BR) system (023/CO_2, Campbell Scientific, Logan UT), was installed in the center of the plot for two consecutive growing seasons (27 June to 3 September 1999 and 24 May to 21 September 2000). In brief, the BR system was mounted on a 3.2 m tall metal tripod tower with a minimum fetch of 300 m in all directions. The infrared gas analyzer (LI-6262, LI-Cor Inc., Lincoln, NE) was mounted on the tripod and calibrated three times per week with a 427 μmol CO_2 mol^{-1} standard gas. The top arm of the BR system was mounted at 2.8 m and the lower arm was mounted 1.4 m below, well above the height of the seedling canopy of approximately 0.7 m in 2000. Net radiation was measured with a Q7 net radiometer (Campbell Scientific) mounted 3.0 m from ground level. Air temperature was measured using 75 μm chromel-constantan thermocouples mounted at the end of each arm. Soil heat flux was measured using two soil heat flux plates placed 8 cm below the ground surface (HFT-3, Campbell Scientific). Soil temperature was measured using four soil thermocouples (TCAV, Campbell Scientific) placed in pairs at 2 and 6 cm depths, as well as at 10 cm at three additional locations (see Pypker and Fredeen, 2002a). Wind speed and direction were also measured at a height of 3.2 m (Wind Sentry, RM Young, Campbell Scientific), relative humidity at 2.1 m (HMP35C, Campbell Scientific), and rainfall with a tipping rain gauge (TE525WS, Campbell Scientific). A full account of the methods can be found in Pypker and Fredeen (2002a,b).

In 2002 and 2004, 8 and 10 years after harvesting, an eddy covariance (EC) system (Campbell Scientific) was installed in the same clearcut location for 6 June to 28 August 2002 and 26 May to 8 October 2004, respectively. The decision to move to an EC approach for years 8 and 10 was driven by its common use for measurement of NEE in the rougher canopies of young and mature conifer forests (e.g. Litvak et al., 2003) and the problems inherent in obtaining good BR measurements from dusk until dawn (e.g. Pypker and Fredeen, 2002b). Net ecosystem CO_2 exchange (NEE) was measured using an open path IRGA (LI-7500, LI-Cor Inc.), accurate within ±0.3 μmol mol^{-1}, and mounted at 2.2 m on a 2.9 m tall tower, well above the height of the spruce seedling crowns (∼0.9 to 1.2 m in height). Wind speed and direction were measured in three dimensions using a sonic anemometer, accurate to ±4 cm s^{-1} (CSAT3, Campbell Scientific) mounted at 2.7 m. Air temperature was measured using a 1.27 μm chromel-constantan thermocouple mounted on the sonic anemometer. Data was averaged over 15 min intervals and stored on a datalogger (23X, Campbell Scientific).

The EC method is based on measurement of gas flux densities, in this case CO_2. The mean flux density for CO_2 (F_c) was computed as the product of the density of CO_2, ρ_c' (kg m^{-3}) and the vertical component of the wind speed w' (m s^{-1}) as it fluctuates from mean values. Both of these variables were measured directly using the EC system and F_c calculated using the Campbell Scientific EC system software.

2.1. Data processing and correction

After correcting for bad or missing data, F_c was assumed to be equivalent to NEE for both BR and EC systems. Data were lost from the BR and EC measurements due to periods of low wind speed, precipitation events, or changes in boundary layer stability, such as at dawn and dusk (Ham and Knapp, 1998). Gaps in the BREB data set were filled using a bottom-up modeling approach (Pypker and Fredeen, 2002a,b). For gaps in the EC data, gap-filling used a combination of interpolation from surrounding measurements for small gaps (less than 3 h) and a regression between CO_2 flux and solar radiation during the day and mean night time respiration values from previous and following nights for larger gaps based on the methods of Ham and Knapp (1998). Almost 28% of the data required filling in 2002 and <25% of the data required filling in 2004. Net growing season flux error estimates were based on the range of data between the maximum and minimum values observed and the mean net flux error was based on the standard error of the estimate.

