Carbon Dynamics of a New England Temperate Forest Five Years After Selective Logging

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Abstract

Rising atmospheric carbon dioxide levels are mitigated by carbon sequestration in the terrestrial biosphere, including temperate forests in the northern hemisphere. Many of the second-growth mixed hardwood forests that cover large parts of eastern North America are significant carbon sinks and are under management that includes harvesting for economic gain. This study examines the effect of a timber harvest typical of the North Quabbin region in north central Massachusetts on the carbon source-sink dynamics of the stand, both annually and cumulatively since the time of harvest. We used plot-based biometric measurements to estimate annual fluxes in the live and dead aboveground carbon pools from 2000 through 2006 in a tract of forest that was selectively logged in 2001. As a control, these measurements were compared to analogous biometric measurements done on an adjacent tract of forest in the footprint of an eddy flux covariance measurement site. We estimated coarse woody debris (CWD) respiration with a linear regression model to compare the dynamics of this pool on the two sites, and tracked the fate of harvested wood by species and timber grade to estimate the annual return of carbon to the atmosphere from removed sawtimber and firewood. Net annual carbon sequestration in on-site carbon pools was initially suppressed by the harvest but recovered to uptake rates similar to those observed on the control site by 2004. Cumulatively since the harvest, the site was a small net carbon sink. If carbon storage in wood products and fossil fuel offsets from firewood use are considered, the system is a larger sink but stored about half as much carbon as the control site over the same time interval. The future carbon dynamics of the harvest site will depend on the trajectories of uptake in tree growth and woody debris respiration as the stand matures.

1. Introduction

1.1 Greenhouse gases and the global carbon cycle

Carbon dioxide is one of several trace atmospheric gases that, through the greenhouse effect, help to make the earth a habitable environment by maintaining a comfortable range of surface temperatures. Solar energy enters the climate system as short wave radiation, mostly visible light, and a portion of this energy warms the land and oceans. These surfaces release the absorbed energy to the atmosphere as infrared radiation and sensible and latent heat. Greenhouse gas molecules absorb infrared radiation moving away from the earth's surface and emit it in all directions returning a portion of the radiated heat back to the earth. This is one component of the global energy budget (Figure 1.1). In the absence of anthropogenic influences, incoming and outgoing radiation are equal in this budget, which ensures climatic stability (Kiehl and Trenberth, 1997).

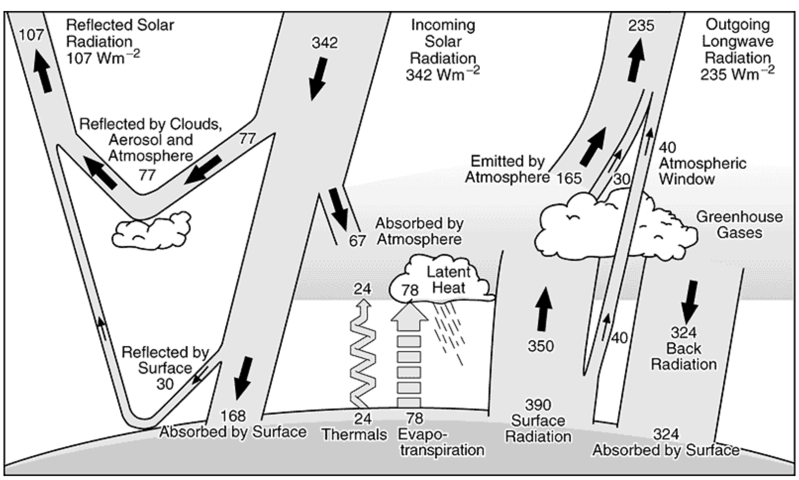

Figure 1.1 Mean annual global energy budget (Wm^{-2}) in the absence of anthropogenic influence. The balance between incoming and outgoing radiation in the global energy budget ensures a stable climate. Back radiation of energy released from the earth's surface by greenhouse gases is an essential part of the climate system, but can result in an imbalance between incoming and outgoing radiation if the concentration of greenhouse gases is altered by anthropogenic activity. Figure reproduced from Kiehl and Trenberth (1997).

Human activities have affected the global energy balance through the increase in greenhouse gas concentrations that has accompanied the rise of agriculture and industry. Scientists now agree with very high confidence (> 90% chance) that these anthropogenic influences are having a net warming effect. Evidence for this includes the fact that 11 of the 12 years with the warmest global surface temperatures in the instrumental record (since 1850) occurred between 1995 and 2006. Over the last 50 years, there has been a warming trend of 0.13 ± 0.03°C per decade. The oceans have absorbed over 80% of the excess heat in the climate system and sea level rise at a rate of 0.31 ± 0.07 m per century was observed between 1993 and 2003 due primarily to melting of the Greenland and Antarctic ice sheets and the thermal expansion of water (IPCC, 2007).

The net perturbation to the global energy balance attributed to emission of a greenhouse gas can be quantified as the anthropogenic radiative forcing for that gas. Five greenhouse gases, CO_2, CH_4, N_2O, CFC-11 and CFC-12, account for 97% of the total direct anthropogenic radiative forcing by long-lived gases (Hofmann *et al.,* 2006). Though the global warming potential (based on per mass radiative forcing and atmospheric lifetime) for CO_2 is much lower than for the other major greenhouse gases (Sihra *et al.,* 2001), the high rate of CO_2 emission makes it the largest and most rapidly increasing contributor to anthropogenic radiative forcing from long-lived gases, currently accounting for 62% of the total (Hofmann *et al.,* 2006).

Since the industrial revolution, atmospheric CO_2 concentrations have increased from an average of 278 ppm in 1750 to 379 ppm in 2005, largely due to the burning of fossil fuels and land-use change (IPCC, 2007). Though the rate of increase in atmospheric CO_2 is very likely (> 90% chance) to be unprecedented in the last 10,000 years (IPCC, 2007), it is much lower than the rate expected based on emissions. Fossil fuel burning released 6.3 ± 0.4 PgC/yr between 1990 and 1999, and land use change accounted for even more carbon flux to the atmosphere. However, the atmospheric CO_2 pool only increased by 3.2 ± 0.1 PgC/yr during the same period (IPCC, 2001). Of the "missing" carbon that did not stay in the atmosphere, about half was dissolved in the oceans (Sabine *et al.,* 2004), and half was taken up by the land biosphere. Atmospheric inversion models show that this terrestrial carbon sequestration is spatially variable (Gurney *et al.,* 2002).

North American temperate forests are considered to be a significant carbon sink based on multiple lines of evidence including atmospheric models (Fan *et al.,* 1998 and Pacala *et al.,* 2001) and empirical studies using both eddy-flux covariance data and biometric measurement of forest carbon pools (Curtis *et al.,* 2002). Several mechanisms for this net uptake have been suggested. Elevated atmospheric nitrogen deposition was cited as a possible reason for terrestrial carbon sequestration (Holland *et al.,* 1997). However, a more recent modeling study found that areas affected by nitrogen deposition are also areas of high tropospheric ozone concentrations, and the deleterious effects of this pollutant on plant health nearly offset the expected growth increase (Ollinger *et al.,* 2002).

Plant fertilization by increased CO_2 levels and forest regrowth following agricultural abandonment have also been suggested as drivers of increased uptake (Schimel *et al.,* 2000), though there has been continuing disagreement about which of these is the dominant factor. Some models suggest that CO_2 fertilization is more important (Houghton *et al.,* 1999; McGuire *et al.,* 2001; Jain and Yang, 2005) while other models (Schimel *et al.,* 2001; Hurtt *et al.,* 2002) and forest inventory analyses (Caspersen *et al.,* 2000; Pacala *et al.,* 2004) concluded that forest regrowth was dominant. Most recently, Albani *et al.* (2006) used the ecosystem demography model (Moorcroft *et al.*, 2001) to simulate carbon uptake rates in the eastern U.S. in the 1980s and 90s with and without CO_2 fertilization. They compared the results of these simulations with observed uptake rates from forest inventory analysis and eddy-flux covariance data and concluded that land-use history factors are most important and that forest harvesting would be a dominating influence on the future carbon dynamics of these forests.

1.2 Carbon accumulation in forests

The rate of carbon influx into or efflux from an ecosystem is defined as net ecosystem productivity (NEP) and is estimated as the difference between total photosynthetic gain or gross primary productivity (GPP) and ecosystem respiration (R) (Woodwell and Whittaker, 1968). A long-standing ecological hypothesis (Odum, 1969) states that as an ecosystem develops following a major disturbance, NEP peaks and declines to zero as the system reaches maturity (Figure 1.2). There is some empirical evidence for this hypothesis in forest ecosystems from studies of aboveground net primary productivity (NPP), the net addition to aboveground biomass of live plants, in forest age chronosequences. A meta-analysis of these studies found that aboveground NPP differed by as much as 76% between developing and mature stands with an average difference of 34% (Gower *et al.,* 1996). The decreased aboveground NPP in older stands may be due in part to changes in allocation, with belowground growth increasing as the soil nutrient supply declines in a maturing stand (Ryan *et al.,* 2004).

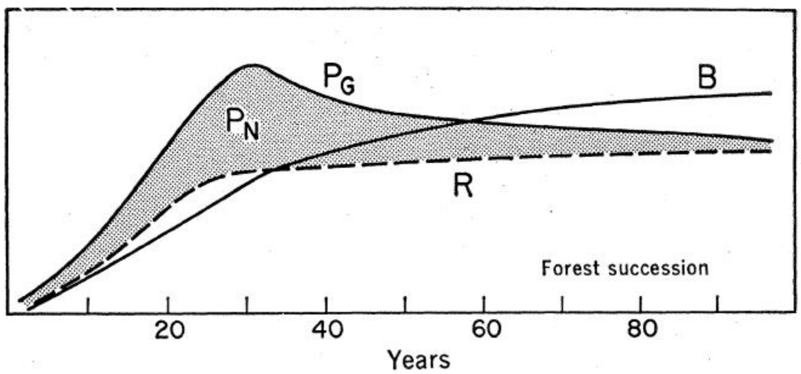

Figure 1.2 The energetics of forest succession. Expected pattern of gross primary productivity (P_G), respiration (R), net ecosystem productivity (shaded area, P_N), and biomass accumulation (B) during forest development following a large disturbance. Figure reproduced from Odum (1969).

A long-term study of carbon exchange in a 75-110 year old mixed hardwood forest at the Harvard Forest Environmental Measurement Site (HFEMS) in north central Massachusetts suggests that NEP is positive and increasing in the temperate forests of New England (Wofsy *et al.,* 1993; Barford *et al.,* 2001; Urbanski *et al.,* in press). Since 1990, the eddy-flux covariance method (Baldocchi *et al.,* 1988) has been used at this site to measure the net ecosystem exchange (NEE) of CO_2 between the forest and the atmosphere. Simultaneous ground-based measurements of forest carbon pools have also taken place at the HFEMS. These measurements are used to relate the overall NEE observed with eddy-flux covariance to fluxes in forest carbon pools such as live biomass and woody debris (Barford *et al.,* 2001). These two methods of measurement indicated that carbon uptake at the HFEMS averaged 2.5 MgC ha^{-1} yr^{-1} between 1992 and 2004, and uptake increased in magnitude by 0.15 MgC ha^{-1} yr^{-1} (Urbanski *et al.*, in press). Primary productivity at this site is dominated by red oak, and a study of the growth rate of this species in stands of varied age across New England, including the HFEMS as one of the mid-successional stands in the study, found that red oak productivity tended to increase with stand age (Pederson, 2005). This suggests that the carbon dynamics of these red oak forests, where uptake rates are increasing over time, are similar to those predicted by Odum's model for an early-successional forest. Eddy-flux measurements in a 200-year-old hemlock stand at the Harvard Forest in 2001 found that it was a carbon sink of slightly lower magnitude than the mixed hardwood stand in that year (Hadley and Schedlbauer, 2002), so it is uncertain at what age New England forests will reach a point where NEP is close to zero.

1.3 Forest harvesting and carbon sequestration

It has been suggested that occasional harvesting may increase the potential of a forest to store carbon by returning mature stands to an earlier successional stage when hypothetical NEP is higher (Roxburgh *et al.,* 2006; Laclau, 2003). In cases where GPP declines with forest age, nutrient limitation in a closed-canopy stand and decreased hydraulic conductance with increased tree height appear to be the major causes of the decline in GPP, while decreased growth due to the aging of trees is not considered to be a factor. This makes it possible to increase the GPP of a stand by thinning it (Martinez-Vilalta *et al.,* 2007).

While it is possible that GPP is higher in recently harvested forests, the resulting carbon uptake may be offset by fluxes to the atmosphere that result from the disturbances associated with harvesting, so leaving mature forests undisturbed may be the best strategy for maximizing carbon storage. Modeling studies estimated that logging of old growth forests would result in large fluxes of CO_2 to the atmosphere (Harmon *et al.,* 1990; Song and Woodcock, 2003). An analysis of the carbon storage potential of Alaska's Tongass National Forest under different future management regimes predicted that cessation of logging would maximize carbon sequestration over the next century (Leighty *et al.,* 2006).

Existing empirical data on harvesting and carbon sequestration are limited to chronosequence studies in boreal (Howard *et al.,* 2004; Martin *et al.,* 2005) and tropical forests (Lasco *et al.,* 2006). Among these studies, there is disagreement on the effect of harvesting. In a study of boreal mixedwood and jack pine logging chronosequences in Manitoba, NPP differed by 24% between the oldest and youngest stand and was greatest in a stand harvested 11 years before the study (Martin *et al.,* 2005). However, on a group of similar sites in Saskatchewan, the most recently harvested site was a significant carbon source while the other sites were slight sinks or sources (Howard *et al.,* 2004). A carbon accounting study of a logging chronosequence and sawmill in an area of South Asian forest that is logged on a 35-year cutting rotation found that stands only reached 70% of pre-harvest carbon stocks before they were harvested again and in each harvest about 60% of the carbon in aboveground biomass was released to the atmosphere. The authors concluded that the forest is not an efficient and sustainable carbon sink because the current harvesting rotation is too short (Lasco *et al.,* 2006). More conclusive data on harvesting and carbon dynamics are needed before forest management plans can be made with the goal of preserving or increasing carbon sequestration in temperate forests, particularly data that tracks the source-sink dynamics of a single site following a harvest.

1.4 Carbon sequestration in wood products

In addition to measuring the on-site carbon fluxes following a harvest, the fate of carbon in the removed wood must be determined to understand the overall biosphere-atmosphere carbon flux resulting from a harvest. Some carbon is stored in long-lived products such as furniture and construction material and returned to the atmosphere slowly through decay in landfills, while carbon in fuelwood or short-lived paper products is returned to the atmosphere more quickly (Skog and Nicholson, 1998; Marland and Marland, 2003). Though a significant portion of the carbon in harvested wood is returned to the atmosphere, the use of some wood products may offset fossil fuel emissions. Wood burned for energy, as firewood in homes or as sawmill residues, which may be burned at the sawmill or sold as fuel, offsets fossil fuel burning, though there are many energy requirements in the harvesting and processing of wood that must also be considered (Gan and Smith, 2006; Raymer, 2006). Using wood in construction may reduce carbon fluxes to the atmosphere by replacing cement, steel, brick and other materials that require greater fossil fuel consumption in production (Werner *et al.,* 2005; Gustavsson and Sathre, 2006).

Forest carbon pools and wood product carbon pools were considered together in a modeling study that found that the source-sink dynamics of the system as a whole were highly dependent on the length of the harvesting rotation, with longer rotations promoting carbon sequestration (Perez-Garcia, 2005). The results were also highly dependent on the amount of detail used in accounting for things like the fuel substitution of wood burned for energy and the decay rate of wood in long-term storage pools. Empirical data considering both forest pools and wood products are needed for comparison with model results.

1.5 Forest harvesting in New England

The second-growth, temperate forests of New England are an ideal setting for a study of the effects of harvesting on forest carbon dynamics, because carbon exchange in forests not subjected to recent harvesting has been studied extensively at the Harvard Forest and because logging is fairly common. The Harvard Forest is part of the heavily forested North Quabbin region, a 168,000 ha landscape that includes the Quabbin reservoir, which supplies water to metropolitan Boston. Between 1984 and 2000, 26% of the region's forests were harvested at least once with the average harvest removing about one fourth of the stand volume. Each year approximately 1.5% (2,000 ha) of the forested area is affected by new harvesting disturbance, with most harvesting occurring in small operations on parcels of land controlled by over 2,500 non-industrial private forest owners in the region (Kittredge *et al.,* 2003). Comparison of harvesting activity in the North Quabbin with databases for Massachusetts (Dickson and McAfee, 1988) and the Northeastern U.S. (Birch, 1996) shows that the region is broadly representative of forest management across Massachusetts and the Northeast (Kittredge *et al.,* 2003).

In an effort to synthesize the management plans of the many government, non-profit, and private landowners in the North Quabbin and across Massachusetts, Foster *et al.* (2005) published *Wildlands and Woodlands*, a large-scale management proposal for the forests of Massachusetts. In this report, the authors identified the goals for forest management in the Northeast: a sustainable source of wood products, maintenance of an assemblage of habitats that promotes biodiversity, areas for recreational and educational experiences, and preservation of ecosystem processes that provide clean air and water, productive soils, and natural flood control. The report does not specifically identify carbon sequestration as a goal for management, perhaps due to the lack of data on the effect that management decisions may have on this aspect of ecosystem functioning.

Incentives for land owners to make management decisions that increase carbon sequestration could be incorporated into cap-and-trade programs for carbon emissions, which are emerging as a strategy for dealing with greenhouse gases in New England and other parts of the country. Under this type of program, a maximum level of emissions is set for a region and emissions allowances are traded on the free market, making emissions-reducing measures a potentially lower-cost option. Cap-and-trade programs are a good option for controlling greenhouse gases because these pollutants have the same effect regardless of where they are released and the cost of reducing emissions is highly variable among different sources (Ellerman *et al.,* 2003). Massachusetts recently joined the Regional Greenhouse Gas Initiative (RGGI), a cap-and-trade program for CO_2 emissions by power plants in nine Northeast and Mid-Atlantic states. Under the current RGGI memorandum of understanding, a source may cover up to 3.3% of its carbon emissions with carbon offsets such as landfill gas capture and combustion and afforestation of cleared land, and the signatory states have agreed to develop other offset categories including certain forestry practices (RGGI, 2005). Before forest management could be included in a program like this, more data is needed on the consequences of harvesting for carbon sequestration.

1.6 Study objectives

This study examines the carbon dynamics of a second-growth hardwood forest in north-central Massachusetts that was selectively logged in 2001 using biometric measurement of forest carbon pools and life-cycle analysis of removed saw timber and firewood. Both the on-site and overall source-sink dynamics are analyzed on an annual basis and cumulatively since the harvest by constructing a carbon budget that includes measurements and estimates of forest carbon pools, carbon storage and fluxes from wood products, fossil fuel emission offsets from the burning of wood for energy, and energy use in harvesting, transport and processing of wood products. The consequences of the results for forest management in New England and the consideration of global-scale consequences in local and regional management plans are addressed.

2. Methods

2.1 Study site

The harvest and control sites are located in the town of Petersham in north-central Massachusetts (42.5 N, 72.2 W). The area is nearly continuously covered by mixed hardwood forest. Dominant species are northern red oak (*Quercus rubra*) and red maple (*Acer rubrum*) with some eastern hemlock (*Tsuga canadensis*), American beech (*Fagus grandifolia*), eastern white pine (*Pinus strobus*), and several species of birch (*Betula spp.*). Most forests in the region established after intense, land-use related disturbances ceased. In the mid-1800's, 70% of the land was cleared for pasture and 17% was tilled. The remaining forests were used as woodlots. Throughout the late 19th and early 20th centuries, farms were gradually abandoned and reforested. Large areas of this second-growth forest were logged and some clear cut areas were converted to conifer plantations (Foster, 1992). Many areas were disturbed by a major hurricane followed by a salvage harvesting operation in 1938 (Foster *et al.,* 1997).

The control site is part of the Prospect Hill research tract of the Harvard Forest. An analysis of tree ring data and historical records found that most of the canopy trees on the site established following large timber harvests between 1890 and 1895 and forest management that occurred from 1925 to 1934. The stand only suffered slight damage in the 1938 hurricane and this disturbance does not appear to be an important factor in the establishment of the current forest (Pederson, 2005). The site encompasses the fetch of an eddy-flux covariance tower, installed in 1989 (Wofsy *et al.,* 1993). To complement the CO_2 exchange data from the tower with ground-based measurements, 40 10 m radius circular plots were established at random locations within 100 m segments of 8 500 m transects extending away from the tower to the northwest and southwest, the direction of most prevailing wind (Figure 2.1). The plots were established in 1993, and annual biometric measurement of forest carbon pools has taken place on them since 1998 (Barford *et al.,* 2001). In 2001, 3 plots on the northwest edge of the site were flooded and excluded from further measurements. Three of the original 40 plots lie on the harvest site. The remaining 33 plots constitute the control site for this study.

The harvest site is located on the privately-owned Simmes Trust land, adjacent to the control site. Before the harvest, the species composition on this site and the control site were similar. The basal area of oak was not significantly different ($P \geq 0.05$), at 12.4 m^2/ha on the harvest site and 12.0 m^2/ha on the control site, and oak was the most prevalent species on both sites. There was also no significant difference in the prevalence of hemlock (4.0 m^2/ha on the harvest site and 5.9 m^2/ha on the control site). There were significant differences in the prevalence of maple (3.4 m^2/ha on the harvest site and 8.3 m^2/ha on the control site) and beech (3.2 m^2/ha on the harvest site and 0.19 m^2/ha on the control site).

In addition to the three plots already established on this site, six plots were added in 2000, one year before the harvest, to better capture the effects of the harvest on the carbon dynamics of the forest. One of these plots was later abandoned because it was unaffected by the harvest. The remaining eight plots were expanded to a radius of 15 m to increase the area surveyed. Comparison of 2001 data from these expanded plots with data based only on the inner 10 m radius plots showed that the expansion did not introduce any significant bias (Figure 2.2).

Figure 2.1 Map of control site and harvest site plots. The area between Pierce road and the dashed line represents the planned harvest area.

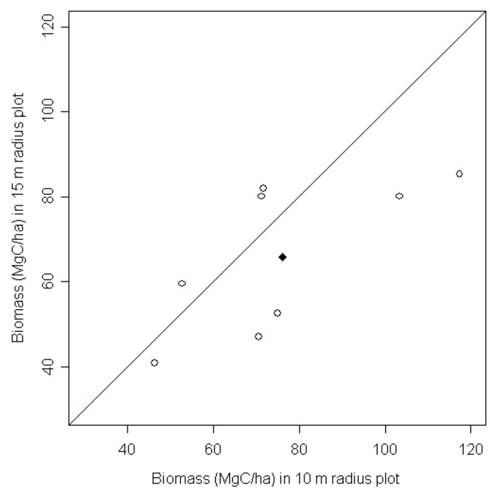

Figure 2.2 Comparison of biomass (MgC/ha) measurements in 2001 based on 10 m radius plots and expanded 15 m radius plots. Solid point indicates the mean biomass for the eight plots. The one to one line is shown. Figure modified from Curry (2002).

The harvest was carried out in the winter of 2001. The volume of wood removed averaged 42.8 m^3/ha, about 26% of the standing biomass. Of this, 36% was saw timber and 64% was firewood (Curry, 2002). The volume removed was typical of harvests in the North Quabbin region, which remove 44.7 m^3/ha on average (Kittredge *et al.,* 2003), though a larger than average proportion of the wood from this harvest was sold as firewood (Curry, 2002).

2.2 Carbon budget

An annual carbon budget for the harvest site from 2000 through 2006 was constructed using data from measurements of the major carbon pools. Biometric measurement of aboveground live and dead carbon pools was conducted, and where direct measurements were not possible, fluxes were inferred from models and other studies. Harvested wood was also accounted for in the carbon budget. The fluxes in these pools were summed to examine the flux of carbon to the atmosphere on an annual basis and cumulatively since the harvest.

2.3 Live biomass measurements

In 1993, the diameter at breast height (DBH) of all trees in the control site plots ≥ 10 cm DBH was recorded. These trees were measured again in 1998 and fitted with stainless steel, spring-mounted dendrometer bands, which expand as the tree grows and allow for precise growth measurements. In 2000 before the harvest, the same was done for all trees ≥ 5 cm DBH in the harvest site plots. Smaller trees were included in the harvest site measurements to capture a potentially different response of smaller trees to the harvest. Every year since the dendrometers were installed, their expansion was measured using calipers several times during the growing season. The expansion of the dendrometers was converted to a change in DBH. From DBH, the woody biomass of each tree was calculated using species specific allometric equations (Appendix A, Telfer, 1969; Grigal and Ohmann, 1977; Brenneman *et al.,* 1978; Young *et al.,* 1980; Jenkins *et al.,* 2003), and the carbon content was estimated as half of the woody biomass (Fahey *et al.,* 2005). The plots were surveyed for trees that had grown into the measured size classes in 1999 on the control site and in 2003, 2004, and 2006 on both sites. The plots were surveyed annually for trees that had died, determined by the absence of live foliage at the peak of the growing season, and these trees were removed from the live tree survey. These measurements were used to calculate the annual change in woody biomass of living trees for the harvest site and control site due to growth, recruitment, and mortality. Interannual and between-site comparisons were made using bootstrapped 95% confidence intervals (Efron and Tibhshirani, 1997).

2.4 Woody debris survey

Coarse woody debris (CWD) was surveyed in all plots on the harvest site in 1999, 2001 (post-harvest), 2003 and 2006, and 27 of 33 control plots in 2000, 15 of 33 plots in 2003 and all 33 plots in 2006. The plots were surveyed for pieces of CWD within the plot boundary ≥ 7.5 cm in diameter at the larger end. These pieces were classified by type (log, log snag, whole tree snag, or stump, Appendix B.1). Logs ≤ 1 m in length were excluded but stumps ≤ 1 m in height were not. Logs, log snags, and stumps were measured so that their volumes could be calculated using equations from Harmon and Sexton (1996, Appendix B.2). The DBH of whole tree snags was measured and used to calculate their biomass using the same allometric equations used for live trees. Each piece was also assigned to a decay class from 1 to 5, with 1 being the least decayed, using criteria from Harmon and Sexton (Appendix B.3). Decay class specific densities calculated for the Harvard Forest (Appendix B.4, Liu *et al.,* 2006) were used to convert volume to biomass for logs, log snags and stumps, and biomass to volume for whole tree snags.

Fine woody debris (FWD, 2-7.5 cm in diameter) was surveyed on 10 m segments of randomly placed line transects using the line-intercept method (Van Wagner 1968; Brown 1974). The line-intercept measurements were extrapolated to volume per area using an equation from Harmon and Sexton (1996, Appendix B.2) and volume was converted to biomass using decay-class specific densities from Liu *et al.* (2006). In 2001, 26 segments were surveyed on each site. In 2003, 40 segments were surveyed on the control site and 47 segments were surveyed on the harvest site. In 2006, 30 segments were surveyed on the control site and 40 segments were surveyed on the harvest site.

2.5 Coarse woody debris respiration model

Wood temperature and moisture content are important variables in determining the rate of carbon loss from woody debris due to respiration (Liu *et al.,* 2006). Micrometeorological data collected at both the harvest and control sites in 2004 and 2005 showed between-site differences in log temperature, log moisture, and the diurnal range of log temperatures (Table 2.1 and Figure 2.3, Curran, 2005). To account for site differences in CWD pool dynamics due to differences in site conditions, a model for hourly carbon flux from CWD to the atmosphere was developed for the periods between CWD surveys. The CWD respiration model uses air temperature and decay class as proxies for wood temperature and moisture to predict log transformed respiration rates (r^2 = 0.32, Table 2.2). In this model, decay classes 4 and 5 were combined into decay class V because the sample size in these classes was low, and decay classes 1 and 2 were combined into decay class I because there was no significant difference in wood density and moisture content for these classes (Liu *et al.,* 2006). Respiration rates for downed CWD (logs) were determined using the following equation:

$$\ln R = \text{Intercept} + T_a + D_{III} + D_V \qquad (1)$$

Where R is respiration (µg C g^{-1} C s^{-1}), T_a is air temperature (K), and D_{III} and D_V are categorical parameters for decay classes III and V. Respiration rates for standing CWD (snags and stumps) were estimated as 40% of downed CWD respiration rates (Erickson *et al.,* 1985; Mattson *et al.,* 1987).