To establish the level of congruence between BR (1999, 2000) and EC (2002, 2004) system measures of NEE, both systems were operated simultaneously for a week (9–16 June 2001) in a sub-boreal pasture site approximately 60 km south east of the ALRF (53°20′N, 122°35′W) (Waughtal, 2003). The site had a minimum of 450 m fetch in all directions, which is well within the distance needed for equilibrium to be reached in the boundary layer (Gash, 1986). The pasture was cleared and pile-burned 18 years prior to measurement (1985) and was subsequently seeded with timothy grass (*Phleum pratense* L.). Other species present in the pasture included buttercup (*Ranunculus acris* L.), clover (*Trifolium hybridum* f. allioideum Dore), orange hawkweed (*Hieracium aurantiacum* L.) and other grass and sedge species (*Bromus*, *Festuca*, *Poa*, and *Carex*). The soils on the site were predominantly Dark Gray Luvisols (P. Sanborn, UNBC, unpublished). Flux comparisons were made between hourly mean instantaneous NEE measured from 08:00 and 17:00 PST.

2.2. Modeling of non-measured fluxes

Estimates of growing-season fluxes were modeled for those days where fluxes were not measured. Based on prior experience in the area, the year was divided into halves; a 6-month 'growing season' extending from 1 May to 31 October, generally coincident with the period of snow-free soils and above-freezing temperatures, with the remaining spring, fall and winter months classed as non-growing season months. Non-measured growing season fluxes were estimated by using standard least squares multiple regression models relating proximal fluxes for a given year to meteorological data collected in a clearcut site ~8 km north-east of the research site (JMP5.1, SAS Institute Inc., NC, USA). Proximal daily NEE and totals or means of radiation (W m^2), soil temperature (°C; 10 cm depth), precipitation (mm), and mean, maximum and minimum air temperatures (°C; 0.15 m in height above the soil surface) were first all entered into the model in a backwards stepwise direction, with variables rejected from the model with a t-test $p > 0.10$. These microclimatic variables were considered for our models because they were available for the flux site area and because radiation, precipitation and temperature are typically key environmental variables correlated with respiration and photosynthesis (e.g. Pypker and Fredeen, 2002a,b, 2003). Growing season NEE multiple regression equations for years 5 and 6 were previously stated in Pypker and Fredeen (2002b) and were generated in a similar fashion to those in years 8 and 10. For years 8 and 10, least squares multiple regression equations were generated for spring and fall data to extrapolate non-measured intervals. Since 2002 had a much shorter period of measurement in spring and fall than 2004, 2004 equations were used to predict 2002 NEE for the earliest and latest portions of the 2002 growing season as indicated in Table 1 to reduce error associated with excessive extrapolation in 2002. Early spring and late fall were problematic because they encompassed periods of the year when deciduous vegetation growth occurs rapidly after snow melt (early spring) and die-back equally rapidly after frosts (late fall). Thus, the use of the 2004 regression equations seemed warranted for 2002 in the early and late fall periods when no proximal measurements of NEE were available in that year.

For non-growing season (1 November to 30 April) fluxes we generated low and high estimates in an attempt to capture the range of possible annual NEE flux totals. The 'low' NEE estimate, non-growing season fluxes were assumed to be a simple mean winter rate (i.e. +0.6 μmol m^{-2} s^{-1}) measured at the ALRF during previous studies (Evans et al., 1998). This rate was always lower than modeled 'high' estimates. For the 'high' estimate of non-growing season fluxes, we used two different relationships between soil temperature and belowground CO_2 fluxes. For the fifth year following harvest we used a function relating belowground respiration to soil temperature as

Table 1
Multiple regression equations $\pm s_{Y.X}$ with correlation coefficients used to predict net ecosystem CO_2 exchange (NEE) in spring and fall of the 2002 and 2004 growing seasons