On-site air temperature was measured at 30-minute intervals in 2004 and 2005 at one harvest site plot and one control site plot. Measurements were made with YSI (Yellow Springs, OH) 44032 Precision Thermistors housed inside Met One (Grant Pass

a)

Measurement	Mean	95% Confidence Int.		Std. Dev.	n
Air Temperature (°C)	18.57	18.41	18.72	3.96	2522
Relative Humidity (%)	83.85	83.33	84.38	13.57	2521
Log Temperature Decay I (°C)	18.38	18.29	18.47	2.84	4184
Log Temperature Decay III (°C)	18.99	18.89	19.09	3.28	4184
Log Temperature Decay V (°C)	18.38	18.3	18.47	2.85	4184
Log Moisture Decay I (VWC)	0.54	0.51	0.57	0.05	12
Log Moisture Decay III (VWC)	0.46	0.44	0.48	0.03	12
Log Moisture Decay V (VWC)	0.80	0.80	0.80	0.00	12

b)

Measurement	Mean	95% Confidence Int.		Std. Dev.	n
Air Temperature (°C)	17.97	17.84	18.10	3.81	3151
Relative Humidity (%)	89.03	88.65	89.40	10.73	3126
Log Temperature Decay I (°C)	17.50	17.42	17.59	2.90	4236
Log Temperature Decay III (°C)	17.20	17.13	17.28	2.44	4236
Log Temperature Decay V (°C)	17.17	17.09	17.25	2.60	4236
Log Moisture Decay I (VWC)	0.40	0.39	0.41	0.02	12
Log Moisture Decay III (VWC)	0.51	0.49	0.54	0.04	12
Log Moisture Decay V (VWC)	0.80	0.80	0.80	0.00	12

c)

Measurement	Mean	95% Confidence Int.		Std. Dev.	n	p
Air Temperature (°C)	0.44	0.42	0.47	41.55	2499	< 0.01
Relative Humidity (%)	-4.51	-4.68	-4.35	-55.1	2417	< 0.01
Log Temperature Decay I (°C)	0.76	0.75	0.77	110.74	4006	< 0.01
Log Temperature Decay III (°C)	1.69	1.65	1.74	77.74	4006	< 0.01
Log Temperature Decay V (°C)	1.1	1.08	1.12	116.12	4006	< 0.01
Log Moisture Decay I (VWC)	0.14	0.12	0.17	12.6	11	< 0.01
Log Moisture Decay III (VWC)	-0.05	-0.07	-0.04	-9.48	11	< 0.01
Log Moisture Decay V (VWC)	0	0	0	--	--	--

Table 2.1 Summary statistics for select micrometeorological measurements from July through September, 2004, at the (a) harvest site and (b) control site, and (c) the results of t-tests on the differences (harvest – control) between the two sites (Curran, 2005).

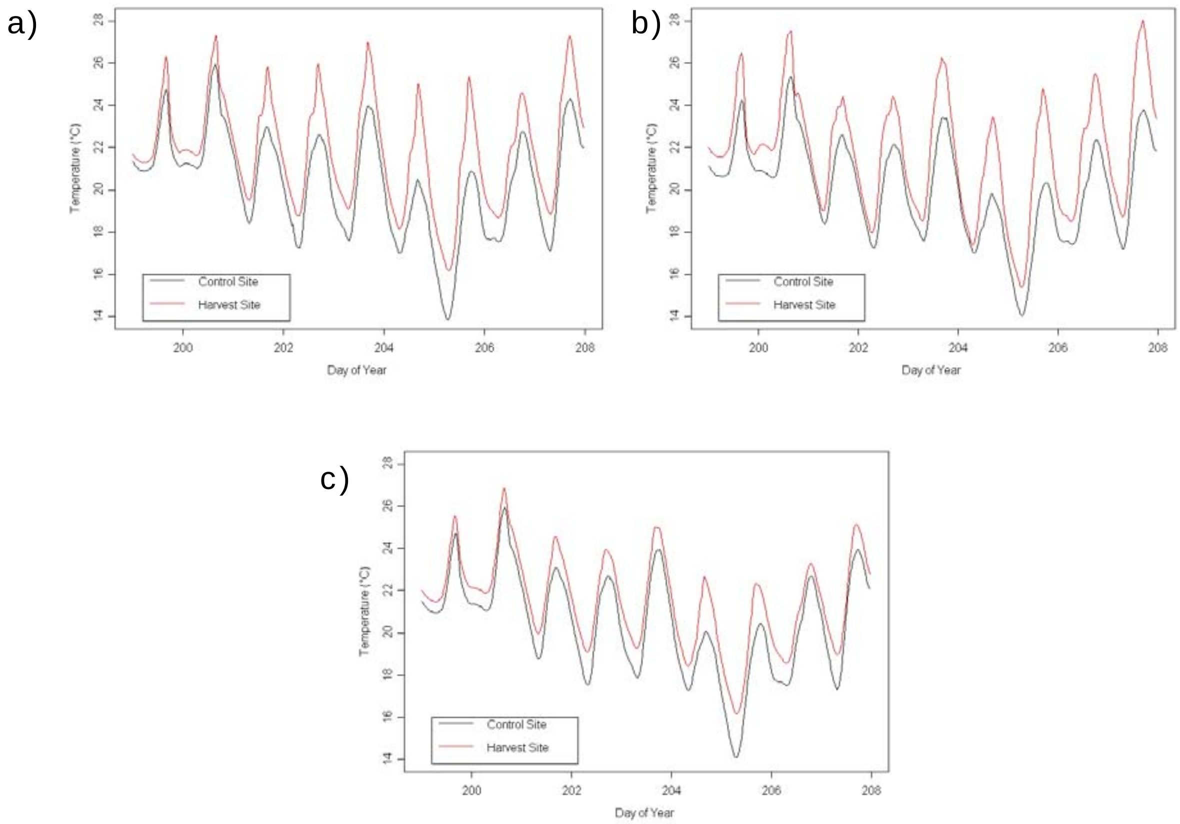

Figure 2.3 Comparison of log temperature over an eight day period in 2004 at the harvest and control sites for (a) decay class I, (b) decay class III, and (c) decay class V downed CWD (Curran, 2005).

	Regression Coefficient	**Standard Error**
Intercept	-28.672	0.066
T_a	0.078	0.003
D_{III}	0.422	0.058
D_V	0.976	0.059

Table 2.2 Linear regression model predicting log transformed woody debris respiration rates (μg C g^{-1} C s^{-1}). T_a is air temperature (K). $r^2 = 0.32$. D_{III} and D_V are categorical parameters for decay class III and decay class V woody debris, respectively.

OR) 076B-4 Aspirated Temperature Shields mounted 1.5 m off the ground at two points 10-15 m apart in each plot (Curran, 2005).

Micrometerological data from 2005 were used to create a linear regression model that predicts hourly on-site air temperature from 2001 through 2006 using hourly air temperature and incoming solar radiation measurements from the Fisher Meteorological

Station (harvest site $r^2 = 0.953$, control site $r^2 = 0.987$, Table 2.3). Because Fisher Station data was not available before 2001, a second linear regression model was made using micrometeorological data from 2004 to predict hourly control-site air temperature using hourly air temperature and photosynthetically active radiation (PAR) measurements from the HFEMS ($r^2 = 0.971$, Table 2.4). This model was applied to both sites for 1999 and 2000, before the harvest site was logged. Modeled air temperatures that were outside the range of air temperature values used in making the respiration model (262.85 – 305.75 K) were replaced with the maximum and minimum temperatures used in the model.

Control Site	**Regression Coefficient**	**Standard Error**
Intercept	8.58	0.431
T_a	0.969	0.00151
ISR	-0.00264	5.73E-05

Harvest Site	**Regression Coefficient**	**Standard Error**
Intercept	42.96	1.26
T_a	0.853	0.00431
ISR	-0.000940	6.20E-05

Table 2.3 Linear regression model predicting hourly on-site air temperature for the control and harvest sites from hourly air temperature and solar radiation measured at the Fisher Meteorological Station. Control site $r^2 = 0.987$, harvest site $r^2 = 0.953$. T_a is air temperature (K). *ISR* is incoming solar radiation (W m^{-2}).

	Regression Coefficient	**Standard Error**
Intercept	25.67	0.812
T_a	0.908	0.00281
PAR	0.000392	2.48E-05

Table 2.4 Linear regression model predicting hourly on-site air temperature for the control site for 1999 and 2000 from hourly air temperature and photosynthetically active radiation (PAR) measured at the Harvard Forest Environmental Measurement Site. $r^2 = 0.971$. T_a is air temperature (K). *PAR* is photosynthetically active radiation (μmol m^{-2} s^{-1}). This model was applied to both sites for the two years before the harvest.

Hourly respiration rates were calculated from modeled hourly air temperature for standing and downed CWD in each decay class on the two sites. These respiration rates were applied to data from the CWD surveys to calculate the change in the sizes of the pools due to respiration using the following equation:

$$p_n = p_0 (1 - R_1)(1 - R_2) \ldots (1 - R_n) \quad (2)$$

Where p_n is the pool size (MgC/ha) after n hours, p_0 is the initial pool size for the model interval, and $R_1, R_2, \ldots R_n$ are the hourly respiration rates (converted to g C g^{-1} C hr^{-1}).

At the start of model intervals which begin in the summer of CWD survey years, the pool sizes were forced to the measured values from the corresponding survey. Four times a year, one fourth of the mortality estimate (MgC/ha) from the following summer's mortality survey was added to the decay class I standing pool, because the mortality estimate includes trees that died episodically between the previous and current summer surveys.

To estimate the rate at which wood shifts to more decayed pools during the time between two surveys, logs that were present in both surveys were identified and the probabilities that they had shifted from class I to class III, from class I to class V, or from class III to class V were calculated. The sample size of standing pieces that could be identified in multiple surveys was too small to calculate these probabilities separately for standing wood, so the shift probabilities for standing wood were estimated as 40% of the probabilities for downed wood, based on the assumed difference in respiration rates. The compound interest formula and between-survey shift probabilities were used to simulate an annual shift of wood to more decayed pools as follows:

$$P = C (1 + r)^t \quad (3)$$

Where, as applied to the pool shifts, C is the size of the less decayed pool in the earlier survey, P is the quantity of wood that shifted from the less decayed pool to the more decayed pool between surveys, r is the annual shift rate, and t is the time in years between surveys. The shift probabilities can be expressed in these terms:

$$S = 1 - P/C \qquad (4)$$

Where S is the calculated probability of wood shifting from one decay class to another. The compound interest formula can then be solved for the annual shift rate:

$$r = (1 - S)^{1/t} - 1 \qquad (5)$$

At the beginning of time intervals starting in the summer of non-survey years, the annual shift rates were used to calculate the size of each pool shift using the following equation:

$$s = p - rp \qquad (6)$$

Where s is the amount of carbon shifted, r is the shift rate, and p is the size of the less decayed pool. The shifts were subtracted from the less decayed pools and added to the more decayed pools.

To estimate the rate at which wood shifted from standing to downed pools between two surveys, the number of snags that were present in both the earlier and later survey was determined and then used calculate the probability that a snag fell between the two surveys. The compound interest formula was used again to convert this probability to an annual shift rate. At the beginning of time steps starting in the summer of non-survey years, the amount of carbon in falling snags of each decay class was calculated using this rate. When a snag falls, it forms one or more logs, which become part of the downed pool, and a stump that stays in the standing pool. To account for this, a log to stump ratio was calculated as follows:

$$LS = (V_{\text{snag}} - V_{\text{stump}}) / V_{\text{snag}} \qquad (7)$$

Where LS is the log to stump ratio, V_{snag} is the average volume of a log snag, and V_{stump} is the average volume of a stump. This ratio was based only on control site data because some stumps on the harvest site were created by cutting and may be different than stumps created by trees falling due to natural causes. The standing to down shifts were multiplied by the log to stump ratio before being subtracted from the standing decay class pools and added to the corresponding downed decay class pools.

Pool dynamics were modeled from the summer of 2000 to the summer of 2006 on the control site and the following equation was used to calculate the annual mass loss due to respiration:

$$m_i = p_i - p_{i\text{-}1} + M_i \qquad (8)$$

Where m_i is the mass lost due to respiration in year i, p_i is the total size of the CWD pools at the end of the last time step in year i, and M_i is the total addition from mortality in year i. If the model is closely representing natural processes, the modeled pool sizes for the summer of a survey year will be slightly larger than the measured pool sizes. The difference accounts for the total mass lost to fragmentation and leaching between two surveys.

Pool dynamics were modeled in the same way for the harvest site from the summer of 1999 to the summer of 2000 and from the summer of 2001 to the summer of 2006. The harvest was treated as an instantaneous event occurring on January 1, 2001, and the time between the summer of 2000 and the summer of 2001 was divided into a 153 day "pre-harvest" period before January 1, 2001, and a 212 day "post-harvest" period after this date. To find the rate of pool shifts and standing to downed shifts for these shortened time intervals, equation (5) was modified as follows:

$$r = (1 - S)^{d/\tau} - 1 \qquad (9)$$

Where r is the shift rate for the time interval, S is the shift probability, d is the length of the shortened interval in days, and τ is the time in days between the two surveys used to calculate S.

Inputs from mortality could not be determined for the harvest year because the mortality survey included trees removed by the loggers. For mortality input dates in the pre-harvest time period, the value for quarterly mortality inputs in 2000 was substituted, and the mortality input value from 2002 was used for post-harvest inputs. Pre-harvest pool dynamics were modeled using these shift rate and mortality input adjustments.

The harvest could not be simulated in the model, so modifications were made to back calculate mass loss from respiration between the CWD survey in the summer of 2001 and the harvest. Starting with the 2001 measured pool sizes, the pool and standing to down shift routines were run backwards by solving the system of equations given below, which accounts for standing to down shifts being applied before pool shifts in the regular model:

$$
\begin{aligned}
{}_{S}I_i &= {}_{S}I_{i\text{-}1} - {}_{S}I_{i\text{-}1}\mu - {}_{S}I_{i\text{-}1}(1-\mu)\,\sigma_{13} - {}_{S}I_{i\text{-}1}\,(1-\mu)\,\sigma_{15} \\
{}_{S}III_i &= {}_{S}III_{i\text{-}1} - {}_{S}III_{i\text{-}1}\,\mu - {}_{S}III_{i\text{-}1}(1-\mu)\,\sigma_{35} + {}_{S}I_{i\text{-}1}(1-\mu)\,\sigma_{13} \\
{}_{S}V_i &= {}_{S}V_{i\text{-}1} - {}_{S}V_{i\text{-}1}\,\mu + {}_{S}I_{i\text{-}1}\,(1-\mu)\,\sigma_{15} + {}_{S}III_{i\text{-}1}(1-\mu)\,\sigma_{35} \\
{}_{D}I_i &= {}_{D}I_{i\text{-}1} + {}_{S}I_{i\text{-}1}\,\mu - ({}_{D}I_{i\text{-}1} + {}_{S}I_{i\text{-}1}\,\mu)\,\delta_{13} - ({}_{D}I_{i\text{-}1} + {}_{S}I_{i\text{-}1}\,\mu)\,\delta_{15} \\
{}_{D}III_i &= {}_{D}III_{i\text{-}1} + {}_{S}III_{i\text{-}1}\,\mu - ({}_{D}III_{i\text{-}1} + {}_{S}III_{i\text{-}1}\,\mu)\,\delta_{35} + ({}_{D}I_{i\text{-}1} + {}_{S}I_{i\text{-}1}\,\mu)\,\delta_{13} \\
{}_{D}V_i &= {}_{D}V_{i\text{-}1} + {}_{S}V_{i\text{-}1}\,\mu + ({}_{D}I_{i\text{-}1} + {}_{S}I_{i\text{-}1}\,\mu)\,\delta_{15} + ({}_{D}III_{i\text{-}1} + {}_{S}III_{i\text{-}1}\,\mu)\,\delta_{35} \qquad (13)
\end{aligned}
$$

Where the measured values for decay class I, III, and V standing and downed pools, are given by ${}_{S}I_i$, ${}_{S}III_i$, ${}_{S}V_i$, ${}_{D}I_i$, ${}_{D}III_i$, and ${}_{D}V_i$ respectively, μ is the standing to downed shift rate, σ_{ab} is the shift rate from decay class a to decay class b for standing CWD, and δ_{ab} is the shift rate from decay class *a* to decay class *b* for downed CWD. The variables subscripted $i - 1$ are the pool sizes for which the system was solved.

The change in the pools for each time step due to respiration was then calculated using a variation on equation (4):

$$p_0 = p_n / (1 - R_1)(1 - R_2) \ldots (1 - R_n) \quad (11)$$

Where p_n is the pool size at the end of the interval (which is the starting value in this case), p_0 is the pool size at the beginning of the interval, and R_1, R_2,…R_n are the hourly respiration rates for the interval. The 2002 quarterly mortality value was subtracted from the decay class I standing pool at points where mortality inputs would normally occur.

To estimate the error in the calculated pool sizes, a simulation of 1000 model runs was developed. For each of the respiration regression parameters, a normal distribution was created using the corresponding mean and standard error of each parameter. These distributions were sampled 1000 times, then run in the model to produce a distribution of pool sizes. The 95% confidence intervals of the respiration model were determined from these pool size distributions. The same was done for the parameters in each of the temperature regressions to determine the error due to this part of the model.

A model was not developed for FWD respiration, so annual FWD flux in the carbon budgets is estimated as the change in biomass between surveys averaged over the time between surveys. The first FWD measurement on both sites was in 2001. The 2001 harvest site FWD flux is estimated as the difference between the 2001 FWD measurement (post-harvest) and the pre-harvest CWD measurement multiplied by the average ratio of FWD to CWD on the control site (~25%). The 2001-2003 annual FWD flux for the control site is used for the 2001 and 2000 flux in the control site carbon budget and the 2000 FWD flux in the harvest site budget.

2.6 Belowground biomass and soil carbon fluxes

Fluxes in belowground biomass were estimated as 20% of fluxes in aboveground biomass, as was done in previous carbon accounting on the control site (Barford *et al.,* 2001). This assumption is based on root excavation and measurement of a second-growth forest in New Hampshire (Whittaker *et al.,* 1974). To test the validity of this assumption in the case of the harvested site, where stand composition has been altered, the ratio of belowground to aboveground fluxes was calculated for this site using allometric equations for belowground biomass (Jenkins *et al.,* 2003). Soil carbon uptake was estimated as 0.2 (± 0.1) Mg/ha yr based on a study of ^{14}C residence time in soils at the control site by Gaudinski *et al.* (2000).

2.7 Wood products accounting

Harvested wood was accounted for as either saw timber or firewood. The sawmill that processed the wood from the harvest (Heyes Forest Products, Orange, MA) estimated the merchantable volume of saw timber logs removed from the harvest site using an international ¼" scaling rule and classified the higher quality saw timber by species and grade, one (lowest quality) through four (highest quality). The lower quality saw timber was designated for use in making industrial pallets or to be processed as pulp. The carbon emissions resulting from fossil fuel use in harvesting and transport to the sawmill were estimated using values from Raymer (2006) (all conversion factors in Appendix

C.1). The sawmill reported a 3% loss of saw timber biomass as sawdust and other sawmill residues (Curry, 2002). Since most mill residues are burned for energy in the place of oil, their heat content (Kuhns and Schmidt, 2003) and the carbon emissions of an equivalent amount of oil (Aubé, 2001) were calculated. This value was subtracted from the calculated carbon emissions for production of end use products from the saw timber (Werner *et al.,* 2005).

Descriptions of the end-use of saw timber by species and grade from Curry (2002) were used to assign the wood products to end use categories given by Skog and Nicholson (1998), for which they calculated the median life in use for the carbon in these products. Equations from the HARVCARB model (Appendix C.2) developed by Row and Phelps (1996) were used to determine the annual retirement of carbon from these pools, accounting for recycling rates of paper and wood (EPA, 2005). Once transferred out of use, the wood products were treated as municipal solid waste (MSW), a portion of which is incinerated to produce energy while the rest is disposed of in a landfill (EPA, 2005). The annual landfill efflux of CO_2 and CH_4 due to the disposed wood products was estimated using the LandGEM v.3.02 model (Alexander *et al.,* 2005). Statistics for the Martone Landfill in Barre, MA, a landfill operated by the major MSW management service provider for the area (Waste Management of Central Massachusetts, West Boylston, MA) were used as model parameters (Appendix C.3, DEP, 2006).

The volume of firewood removed in the harvest was measured in cords, but data on the species composition of this wood was not available. To apply species specific values for oven dry weight and heat content per cord (Kuhns and Schmidt, 2003), which fell within a fairly small range for the major species on the harvest site, the pre-harvest species composition of the harvest site was used to calculate a weighted average of these values for the firewood. Biomass per cord was used to calculate the amount of carbon in this pool, which was treated as a direct flux to the atmosphere. The use of wood for heating offsets fossil fuel burning, so the emissions from burning heating oil (Aubé, 2001) and the relative efficiency of wood and oil heat were used to find the carbon content of the offset emissions. This value was subtracted from the atmospheric flux attributed to firewood.

2.8 Ecological measurements

The abundance of small trees and woody shrubs and leaf area index (LAI) were measured at the harvest and control sites. Though they do not measure major carbon pools, these measurements were useful in characterizing the effect of the harvest on ecosystem function, quantifying recovery, and predicting the carbon dynamics of the harvest site beyond the time frame of this study.

The DBH and species of all trees and woody shrubs 1-5 cm DBH on the harvest site plots was recorded in November, 2006. The same was done on the control site plots for trees and woody shrubs 1-10 cm DBH. These data and 2006 dendrometer data for trees 5-10 cm DBH on the harvest site were used to calculate the frequency and basal area of trees and woody shrubs 1-10 cm DBH by species on the two sites.

LAI was measured several times during the growing season in 1999, 2005, and 2006 using a LI-COR 2000 system (LI-COR Biosciences, Lincoln, NE). At each plot on both the harvest and control sites, one measurement was made at the plot center and four measurements were made 2 m from the plot center in each of the cardinal directions.

Readings from the outer ring of the sensor were masked to eliminate sunspots that may have occurred in this ring. Measurements were processed using the program LI-COR C2000. A LAI value for each plot was calculated as the average of the five subplot readings.

3. Results

3.1 Live biomass

Carbon uptake in live biomass due to growth and recruitment of trees ≥ 10 cm DBH was initially suppressed by the harvest, with additions due to growth and recruitment on the harvest site totaling 1.11 ± 0.39 MgC/ha in 2001 compared to 2.01 ± 0.26 MgC/ha on the control site (numbers of variance give bootstrapped 95% confidence intervals, Figure 3.1). In contrast, growth and recruitment were not significantly different ($P = 0.10$) between the two sites in the year before the harvest. By 2004 and 2005, growth and recruitment on the harvest site increased to 2.39 ± 0.35 and 2.62 ± 0.35, respectively. These fluxes were not significantly different from live biomass additions on the control site, where growth fluctuated due to climatic variability and periodic disturbance. Mortality was high in the harvest year at 13.95 ± 9.43 MgC/ha, because trees removed in the logging operation are included in this figure. Mortality on the harvest site and control site were not significantly different in subsequent years.

The average percent increase in biomass of oaks on the harvest site was 11 ± 3% and 19 ± 4% compared to 22 ± 3% and 27 ± 3% on the control site in 2001 and 2002, respectively, but recovered to values similar to those on the control site in following years (Figure 3.2a), indicating that the harvest had a short-term growth suppression effect on oaks. Red maple and birch, which are early successionals, showed evidence of growth release almost immediately after the harvest, increasing significantly in average percent growth between 2000 and 2002 and surpassing average percent growth for the same species groups on the control site in following years (Figures 3.2b and c). Hemlock, a late successional, increased in average percent growth (7 ± 2% to 12 ± 2%) between 2001 and 2002 and

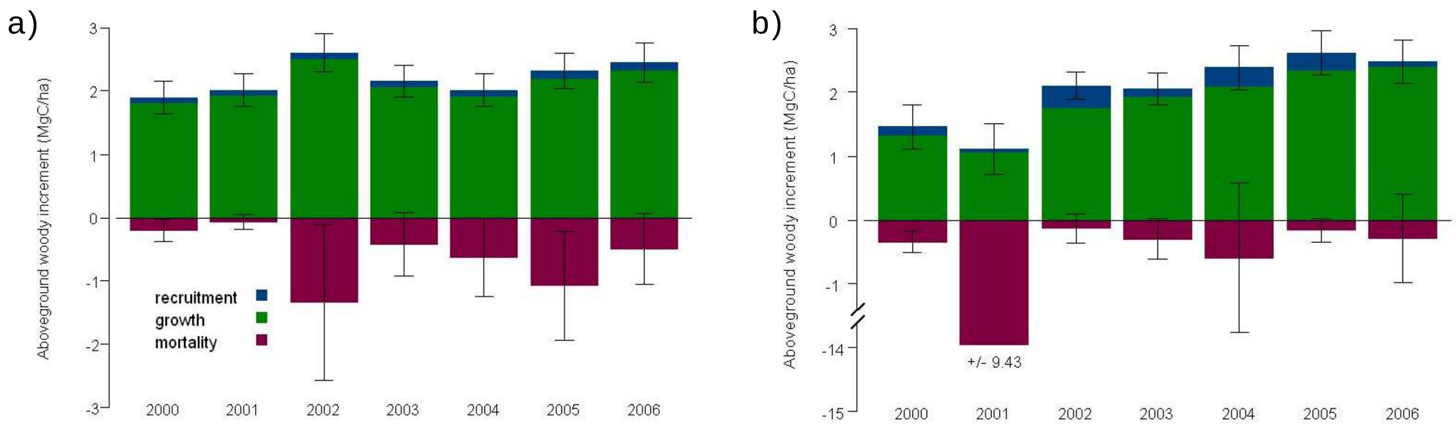

Figure 3.1 Gross annual fluxes in live biomass for 2000-2006 on the (a) control site and (b) harvest site. Green bars represent growth of trees ≥ 10 cm DBH and blue bars represent recruitment of trees into the measured diameter class. Red bars represent mortality. There is a break in the negative *y*-axis on the harvest site figure to display 2001 mortality. Error bars show bootstrapped 95% confidence intervals of growth + recruitment and mortality.

remained greater at the harvest site from 2003 to 2006, though the between-site difference was only significant in 2004 and 2005. The average percent growth for beech, another late successional, increased between 2001 and 2002 (14 ± 4% to 24 ± 5%) on the harvest site, but a between-site comparison was not possible because beech is infrequent on the control site.

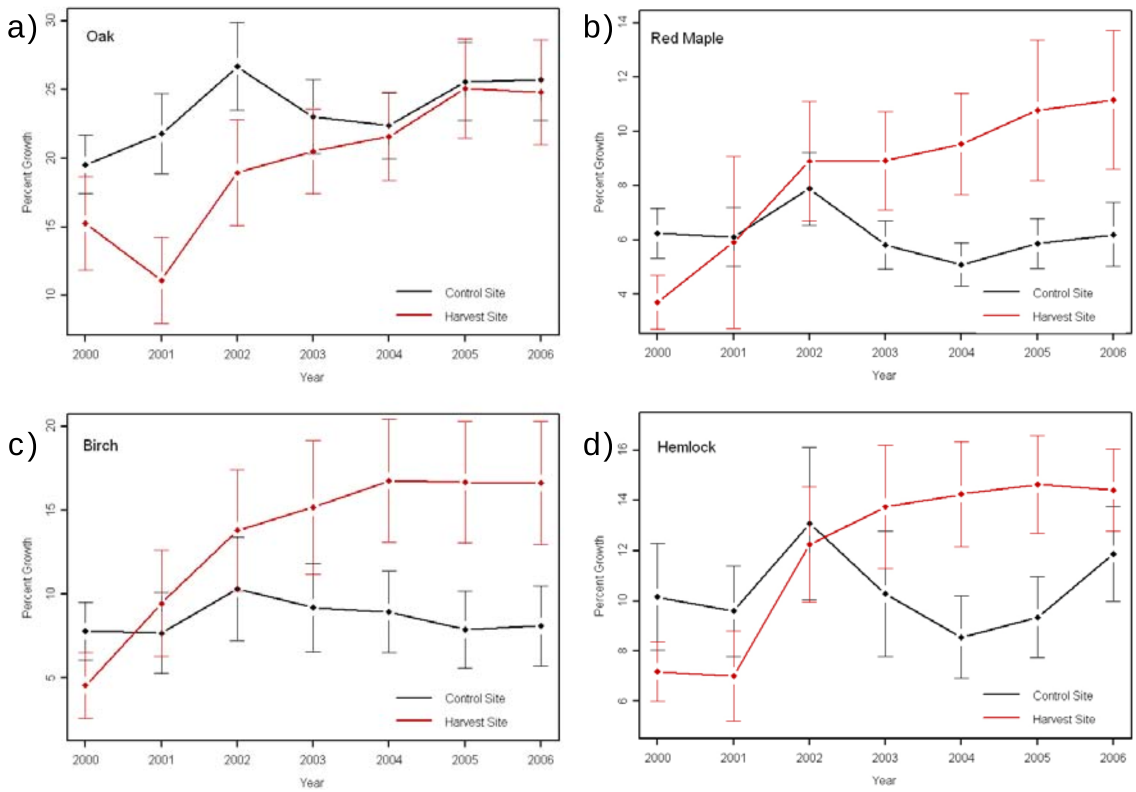

Figure 3.2 Mean percent growth per tree for (a) oak, (b) maple, (c) birch, and (d) hemlock on the harvest and control sites from 2000 to 2006. Error bars show bootstrapped 95% confidence intervals.