Year	Multiple regression equations[a]	R^2
2002 spring[b] (15 May to 5 June)	$\text{NEE} = -0.0152a + 0.391c - 3.01 \pm 0.98$ ($n = 17$; $p = 0.0003$)	0.68
2002 fall[c] (28 August to 30 September)	$\text{NEE} = -0.0111a + 0.109b + 0.194e - 1.591 \pm 0.37$ ($n = 12$; $p = 0.0006$)	0.87
2004 spring[d], and 2002 spring (1–15 May only)	$\text{NEE} = -0.0358a + 0.377b - 0.305d + 1.386 \pm 0.79$ ($n = 16$; $p < 0.0001$)	0.84
2004 fall[e], and 2002 fall (1–31 October only)	$\text{NEE} = 0.0985b + 0.172c - 0.0247e - 1.588 \pm 0.23$ ($n = 20$; $p = 0.0003$)	0.48

[a] a: radiation (W m^{-2} day^{-1}); b: average air temperature at 15 cm (°C); c: average soil temperature at 10 cm (°C); d: minimum air temperature at 15 cm (°C); e: daily rainfall (mm).

[b] Modelled NEE data were 7 June to 23 June 2002.

[c] Model NEE data were from 16 August to 27 August 2002.

[d] Model NEE data were from 27 May to 13 June 2004.

[e] Model NEE data were from 16 August to 27 August 2004.

measured in the spring and fall of 1999 (Pypker and Fredeen, 2002a). The relationship between soil temperature and below ground respiration in 2000 was used to predict the CO_2 fluxes for the remaining 3 years (2000, 2002 and 2004) because, (1) measurements of component belowground CO_2 fluxes were not made in 2002 and 2004, and (2) spring and fall NEE was highest in 2000 among the 3 years making it more likely that this function would provide an upper bound for positive CO_2 fluxes (Pypker and Fredeen, 2002b), and (3) previous work in this area had shown that belowground fluxes were relatively invariant across clearcuts from 0 to 10 years after harvest (Pypker and Fredeen, 2003).

3. Results

Between-year comparisons of Bowen ratio (BR) measurements of NEE in 1999 and 2000 and eddy covariance (EC) measurements of NEE in 2002 and 2004 relied on a level of equivalency of BR and EC NEE estimates. Hourly mean diel EC NEE (NEE_{EC}) plotted against BR NEE (NEE_{BR}) exhibited a relatively linear relationship ($r^2 = 0.58$), with NEE_{EC} being less negative than NEE_{BR} at 'high' flux levels (<-6 μmol m^{-2} s^{-1}), similar to NEE_{BR} at mid-range flux levels, and lower than NEE_{BR} at 'low' flux levels (>-6 μmol m^{-2} s^{-1}) (Fig. 1). Overall, mean diel NEE_{BR} (−6.7 μmol CO_2 m^{-2} s^{-1}) for the site was 11% higher (more negative) than NEE_{EC} (−6.0 μmol CO_2 m^{-2} s^{-1}). No adjustments were made in NEE estimates since it was unclear from our analysis whether BR estimates were 'high' or EC estimates were 'low'.

Average diurnal trends in instantaneous NEE (μmol CO_2 m^{-2} s^{-1}) were computed for the planted clearcut 5, 6, 8, and 10 years post-harvest for a 14-day period in mid-July (Fig. 2A) and mid-August (Fig. 2B). General trends in mid-growing season fluxes were broadly indicative of the growing sink capacity of the clearcut with time since harvest. For example, both years 5 and 6 post-harvest had lower cumulative diurnal C uptake in July (less negative NEE: −1.28 and −0.45 g C m^{-2} day^{-1}, respectively) than at 8 and 10-year post-harvest (−3.28 and −2.77 g C m^{-2} day^{-1}, respectively).