Following the harvest, uptake was greater for trees in smaller diameter classes on a per area basis (Figure 3.3). In 2006, trees 10-30 cm diameter at breast height (DBH) accounted for 59 ± 15% of uptake on the harvest site and 33 ± 7% of uptake on the control site. The number of trees per hectare 10-20 and 20-30 cm DBH was not significantly higher on the harvest site in any of the study years ($0.28 < P < 0.88$), so this difference in per area uptake is attributable mostly to growth release and not to recruitment or pre-harvest differences in stand structure. The average uptake for trees in the 10-20 cm diameter class on the harvest site increased from 0.84 ± 0.19 kgC tree^{-1} yr^{-1} in 2000 to 1.84 ± 0.35 kgC tree^{-1} yr^{-1} in 2002, compared to 0.85 ± 0.13 and 1.05 ± 0.19 kgC tree^{-1} yr^{-1} on the control site, and was significantly higher on the harvest site than the control site from 2003 through 2006 (Figure 3.4a). Trees in the 20-30 cm diameter class showed a similar growth increase between 2000 and 2003, going from 2.01 ± 0.49 to 4.48 ± 0.64 kgC tree^{-1} yr^{-1} compared to 2.60 ± 0.31 and 2.86 ± 0.47 kgC tree^{-1} yr^{-1} on the control site (Figure 3.4b). Growth was higher on the harvest site for trees 30-40 cm DBH in 2003 through 2006, though the difference was only significant in 2005 and was proportionally smaller than the between-site difference for trees in the smaller diameter classes (Figure 3.4c). Per tree uptake for trees 40 cm DBH and larger was significantly higher on the control site before the harvest, 11.25 ± 2.11 compared to 6.79 ± 1.01 kgC tree^{-1} yr^{-1}. The between-site difference in this diameter class was not significant for 2001 through 2006, but harvest site uptake increased significantly over this interval to 14.1 ± 3.18 kgC tree^{-1} yr^{-1} (Figure 3.4d).

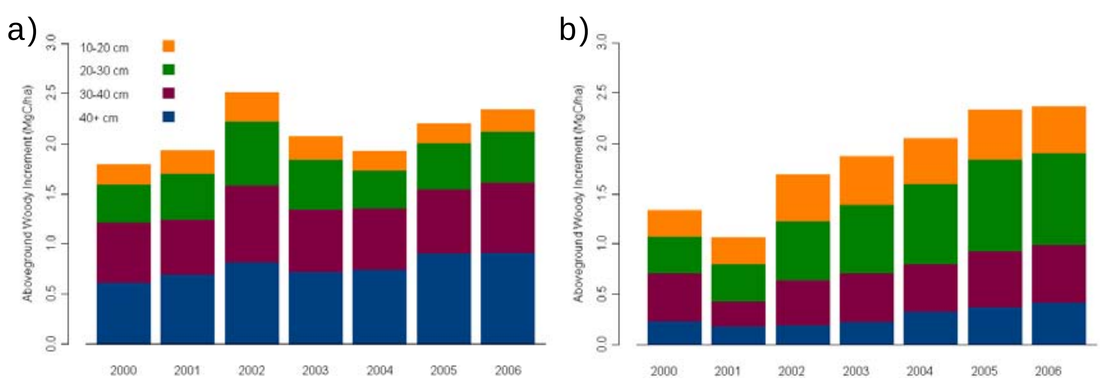

Figure 3.3 Annual increase in aboveground live biomass (MgC/ha) due to growth and recruitment in 2000-2006 on the (a) control site and (b) harvest site, partitioned by diameter class in 10 cm increments. Bottom sections represent trees 40 cm DBH and larger and top sections represent trees 10-20 cm DBH.

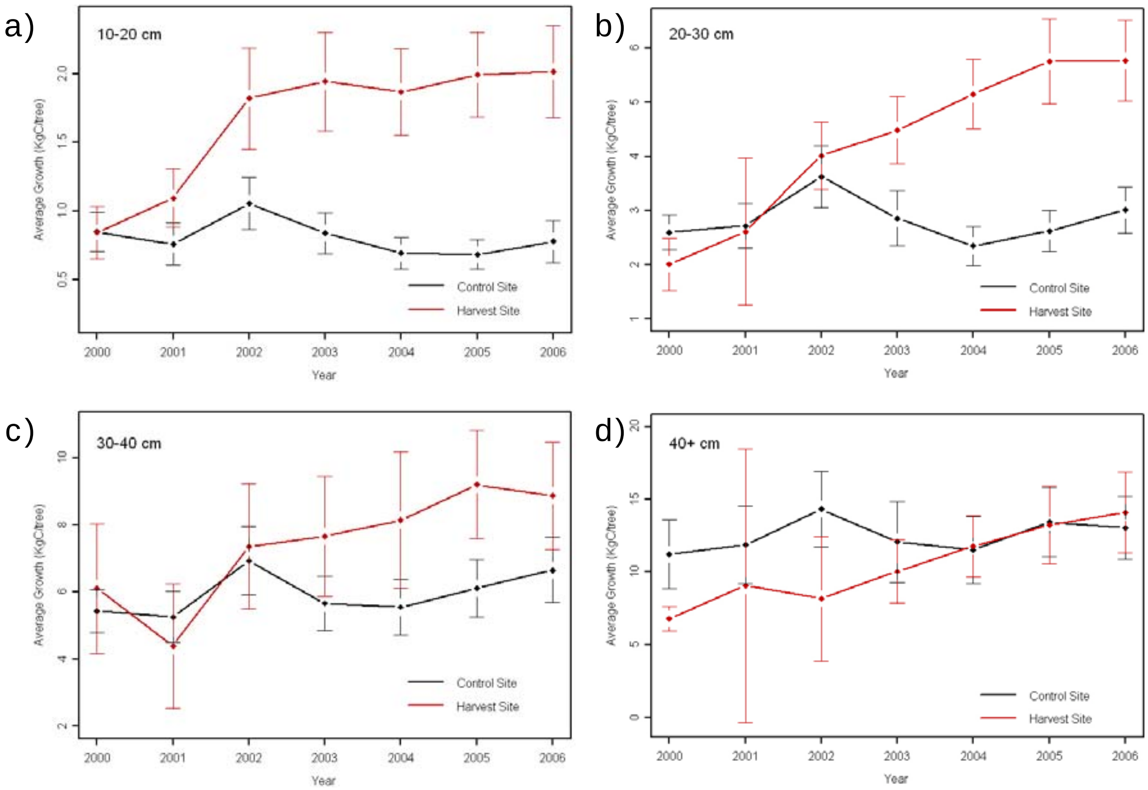

Figure 3.4 Mean annual growth by diameter class per tree (kgC/tree), 2000 – 2006, on the harvest and control sites for trees (a) 10 – 20 cm DBH, (b) 20 – 30 cm DBH, (c) 30 – 40 cm DBH and (d) > 40 cm DBH from 2000 to 2006. Error bars show bootstrapped 95% confidence intervals.

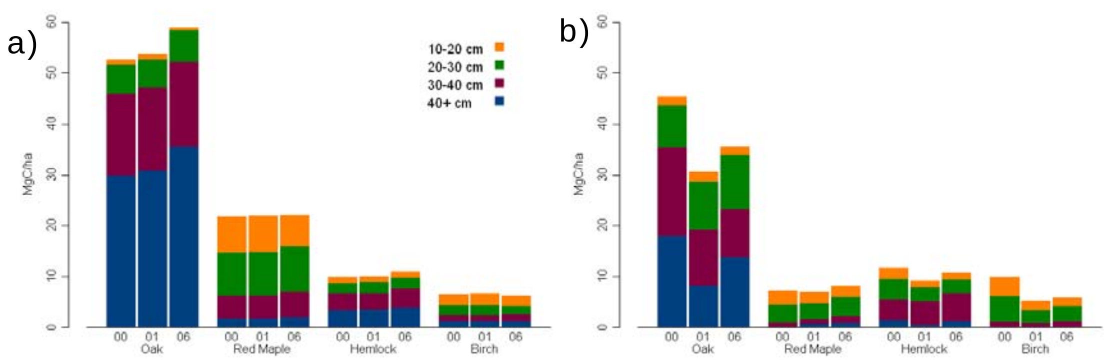

Figure 3.5 Distribution of aboveground biomass (MgC/ha) of major species and species groups by diameter class in 10 cm increments on the (a) control site and (b) harvest site before the harvest (2000), shortly after the harvest (2001), and five years after the harvest (2006). Bottom sections represent trees ≥ 40 cm DBH and top sections represent trees 10-20 cm DBH.

The diameter class distributions of individual species may confound these between-site differences in growth by species and size class. The majority of the biomass (92 ± 18%) of red maple, the species showing the greatest growth increase, was in trees <

30 cm DBH, the size classes showing the greatest growth increase. Only 39 ± 26% of oak was in these diameter classes (Figure 3.5).

Belowground biomass on the harvest site, as estimated with allometric equations from Jenkins *et al.* (2003), equaled 19.4% of aboveground biomass, which agreed well with the estimate that belowground biomass equals 20% of aboveground biomass from Whittaker *et al.* (1974). The 20% estimate was used to be consistent with previous carbon accounting on the study site (Barford *et al.,* 2001).

3.2 Woody debris

CWD biomass was higher on the harvest site than on the control site before the harvest, though this difference was not significant (Figure 3.6). Following the harvest, the carbon stored in CWD biomass was 13.32 ± 3.32 MgC/ha, which was not significantly different than the 1999 harvest site measurement (9.64 ± 3.32 MgC/ha) but was significantly higher than the 2000 control site measurement (5.21 ± 1.39 MgC/ha). Though the decrease in harvest site CWD between 2001 and 2006 was not significant, 2006 was the first survey year since the harvest for which there was not a significant between-site difference in CWD biomass. On the control site, CWD biomass did not change significantly over the study period, and CWD volume remained nearly constant over the study period on both sites (Figure 3.7).

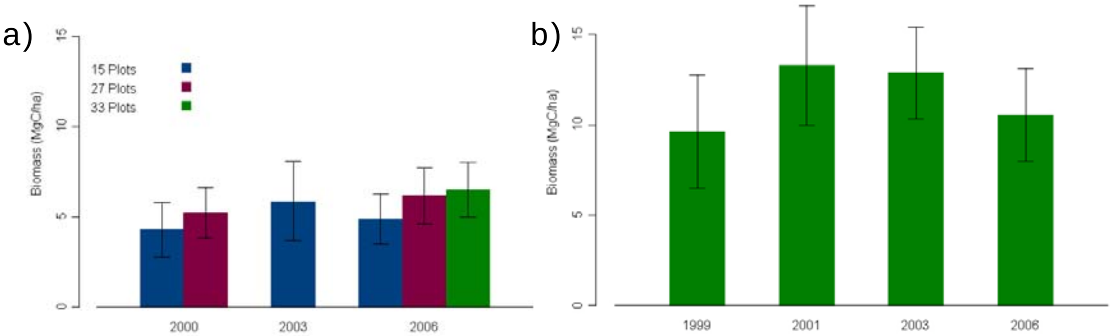

Figure 3.6 Biomass (MgC/ha) of coarse woody debris in survey years on the (a) control and (b) harvest site plots. Control site data is only available for 27 of 33 plots in 2000 and 15 of 33 plots in 2003, so biomass calculated for these subsamples is shown in years when more plots were surveyed. Eight plots were surveyed in all years on the cut site, but the survey in 1999 was done before the plots were expanded from a 10 m to a 15 m radius. Error bars show bootstrapped 95% confidence intervals.

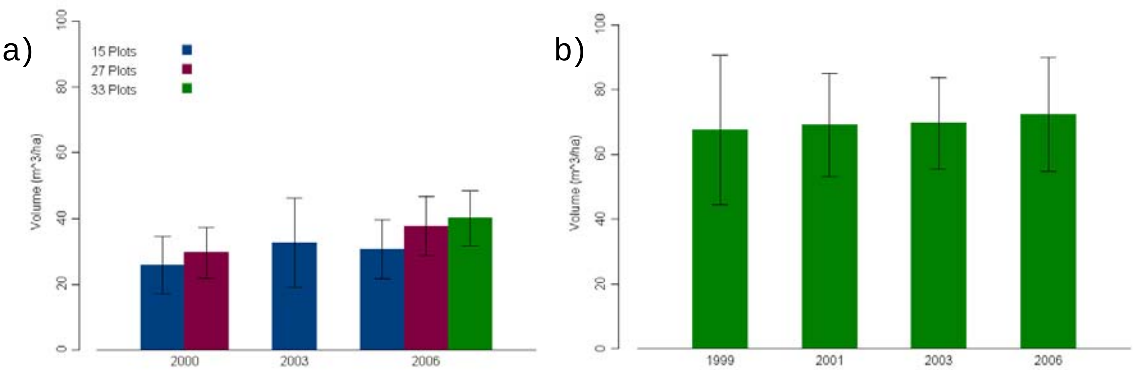

Figure 3.7 Volume (m^3/ha) of coarse woody debris in survey years on the (a) control and (b) harvest site plots. Control site data is only available for 27 of 33 plots in 2000 and 15 of 33 plots in 2003, so biomass calculated for these subsamples is shown in years when more plots were surveyed. Eight plots were surveyed in all years on the cut site, but the survey in 1999 was done before the plots were expanded from a 10 m to a 15 m radius. Error bars show bootstrapped 95% confidence intervals.

The volume of CWD in less decayed classes (1 & 2) on the harvest site increased from 10.02 ± 7.77 to 43.96 ± 14.16 m^3/ha between 1999 and 2001 (Figure 3.8b). The decay class distribution of harvest site CWD volume between 2001 and 2003 was nearly static. Between 2003 and 2006 there was a decrease in the volume of decay classes 1 and 2 from 42.60 ± 11.47 to 5.62 ± 3.44 m^3/ha and a corresponding increase in decay class 3 volume from 11.31 ± 4.35 to 40.06 ± 15.82 m^3/ha. There were no significant changes in the decay class distribution on the control site between 2000 and 2006 (Figure 3.8a). The volume of harvest site CWD in snags decreased from 27.27 ± 13.11 to 8.60 ± 2.89 m^3/ha, while the volume of CWD in stumps increased from 0.76 ± 0.93 to 3.59 ± 0.97 m^3/ha after the harvest, but the harvest site CWD type distribution didn't change significantly between 2001 and 2006 (Figure 3.9a). The CWD type distribution did not change significantly on the control site (Figure 3.9b).

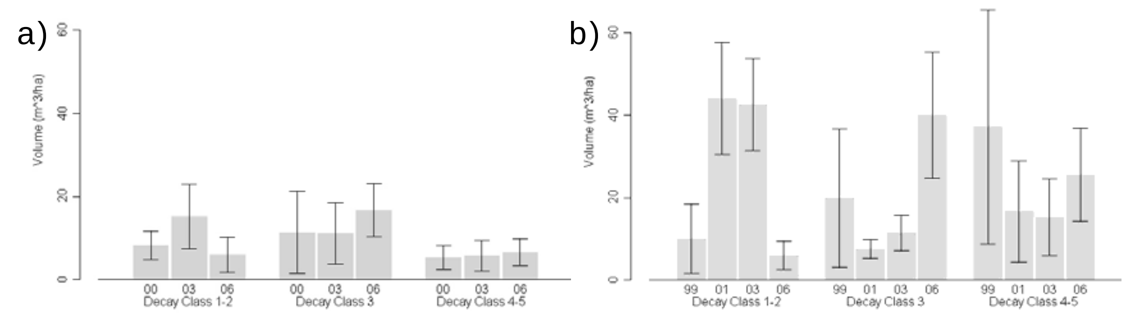

Figure 3.8 Volume of CWD by decay class (m^3/ha) in survey years on the (a) control site and (b) harvest site. Control site data is the 15 plots for which data is available in all survey years. Eight plots were surveyed in all years on the cut site, but the survey in 1999 was done before the plots were expanded from a 10 m to a 15 m radius. Error bars show bootstrapped 95% confidence intervals.

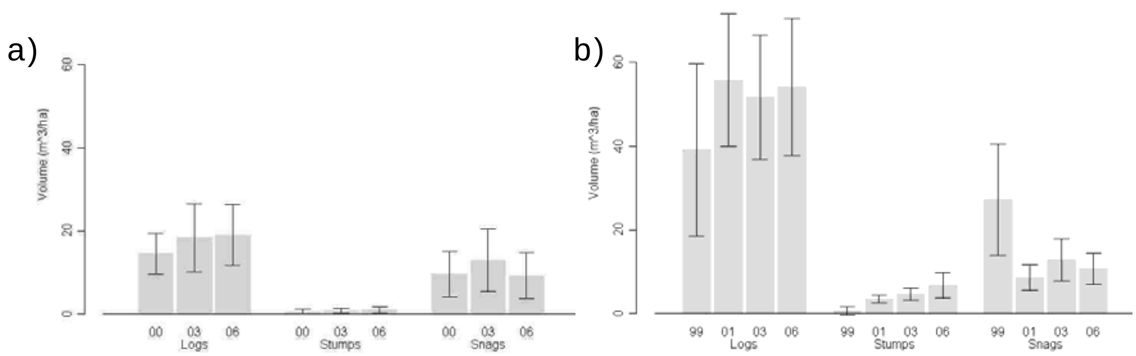

Figure 3.9 Volume (m^3/ha) of CWD by type in survey years on the (a) control site and (b) harvest site. Control site data is based on the 15 plots for which data is available in all survey years. Eight plots were surveyed in all years on the cut site, but the survey in 1999 was done before the plots were expanded from a 10 m to a 15 m radius. Error bars show bootstrapped 95% confidence intervals.

Carbon storage in fine woody debris (FWD) on the harvest site in 2001, 7.81 ± 2.77 MgC/ha, was much higher than on the control site, 1.05 ± 0.40 MgC/ha, and was also significantly higher in 2003 and 2006. There was no significant change in this pool between years on either site (Figure 3.10).

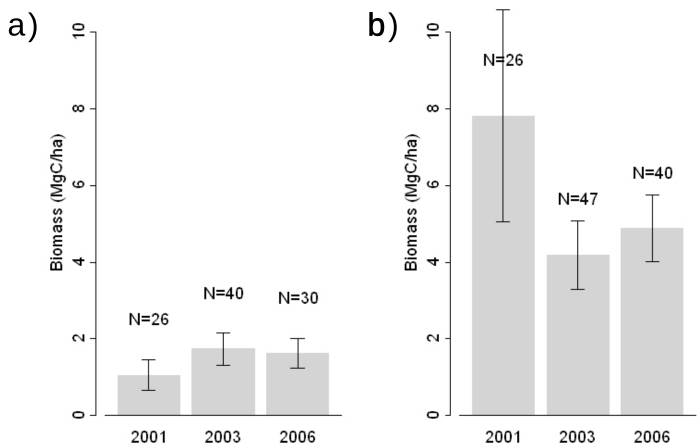

Figure 3.10 Biomass of fine woody debris (MgC/ha) in survey years on the (a) control site and (b) harvest site and the number of segments surveyed. Error bars show bootstrapped 95% confidence intervals.

3.3 Coarse woody debris respiration

Mean modeled site temperatures from 2001 through 2006 were 9.46 ± 0.08°C for the harvest site and 7.52 ± 0.09°C for the control site, which resulted in higher mean CWD respiration rates by decay class on the harvest site. Respiration rates based on modeled site-specific temperatures were similar to those calculated by Liu *et al*. (2006) which were based on a single mean air temperature (7.88°C) measured at the HFEMS (Table 3.1). The average respiration rate, weighted by decay class and standing-downed biomass distribution, was significantly higher on the harvest site (0.0667 ± 0.0008 gC $g^{-1}C\ yr^{-1}$) than on the control site (0.0426 ± 0.0009 gC $g^{-1}C\ yr^{-1}$) before the harvest because the largest portion of the CWD pool on the harvest site (37%) was decay class V and downed which has the highest decay rate, while the largest CWD pool on the control site (39%) was decay class I and standing, which has the lowest decay rate. The weighted average respiration rates for downed CWD from 2001 to 2006 were similar to those reported by Liu *et al.* (2006) for the harvest and control sites, though they resulted in slightly different estimates for the lifetime of downed CWD. Even though the harvest decreased the pool size of more decayed classes and increased the pool size of less decayed classes, weighted average respiration rate was still higher on the harvest site than on the control site for 2001 through 2006, 0.0604 ± 0.0004 compared to 0.0446 ± 0.0003 gC g^{-1}C yr^{-1} (Table 3.2, Figure 3.11). This is due to the decrease in standing CWD, which decays more slowly, after the harvest.

	Control Site	Harvest Site	Liu *et al.*, 2006
Decay Class I	0.0475 ± 0.0003	0.0529 ± 0.0003	0.06 ± 0.02
Decay Class III	0.0725 ± 0.0005	0.0806 ± 0.0005	0.10 ± 0.03
Decay Class V	0.1261 ± 0.0008	0.1403 ± 0.0008	0.14 ± 0.07

Table 3.1 Average annual respiration rate (g C g^{-1} yr^{-1}) for downed coarse woody debris. Control site and harvest site averages are the mean of hourly respiration rates calculated for each decay class using modeled on-site air temperature from 2001 – 2006. Decay rates reported by Liu *et al.* (2006) were calculated using the mean annual air temperatures recorded from 1992 – 2003 at the HFEMS.

	a) Hourly model, 2001-2006		b) Liu *et al.* (2006)	
	Harvest	Control	Harvest	Control
Downed CWD decay rate (yr^{-1})	0.0752 ± 0.0005	0.0739 ± 0.0005	0.08 ± 0.04	0.09 ± 0.04
Total CWD decay rate (yr^{-1})	0.0604 ± 0.0004	0.0446 ± 0.0003	N/A	N/A
Downed CWD lifetime (yr)	13	14	13	11
Total CWD lifetime (yr^{-1})	17	22	N/A	N/A

Table 3.2 Average respiration rates, weighted by decay class and standing-downed distribution, and coarse woody debris lifetimes calculated by (a) the CWD hourly respiration model from 2001 – 2006 and (b) Liu *et al.* (2006), who used the mean annual air temperatures recorded from 1992 – 2003 at the HFEMS.

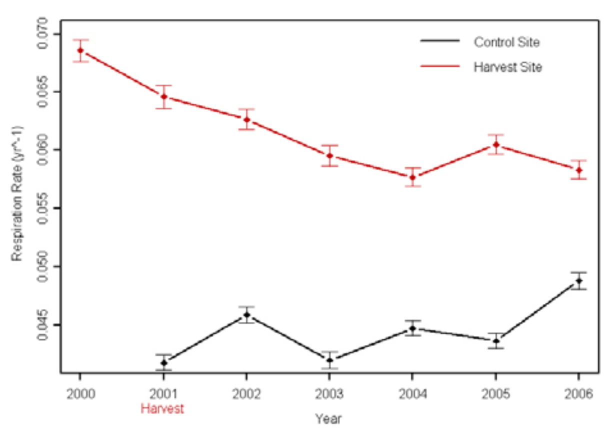

Figure 3.11 Annual CWD respiration rate (gC g^{-1}C yr^{-1}) from 2000 – 2006 on the harvest site (red) and 2001 – 2006 on the control site (black) weighted by decay class and standing-downed biomass distribution. Error bars show 95% confidence intervals.

CWD mass loss due to respiration from 2001 to 2006 totaled 4.7 MgC/ha on the harvest site and 1.6 MgC/ha on the control site (Figure 3.12). Mass losses due to fragmentation and leaching on the control site were 0.42 ± 1.34 MgC/ha between 2000 and 2003 and 0.72 ± 2.37 MgC/ha between 2003 and 2006, the equivalent of 53% and 90% of the losses due to respiration, respectively. On the cut site, fragmentation and leaching accounted for a loss of 1.07 ± 5.31 MgC/ha between 2003 and 2006, 45% of losses to respiration. For the interval from 1999 to 2003 the model yielded a CWD biomass value that was smaller than what was measured so no fragmentation and leaching term could be calculated.

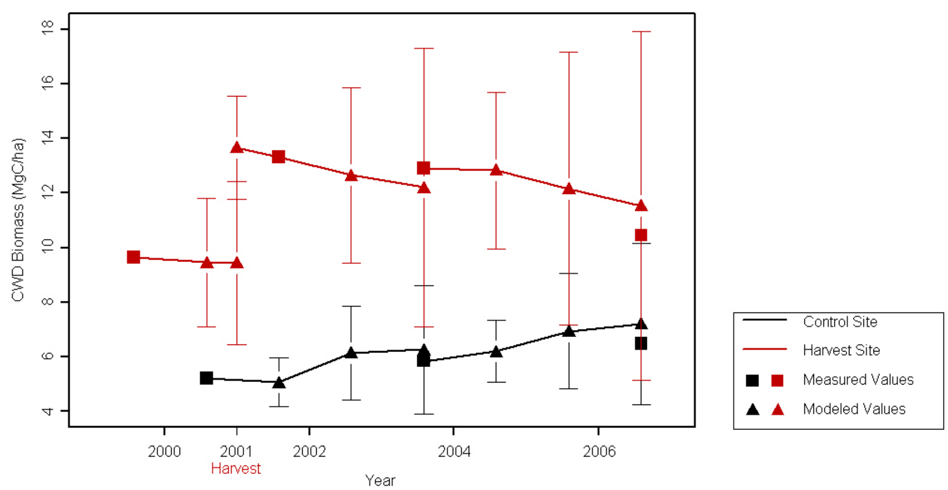

Figure 3.12 Modeled (triangles) and measured (squares) coarse woody debris pool sizes. Squares show measured CWD pool sizes (MgC/ha) and triangles show pool sizes predicted by the CWD respiration model from 1999 – 2006 on the harvest site (red) and 2000 – 2006 on the control site (black). In years for which there is a modeled value and a measured value, the difference between the two is assumed to be the mass lost to fragmentation and leaching between the two surveys. Error bars show 95% confidence intervals of the respiration model. Error from the temperature models was less than 13% of the error from the respiration model in all years.

3.4 Wood products

The high proportion of harvested wood sold as firewood resulted in a large short-term flux of carbon to the atmosphere of 381 MgC due to fuelwood burning, including sawmill residues as fuelwood. If this wood is considered as a substitute for fossil fuel burning, 217 MgC (58%) in emissions are offset (Table 3.3). Carbon emission from fossil fuel burning in harvesting and transport of saw timber and firewood (3.9 MgC) was small relative to the other short-term fluxes. Emissions from timber and pulp processing totaled 70 MgC, with the largest contributions coming from the processing of wood used in flooring (23 MgC), paper (20 MgC), furniture (15 MgC), and construction beams (13 MgC).

	MgC	MgC/ha
Harvesting & Transport	**-3.86**	**-0.09**
Timber & Pulp Processing	**-70.03**	**-1.62**
Sawdust & Sawmill Residue		
Burning	-4.87	-0.11
Fossil Fuel Offset	3.18	0.07
Net Flux	**-1.66**	**-0.04**
Firewood		
Burning	-375.96	-8.72
Fossil Fuel Offset	214.16	4.97
Net Flux	**-161.81**	**-3.75**
Total Short-term Flux	**-237.39**	**-5.51**

Table 3.3 Short-term fluxes of carbon to the atmosphere due to fossil fuel use in logging, transport, and production of wood products and burning of harvested wood with fossil fuel offsets. Negative values represent a net flux of carbon to the atmosphere.

Two-thirds of the sawtimber (105 MgC) entered end-use pools with a median life in use (MLU) of 30 years or more (Table 3.4). As predicted by the HARVCARB model using MLUs adjusted to account for recycling, 73-98% of wood products in these categories will be in use 25 years after production and 28-96% will be in use 50 years after production (Figure 3.13). In the five years following the harvest, the model estimated that the majority (83%) of the paper made from harvested pulpwood and 9.2 of the 29.9 MgC (31%) used to make industrial pallets were disposed, but no more than 6% of wood in the other end-use categories was disposed. The total estimated disposal from 2001 through 2006 was 20.2 MgC (Figure 3.14). Of the wood products disposed, 2.8 MgC were incinerated and 17.4 MgC entered landfills.

Pool	MgC	MgC/ha	MLU	Adjusted MLU
Residential Construction	48.23	1.12	100	110
Non-Residential Construction	31.31	0.73	67	74
Furniture	25.69	0.6	30	33
Manufacturing	11.69	0.27	12	13
Industrial Pallets	29.89	0.69	6	7
Paper	11.43	0.27	1	2
Total	158.24	3.68		

Table 3.4 Initial pool sizes of carbon from saw timber removed from the harvest site into end-use categories based on description of timber use by species and grade for wood from this harvest (Curry 2002). Median life in use (MLU) determined for each category by Skog and Nicholson (1998), is given with the adjusted MLU, which is increased to account for recycling.