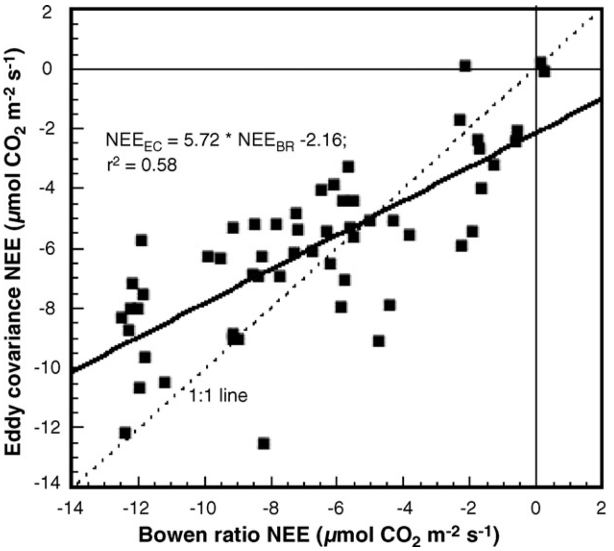

Fig. 1. Congruence between eddy covariance (EC) system measures of net ecosystem CO_2 exchange (NEE_{EC}) and Bowen ratio (BR) system measures of NEE (NEE_{BR}) from 9 to 16 June 2001 in a sub-boreal pasture site in central British Columbia ($s_{Y.X} = 1.83$).

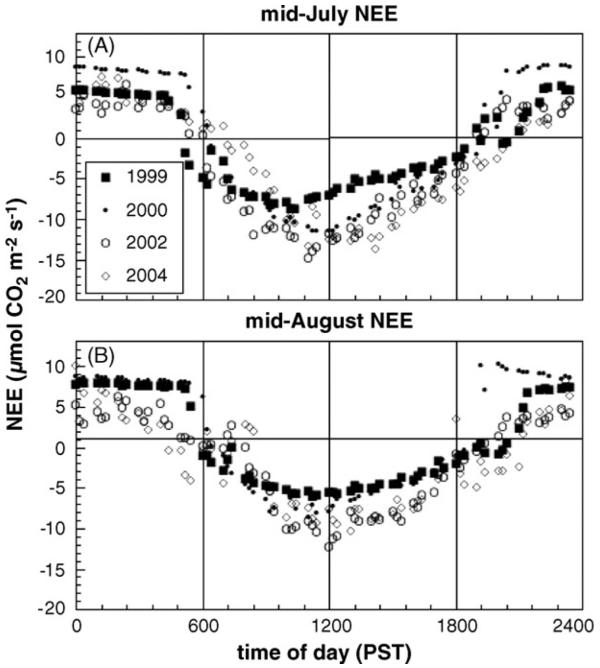

Fig. 2. Comparison of 2-week diurnal averages of (A) mid-July and (B) mid-August net ecosystem CO_2 exchange (NEE) for a sub-boreal clearcut 5 (1999), 6 (2000), 8 (2002), and 10 (2004) years post-harvest measured using Bowen ratio (solid symbols) and eddy covariance (open symbols) approaches. Coefficients of variation for means of midday NEE averaged 34%, 47%, 49%, and 52% in years 5 through 10, respectively.

Trends in measured daily totals of NEE (g C m^{-2} day^{-1}) across the growing season intervals of 1999, 2000, 2002, and 2004 suggest a high level of interannual variability (Fig. 3A–D, respectively), even after growth of spruce seedlings is considered. For example, the clearcut in 2000 was in all

Table 2
Total net ecosystem CO_2 exchange (NEE; g C m^{-2}) from a sub-boreal clearcut 5 (1999), 6 (2000), 8 (2002) and 10 (2004) years after harvest measured over a 63-day growing season interval in common between all years (27 June to 28 August) and estimated based on measured and modeled NEE for a 6-month 'growing season' interval (1 May to 31 October) or for the entire calendar year (1 January to 31 December)

Year	Measured 63-day growing season NEE	Estimated 6-month growing season NEE[a]	Estimated annual NEE[b]
1999	−29	134	208–243
2000	56	227	336–487
2002	−116	−298	−189 to −52
2004	−186	−294	−185 to −48

[a] Non-measured growing season fluxes were estimated using multiple regression models based on proximal fluxes and Aleza Lake Research Forest meteorological data as in Pypker and Fredeen (2002a,b).