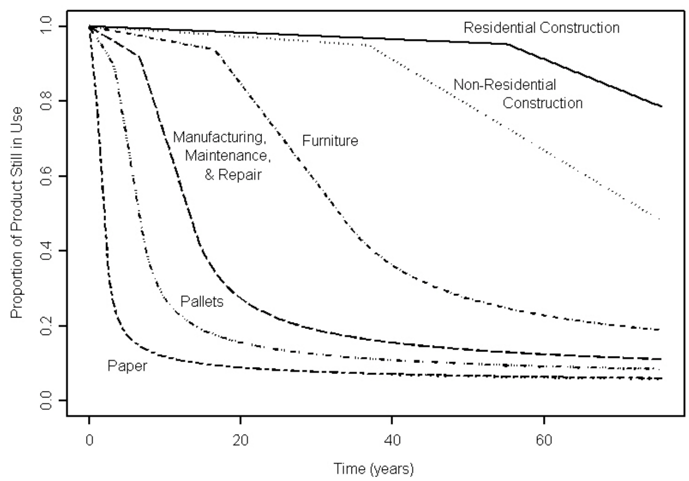

Figure 3.13 Proportion of original wood products still in use by end-use category from 0 to 80 years after production as projected by the HARVCARB model (Row and Phelps, 1996) using median life in use times from Skogg and Nicholson (1998), adjusted to account for recycling.

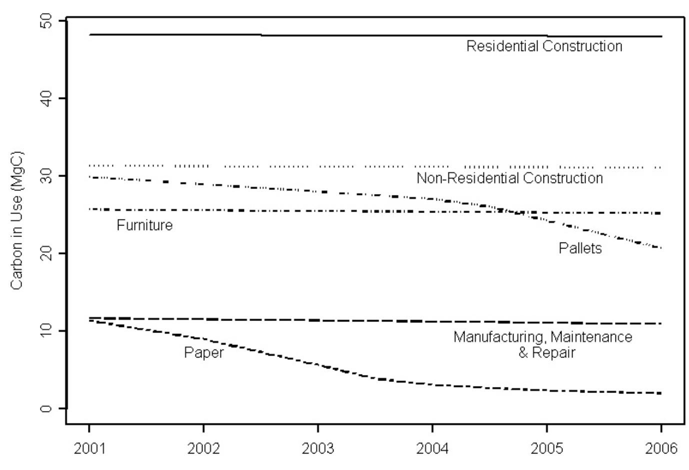

Figure 3.14 Carbon storage in wood products originating from the 2001 harvest from 2001 to 2006 as projected by the HARVCARB model (Row and Phelps, 1996) using median life in use times from Skogg and Nicholson (1998), adjusted to account for recycling.

As the total estimated landfill input from harvested wood products increases, a portion of the carbon from the harvested wood will be returned to the atmosphere through the efflux of landfill gases. The LandGEM model estimated that the annual rate of landfill gas efflux due to inputs of wood products from the harvest site will increase from 0.03 MgC/yr in 2002 to 0.23 MgC/yr by 2010. About 50% of this carbon by mass is decayed through anaerobic respiration and returned to the atmosphere as methane. A flux of 0.23 MgC to the atmosphere as landfill gases has the same global warming potential (GWP) as a flux of 1.01 MgC as CO_2 (Figure 3.15).

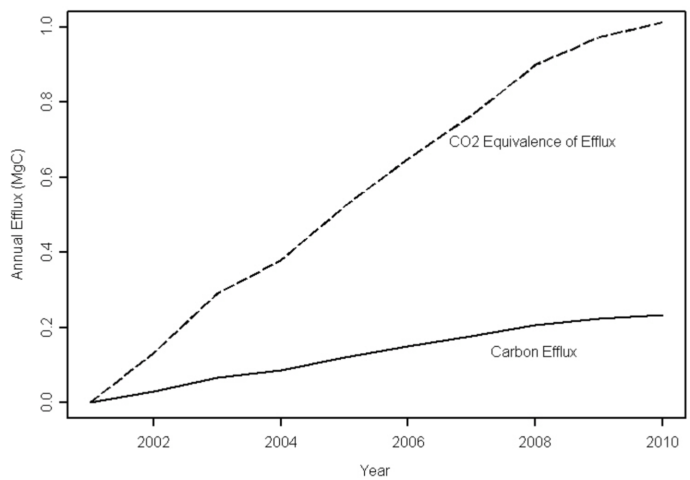

Figure 3.15 Annual landfill gas efflux (MgC), shown by the solid line, due to disposed wood products from the 2001 harvest as modeled using LandGEM v3.02 with model parameters for the Martone Landfill in Barre, MA. Efflux rates are shown from the time of the harvest through 2010, when the landfill is scheduled to close. The dashed line shows the carbon content of a mass of CO_2 with a global warming potential equal to the annual landfill gas efflux.

Between 2001 and 2006, LandGEM estimated that, based on the rate of disposal predicted by HARVCARB and the rate of decay in this landfill, 0.46 MgC from the harvest would be returned to the atmosphere as landfill gases with a GWP equivalent to 1.97 MgC as CO_2. This is not significant on a per hectare basis, so it is not included in the carbon budget. The Martone Landfill closes in 2010, so the combined effects of the disposal rate and efflux rate could not be modeled past this date, but disposed wood products will have a significant effect on the carbon dynamics of the system beyond this 10-year time scale. Under the conditions estimated for disposed wood products from this harvest, 1 MgC disposed in 2001 will result in the efflux of 0.34 MgC in landfill gases with a GWP equivalent to 1.46 MgC as CO_2 by 2100 (Figure 3.16).

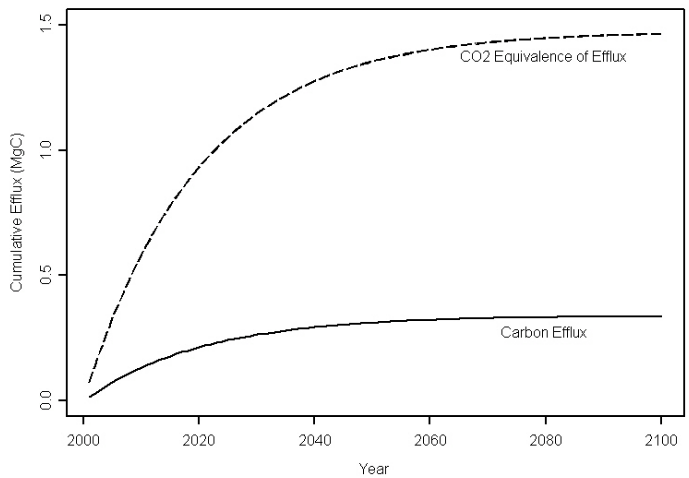

Figure 3.16 Cumulative efflux of 1 MgC in landfill waste, 2001-2100. The solid line shows the cumulative carbon content of landfill gas efflux due to wood products containing 1 MgC disposed of in 2001 over 100 hundred years as predicted by LandGEM v3.02. The dashed line shows the carbon content of a mass of CO_2 with a global warming potential equivalent to the cumulative landfill gas efflux.

3.5 Carbon budget

There was a large flux of 15.5 ± 10.7 MgC/ha out of the live carbon pools in 2001 due to the harvest (Table 3.5). The aboveground portion of this went into CWD (4.2 ± 3.7 MgC/ha, calculated using the CWD respiration model), FWD (5.5 ± 4.6 MgC/ha), wood products (3.7 MgC/ha), and fuelwood (8.8 MgC/ha) pools. The estimated sizes of these smaller pools total 22.1 MgC/ha, which agrees reasonably well with the aboveground mortality estimate (14.0 ± 11.2 MgC/ha, 18.4 ± 11.3 MgC/ha if trees 5-10 cm DBH are included). After short-term fluxes are accounted for, harvested wood did not contribute to the net annual atmosphere-biosphere carbon flux for the harvest site because landfill gas efflux and MSW incineration is not significant over the time interval included in the carbon budget. In 2002 and 2003, the fluxes of carbon to the atmosphere due to woody debris decay and to the ecosystem due to tree growth nearly balanced out, resulting in net carbon fluxes close to zero. On-site carbon uptake increased to a level similar to that of the control site between 2003 and 2004 (Table 3.6), though this increase can be attributed almost entirely to the stabilization of the FWD pool, for which uncertainty is high. The harvest site was a net carbon sink with a magnitude similar to the control site from 2004 to 2006.

The loss of carbon from the harvest site in 2001 was counterbalanced by uptake in subsequent years, making the on-site carbon pools a small net carbon sink of 1.4 MgC/ha over the five year period since the harvest. Fluxes in the aboveground carbon pools, which were measured directly, show the same pattern of decrease and recovery following the harvest as the overall on-site carbon dynamics (Table 3.7). However, these pools are a net carbon source in 2002 and 2003 while the site as a whole is close to neutral. Considering on-site carbon pools, short-term carbon fluxes, fossil fuel offsets, and carbon storage in wood products together, the system is a net carbon sink of 8.4 MgC/ha over the five year period (Figure 3.16). The magnitude of this sink is still about half that of the control site, which sequestered a total of 16.4 MgC/ha over the same time interval.

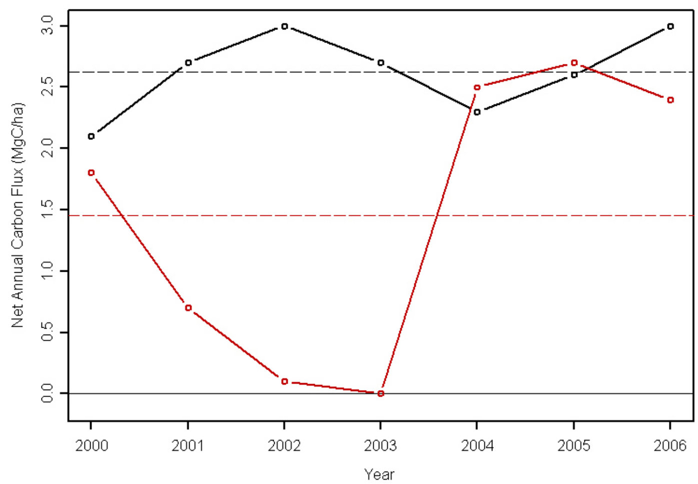

Figure 3.16 Annual carbon flux (MgC/ha) from the year before the 2001 harvest through 2006 in relation to the atmosphere for the control site and the harvest site including carbon in harvested wood. Negative values indicate a net flux to the atmosphere. The dashed lines show the mean annual net fluxes over the time period for the two sites.

	2000	2001	2002	2003	2004	2005	2006	**Total, 2001 – 2006**
Live Biomass								
Aboveground								
Growth	1.5±0.5	1.1±0.4	2.1±0.3	2.1±0.3	2.4±0.4	2.6±0.4	2.5±0.4	**12.8**
Mortality	-0.3±0.6	-14.0±11.2	-0.1±0.2	-0.3±0.3	-0.6±1.2	-0.2±0.2	-0.3±0.7	**-15.5**
Belowground								
Growth	0.3±0.1	0.2±0.1	0.4±0.1	0.4±0.1	0.5±0.1	0.5±0.1	0.5±0.1	**2.5**
Mortality	-0.1±0.1	-2.8±2.2	0.0±0.0	-0.1±0.1	-0.1±0.2	0.0±0.0	-0.1±0.1	**-3.1**
Subtotal	1.6±0.5	-15.5±10.7	2.4±0.3	2.1±0.4	2.2±1.2	2.9±0.4	2.6±0.8	**-3.3**
Woody Debris (WD)								
Coarse WD								
Mortality Input	0.3±0.6	4.2±3.7	0.1±0.2	0.3±0.3	0.6±1.2	0.2±0.2	0.3±0.7	**5.7**
Respiration	-0.6±1.8	-0.7±1.9	-0.8±2.5	-0.8±1.6	-0.7±2.2	-0.8±1.8	-0.9±1.2	**-4.7**
Fine WD	0.3±0.6††	5.5±4.6†	-1.8±2.4*	-1.8±2.4*	0.2±1.2*	0.2±1.2*	0.2±1.2*	**2.5**
Subtotal	0.0	9.0	-2.5	-2.3	0.1	-0.4	-0.4	**3.5**
Net Soil C Flux	0.2±0.1	0.2±0.1	0.2±0.1	0.2±0.1	0.2±0.1	0.2±0.1	0.2±0.1	**1.2**
On-site total	1.8	-6.3	0.1	0.0	2.5	2.7	2.4	**1.4**
Harvested Wood								
Fossil Fuel Use		-1.7						**-1.7**
Wood Products		3.7	-0.1	-0.1	-0.1	-0.1	-0.1	**3.2**
Fuel Wood		8.8						**8.8**
Fuel Wood Burned		-8.8						**-8.8**
Landfill Input			0.1	0.1	0.1	0.1	0.1	**0.5**
Fossil Fuel Offsets		5.0						**5.0**
Subtotal		7.0	0.0	0.0	0.0	0.0	0.0	**7.0**
Annual Total	**1.8**	**0.7**	**0.1**	**0.0**	**2.5**	**2.7**	**2.4**	**8.4**

Table 3.5 Annual fluxes (MgC/ha) in carbon pools on the harvest site, 2000-2006. Positive annual totals indicate flux from the atmosphere to terrestrial carbon pools. Growth in aboveground live biomass includes inputs from recruitment and is calculated for trees ≥ 10 cm DBH. Breakdown of coarse woody debris flux was estimated using the woody debris respiration model. Belowground live biomass fluxes are estimated as 20% of aboveground fluxes based on Whittaker et al. (1974). Soil carbon flux is estimated based on measurement of the residence time of ^{14}C in Harvard Forest soils (Gaudinski et al. 2000). *Multiyear averages. †Estimated as the difference between measured post-harvest FWD and 25% of pre-harvest CWD because there was no reference FWD measurement. ††Estimated as equal to the 2001 – 2003 annual FWD flux on the control site. Bootstrapped 95% confidence intervals are given where they could be calculated.