[b] Low estimates of non-growing season fluxes (November–April) were taken as those from Evans et al. (1998); High estimates of non-growing season fluxes were estimated based on a 1999 belowground respiration model (Pypker and Fredeen, 2002a) for 1999, and a similar model based on soil respiration data for year 2000 for other years (Pypker and Fredeen, 2002b).

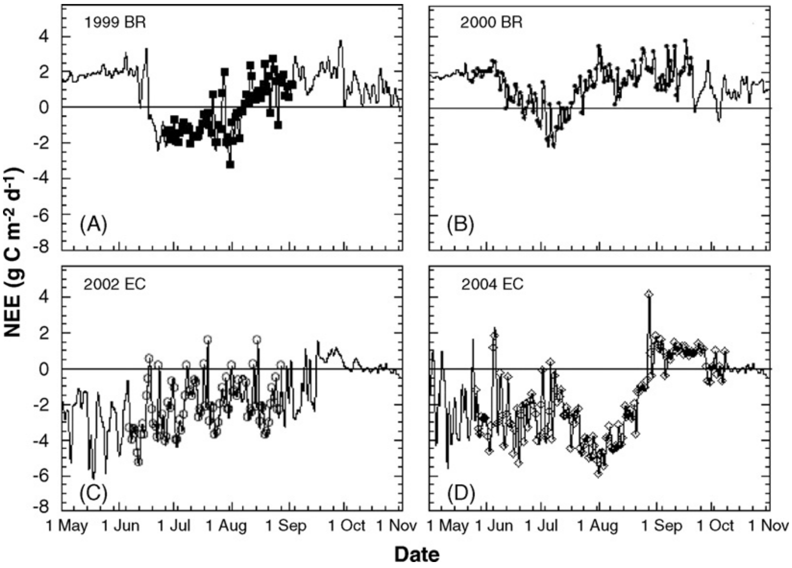

Fig. 3. Daily sums of measured (symbols) and modeled (solid line) net ecosystem CO_2 exchange (NEE) for a sub-boreal clearcut for growing seasons 5(A), 6(B), 8(C), and 10(D) years post-harvest using Bowen ratio (BR: solid symbols) and eddy covariance (EC: open symbols) approaches.

respects a greater C source than in 1999, and similarly, 2004 showed little increase in sink status over 2002. Never-the-less, growing season daily net uptake of CO_2 in the clearcut was greater across July and August and was negative earlier as well as later in the season in years 8 and 10 than in years 5 and 6, consistent with the expected increase in C sink status of the clearcut with years since harvest. Furthermore, when we examined the 63-day growing season interval (27 June to 28 August) that was common to all years, NEE was generally more negative with times since harvesting with the exception of year 6 (2000) that was also the only year in which NEE was positive over the 2-month interval (Table 2). Over the growing season, non-measured daily NEE values were estimated (Fig. 3A–D) and added to measured NEE daily totals to obtain total ‘growing season’ NEE estimates (Table 2).

4. Discussion

The question of when clearcuts become sinks for CO_2 after harvest is of interest to carbon budget modelling efforts (e.g. Kurz and Apps, 1994) to establish the importance of forests and forest disturbance to landscape-level carbon budgets, and ultimately to greenhouse gas contributions to the atmosphere resulting from forest management activities. Forest harvesting is becoming the most important and consistent disturbance type in forest ecosystems across in British Columbia (BC) with more than 90% of this harvesting occurring through clearcut harvesting within BC and also across Canada. Mean annual areal forest harvest rates in BC currently stand at approximately 190,000 ha year^{-1} (mean: 1990–2002), greatly exceeding that resulting from fire (65,000 ha year^{-1}, mean: 1970–2003), and possibly even insects (1,110,000 ha year^{-1}, mean: 1975–2003); were actual areas of mortality as opposed to affected trees known in the latter (e.g. Hall and Moody, 1994; Armour et al., 2003; Hogg et al., 2002; Canadian Council of Forest Ministers, 2006). Irrespective of the relative magnitudes of C losses, forest harvesting, and in particular clearcut harvesting, cannot be ignored in the accounting of anthropogenic C emissions. This is of particular importance when harvesting occurs in C-rich old-growth or primary forests such as are prevalent in wetter sub-boreal spruce and fir forests encompassing the clearcut in the present study (Burton et al., 1999) and where losses of C resulting from this harvesting are substantial due to near complete removal or ultimate loss of tree biomass and coarse woody debris C from the sites (Fredeen et al., 2005).

In this study, measurements of clearcut NEE made with a Bowen ratio energy balance system (BR) in years 5 and 6 were contrasted with NEE measurements made with an eddy covariance (EC) system in years 8 and 10 after harvest. While there was considerable uncertainty in annual NEE for years 8 and 10, there were a couple of reasons to have reasonable confidence in the overall results. First, while the BR and EC equipment and approaches were not found to be equivalent in our study, as has been observed by others (e.g. Rana and Katerji, 1996; Liu and Foken, 2001), our cross-comparison of the two systems suggests that if a bias had resulted, it would have been to underestimate the measured C fluxes of years 8 and 10, and therefore the magnitude of the C sink (relative to the BR estimates), rather than inflate them. Secondly, measured growing season total and diel NEE were substantially more negative in years 8 and 10 than in years 5 and 6, consistent both with the increasing vegetation and therefore photosynthetic C uptake on site, and to a lesser extent with reductions in heterotrophic respiration including decomposition of coarse woody debris and large tree root after harvesting, though this has not been seen for belowground respiration in chronosequence studies of others (Kolari et al., 2004) as well as by the authors (Pypker and Fredeen, 2003).

The clearcut was in most respects found to be a C-source early on after harvesting (years 5 and 6 post-harvest; Table 2), corroborating previously determined annual totals for these 2 years using different estimation techniques for non-measured and non-growing fluxes (Pypker and Fredeen, 2002a,b). However, the same clearcut was primarily or entirely in the negative or C-sink range in years 8 and 10 post-harvest, respectively, suggesting that the clearcut was a probable C sink by year 8 and more conclusively a sink by year 10. Our results for years 8 and 10 generally corroborate the model determination for the timing of transition of clearcuts from source to sink (Kurz and Apps, 1994), as well as results from a variety of chronosequence studies: (1) net ecosystem productivity simulations for boreal Jack pine (*P. banksiana*) stands suggested that 0 and 5 years old stands were sources for C while 10 years old stands were small C-sinks (Howard et al., 2004), (2) flux measurements from boreal forest in northern Manitoba indicated a small C-sink 11 years post disturbance (Litvak et al., 2003), and (3) net annual fluxes of CO_2 for a boreal Scots pine stand in Southern Finland were slightly negative 12-year post-harvest (Kolari et al., 2004). While still in the early stages, ecosystem fluxes from a Douglas-fir plantation 3-year post harvest in coastal British Columbia indicated that it was still a net source for CO_2 (Humphreys et al., 2005). Finally, our annual estimates for NEE for our sub-boreal clearcut in years 8 and 10, though spanning a large range from near C neutrality (-52 to -48 g C m^{-2}) to sink (-189 to -185 g C m^{-2}) status, bracket the expected NEE values for evergreen forests of various ages with carbon uptake periods (i.e. growing seasons) of approximately 6 months (Churkina et al., 2005).

The 6-month time interval chosen to reflect the growing season of the clearcut (1 May to 31 October) is largely reflective of the average snow-free period with soil temperatures >0 °C and plant physiological activity in the northern central plateau of BC. However, it was otherwise subjective in that neither vegetation, which became progressively conifer-dominated with time, nor growing season climatic conditions between years, were equivalent. This was evident in the non-linear progression of measured NEE (63-day period of measured growing season NEE in common across all years) as well as the measured plus estimated (6-month ‘growing season’) NEE with time since harvest. Both measured 63-day ‘growing season’ and estimated 6-month ‘growing season’ NEE was less negative (i.e. less ecosystem-level net photosynthesis) in year 6 than in year 5, a result which was previously shown to be largely attributable to higher belowground respiration and not lower photosynthesis in 2000 than in 1999 (Pypker and Fredeen, 2002b). The trends in growing season NEE (Fig. 3) also suggest that the clearcut in years 5 and 6 had more days with net ecosystem respiration rates in spring and fall combined with shorter periods of net ecosystem photosynthesis in the summer when compared to years 8 and 10.

The larger issue of the implications of clearcut harvesting to C emissions is related to two separate primary processes, loss of mature or old-growth C stocks resulting from forest harvesting and the loss of net photosynthetic uptake of CO_2 during the re-establishment phase after harvesting. With respect to the former, previous work in sub-boreal forests suggests that harvesting of primary forest, most of this old-growth, results in losses of 18,000–27,000 g C m^{-2} (Fredeen et al., 2005). This can be compared to the losses associated with lost net ecosystem productivity after harvest. If we assume conservatively that old-growth forests are neutral with respect to CO_2 uptake, then by plotting our four average annual NEE against time since harvest, adding in a belowground respiration point for a new clearcut at the ALRF (560 g C m^{-2} $year^{-1}$; Pypker and Fredeen, 2003), we can integrate under a curve to obtain total C lost during re-establishment. Based on a linear regression equation fit to this data [annual NEE (g C m^{-2}) = -97 (years since harvest) + 804; $r^2 = 0.67$], our numbers would indicate that our clearcut is a source of CO_2-C for 8.3 years ($p = 0.15$) to the amount of nearly 3.3 kg C m^{-2}. Though this loss is not trivial, it is 6.3- to 9.4-fold lower than the C lost through depleted C stocks after clearcut harvesting (Fredeen et al., 2005). While mid-aged stands are generally more vigorous C sinks than old-growth stands (Coursolle et al., 2006), managed forests with relatively short harvest intervals will both increase the frequency of clearcutting C flux losses and reduce the age and maximal C stocks of these forests (Kurz et al., 1998; Harmon and Marks, 2002).

The soil at the specific location of our clearcut instrumentation was part of a larger landscape mosaic of imperfectly to poorly drained Gleyed Luvisols and Luvic Gleysols. These soils have formed on fine-textured glaciolacustrine sediments, that are the predominant soil parent material in the ALRF, and are widespread in the Fraser River valley both above and below Prince George, as well along other major valleys in the central interior (P. Sanborn, UNBC, personal communication). The latest Prince George soil survey report (Dawson, 1989) indicates that map units in which fine-textured glaciolacustrine materials predominate occupy about 20% of the 1.5 million ha covered by the report. The majority of these areas have imperfectly to poorly drained soils (Gleyed subgroups of Luvisols, Gleysols) as dominant or significant components of the map units. Because of this distribution pattern, such fine-textured materials occupy a large proportion of the areas that were first settled and logged in the central interior, and because of this accessibility, such sites are more likely to be more intensively managed than more remote areas in the future. Unpublished carbon stocks data for the ALRF (C. Bois, UNBC) are suggestive of highest total (forest + soil) carbon stocks on subhygric sites and lowest total carbon stocks on hygric sites. Since our flux site contained both subhygric and hygric conditions within the flux tower footprint, it is possible that the carbon sequestration we observed here is in an average sense reflective of cutblocks occurring over a broader range of moisture regimes in the relatively wet and cool sub-boreal forests of central British Columbia.

5. Conclusions

Based on our estimates of annual NEE, sub-boreal clearcuts appear to be clear sources for CO_2 5 and 6 years after harvesting that become sinks for CO_2 at some time between 8 and10 years post-harvest. Coincident with this apparent source to sink transition, vegetation in the clearcut changed greatly between 1999 and 2004 with a predominance of deciduous herb and shrub vegetation in year 5 giving way to a predominance of spruce by the end of year 10.

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