	2000	2001	2002	2003	2004	2005	2006	**Total, 2001 – 2006**
Live Biomass								
Aboveground								
Growth	1.9±0.2	2.0±0.3	2.6±0.3	2.2±0.3	2.0±0.3	2.3±0.3	2.5±0.3	**13.6**
Mortality	-0.2±0.2	-0.1±0.1	-1.3±1.2	-0.4±0.5	-0.6±0.6	-1.1±0.9	-0.5±0.5	**-4.0**
Belowground								
Growth	0.4±0.0	0.4±0.0	0.5±0.1	0.4±0.1	0.4±0.1	0.5±0.1	0.5±0.1	**2.7**
Mortality	0.0±0.0	0.0±0.0	-0.3±0.3	-0.1±0.1	-0.1±0.1	-0.2±0.2	-0.1±0.1	**-0.8**
Subtotal	2.1±0.2	2.3±0.3	1.5±0.5	2.1±0.4	1.7±0.3	1.5±0.4	2.4±0.3	**11.5**
Woody Debris (WD)								
Coarse WD								
Mortality Input		0.1±0.1	1.3±1.2	0.4±0.5	0.6±0.6	1.1±0.9	0.5±0.5	**4.0**
Respiration		-0.2±0.9	-0.3±1.7	-0.3±2.3	-0.3±1.1	-0.3±2.1	-0.2±3.0	**-1.6**
Fine WD		0.3±0.6†	0.3±0.6*	0.3±0.6*	0.1±0.6*	0.1±0.6*	0.1±0.6*	**1.2**
Subtotal		0.2	1.3	0.4	0.4	0.9	0.4	**3.6**
Soil Carbon		0.2±0.1	0.2±0.1	0.2±0.1	0.2±0.1	0.2±0.1	0.2±0.1	**1.2**
Total	**2.1****	**2.7**	**3.0**	**2.7**	**2.3**	**2.6**	**3.0**	**16.3**

Table 3.6 Annual fluxes (MgC/ha) in carbon pools at the control site, 2000-2006. Positive annual totals indicate flux from the atmosphere to terrestrial carbon pools. Growth in aboveground live biomass includes inputs from recruitment and is calculated for trees ≥ 10 cm. Breakdown of coarse woody debris flux was estimated using the woody debris respiration model. Belowground live biomass fluxes are estimated as 20% of aboveground fluxes based on Whittaker et al. (1974). Soil carbon flux is estimated based on measurement of the residence time of ^{14}C in Harvard Forest soils (Gaudinski et al. 2000). *Multiyear averages. **Eddy flux measurement (Barford et al. 2001). †Estimated as equal to the 2001 – 2003 average annual flux because 2001 is the first measurement year for FWD. Bootstrapped 95% confidence intervals are given where they could be calculated.

a) Harvest Site	2000	2001	2002	2003	2004	2005	2006	**Total, 2001 – 2006**
Live Biomass								
Growth	1.5±0.5	1.1±0.4	2.1±0.3	2.1±0.3	2.4±0.4	2.6±0.4	2.5±0.4	**12.8**
Mortality	-0.3±0.6	-14.0±11.2	-0.1±0.2	-0.3±0.3	-0.6±1.2	-0.2±0.2	-0.3±0.7	**-15.5**
Subtotal	1.2±0.5	-12.9±9.8	2.0±0.2	1.8±0.3	1.8±1.0	2.4±0.3	2.2±0.6	**-2.7**
Woody Debris (WD)								
Coarse WD								
Mortality Input	0.3±0.6	4.2±3.7	0.1±0.2	0.3±0.3	0.6±1.2	0.2±0.2	0.3±0.7	**5.7**
Respiration	-0.6±1.8	-0.7±1.9	-0.8±2.5	-0.8±1.6	-0.7±2.2	-0.8±1.8	-0.9±1.2	**-4.7**
Fine WD	0.3±0.6†	5.5±4.6††	-1.8±2.4*	-1.8±2.4*	0.2±1.2*	0.2±1.2*	0.2±1.2*	**2.5**
Subtotal	0.0	9.0	-2.5	-2.3	0.1	-0.4	-0.4	**3.5**
Total	**1.2**	**-3.9**	**-0.5**	**-0.5**	**1.9**	**2.0**	**1.8**	**0.8**

b) Control Site	2000	2001	2002	2003	2004	2005	2006	**Total, 2001 – 2006**
Live Biomass								
Growth	1.9±0.2	2.0±0.3	2.6±0.3	2.2±0.3	2.0±0.3	2.3±0.3	2.5±0.3	**13.6**
Mortality	-0.2±0.2	-0.1±0.1	-1.3±1.2	-0.4±0.5	-0.6±0.6	-1.1±0.9	-0.5±0.5	**-4.0**
Subtotal	1.7±0.2	1.9±0.3	1.3±0.4	1.8±0.3	1.4±0.2	1.2±0.3	2.0±0.3	**9.6**
Woody Debris (WD)								
Coarse WD								
Mortality Input		0.1±0.1	1.3±1.2	0.4±0.5	0.6±0.6	1.1±0.9	0.5±0.5	**4.0**
Respiration		-0.2±0.9	-0.3±1.7	-0.3±2.3	-0.3±1.1	-0.3±2.1	-0.2±3.0	**-1.6**
Fine WD		0.3±0.6†	0.3±0.6*	0.3±0.6*	0.1±0.6*	0.1±0.6*	0.1±0.6*	**1.2**
Subtotal		0.2	1.3	0.4	0.4	0.9	0.4	**3.6**
Total		**2.1**	**2.6**	**2.2**	**1.8**	**2.1**	**2.4**	**13.2**

Table 3.7 Annual fluxes (MgC/ha) in on-site, aboveground carbon pools on the (a) harvest site and (b) control site, 2001-2006. Growth in aboveground live biomass includes inputs from recruitment. Breakdown of coarse woody debris flux was estimated using a woody debris respiration model. *Multiyear averages. †Estimated as equal to the 2001 – 2003 average annual FWD flux for the control site because a measurement is not available. ††Estimated as the difference between measured post-harvest FWD and 25% of pre-harvest CWD because there was no reference FWD measurement. Bootstrapped 95% confidence intervals are given where they could be calculated.

3.6 Sapling abundance

The frequency of trees and woody shrubs 1-10 cm DBH was not significantly different between the two sites, with 1239 ± 293 stems/ha on the control site compared to 1135 ± 554 stems/ha on the harvest site, but the basal area of this size class was significantly higher on the harvest site at 2.89 ± 0.84 m^2/ha than on the control site at 1.65 ± 0.32 m^2/ha (Figure 3.17). Woody shrubs were more frequent on the control site at 453 ± 218 stems/ha than on the harvest site at 60 ± 49 stems/ha. Beech was the most frequent species in this diameter class on the harvest site with 292 ± 99 stems/ha, and was significantly more frequent than oak with 21 ± 32 stems/ha. Both red maple and beech accounted for significantly more of the basal area in this diameter class than oak on the harvest site and the control site (Figure 3.18).

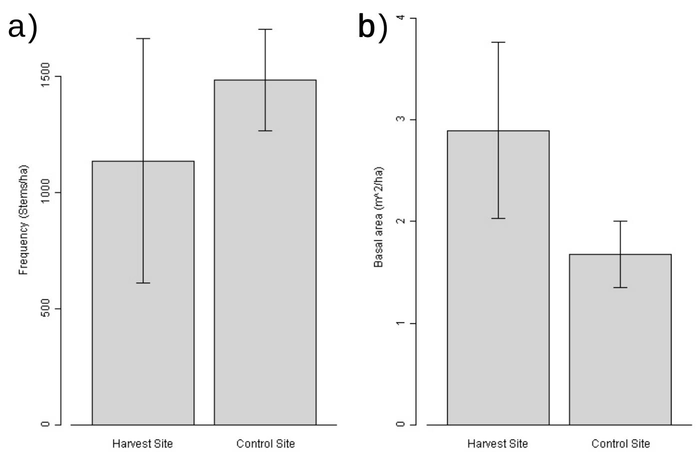

Figure 3.17 Abundance of trees and woody shrubs 1-10 cm DBH on the harvest and control sites in 2006 given by (a) frequency (stems/ha) and (b) basal area (m^2/ha). Error bars show bootstrapped 95% confidence intervals.

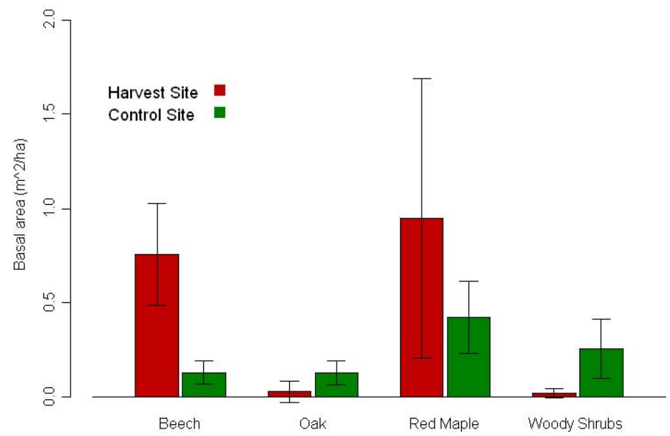

Figure 3.18 Basal area (m^2/ha) of trees 1-10 cm DBH in major species groups on the harvest site (red bars) and control site (green bars) in 2006. Error bars show bootstrapped 95% confidence intervals.

3.7 Leaf area index

Before the harvest, there was no significant between-site difference in peak LAI, which was 5.06 ± 0.57 on the harvest site and 4.96 ± 0.27 on the control site in 1999. Four and five years after the harvest, peak LAI was significantly lower on the harvest site at 4.21 ± 0.33 and 4.07 ± 0.38 in 2005 and 2006, respectively, compared to 5.49 ± 0.27 and 5.02 ± 0.32 on the control site. Though peak LAI was lower on the harvest site, there were no between-site differences in LAI early and late in the growing season (Figure 3.19).

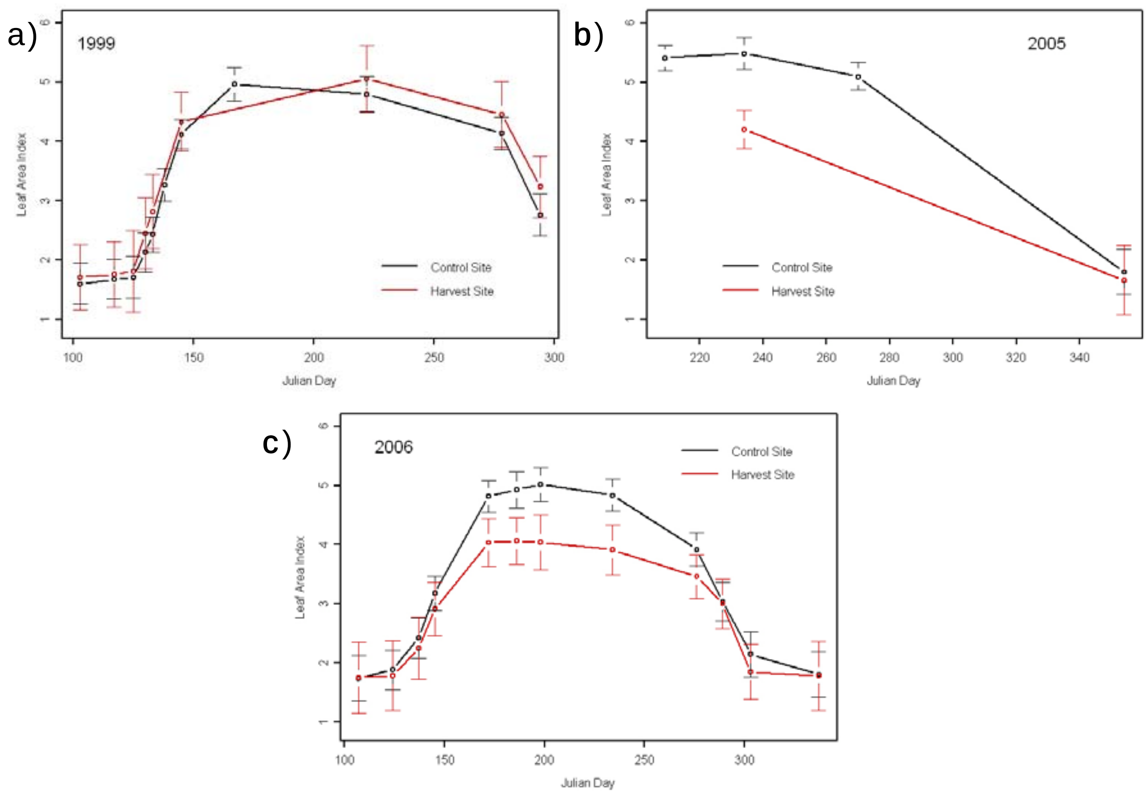

Figure 3.19 Leaf area index (LAI) measured throughout the growing season before the harvest in (a) 1999, and after the harvest in (b) 2005 and (c)2006 on the harvest site (red lines) and control site (black lines). Error bars show bootstrapped 95% confidence intervals.

4. Discussion

4.1 Changes in stand dynamics

Though the overall rate of carbon uptake in live biomass recovered quickly following the harvest, the increase in growth of smaller trees and early-successional species suggests that the growth dynamics of the stand were altered. The rapid growth of species like red maple and black birch typically dominate the early stages of stand development in New England mixed hardwood forests. Oak may gradually outcompete these trees and becomes the dominant species (Oliver and Larson, 1996). This is how the productive red oak stands on the control site developed (Wofsy, 2004), and may be the future trajectory of the harvest site. However, the high abundance of beech in comparison to oak in the 1-10 cm DBH size class could indicate that beech will move into the canopy as early-successional species die out.

The harvest caused a large reduction in total standing biomass and LAI measurements in 2005 and 2006 showed that the canopy has not fully recovered on the harvest site. Despite this, carbon uptake in live biomass was not significantly different between the sites in 2002 through 2006, indicating that the rapid growth of early-successional species following the harvest gave this pool a high degree of resilience. It also suggests that the rate of carbon uptake in growth by a young, early-successional stand is similar to that of a mature mid-successional stand, like the red oaks on the control site. If growth on the harvest site increases over the next 10 to 15 years, uptake could exceed the control site, leading to a peak in uptake in the early-successional stage. If this is the case, it is uncertain what the carbon dynamics of a stand that is between an early-successional peak and the mid-successional stands in which increasing growth rates have been observed (Pederson, 2005; Urbanski *et al*., in press). These growth dynamics also imply that the pattern of productivity during stand development and succession in these forests may be more complex than the single peak and decline suggested by Odum's model of ecosystem development (Odum, 1969).

After the initial recovery from growth suppression due to the harvest, it does not appear that there was an upward trend in growth on the harvest site between 2002 and 2006. However, it took over 10 years of data collection before a trend was evident in uptake at the HFEMS (Urbanski *et al.*, in press), so it may be too early to observe increasing uptake on the harvest site. The dynamics of carbon uptake as the growth of early-successional species slows and oak or beech begins to dominate uptake on the harvest site are uncertain, and measurement of tree growth on the harvest site should continue to observe this potential transition.

4.2 Coarse woody debris pool dynamics

The harvest affected the distribution of the CWD pool in ways that have consequences for the rate at which the carbon in this pool will be respired to the atmosphere. Logs left in slash piles during the harvest seem to be the cause of the large increase in the volume of CWD in the less decayed classes in 2001 (Figure 3.8). It appears that most of this volume transitioned to decay class 3 between the 2003 and 2006 CWD surveys, indicating that a similar shift to decay classes 4 and 5 may occur in the next three to six years. Because the more decayed classes respire more quickly, this could lead to a period of very rapid carbon flux to the atmosphere from the CWD pool, possibly altering the annual carbon dynamics of the site. Measurement of this pool should continue to monitor these future dynamics. The three year measurement interval seemed to effectively capture the major changes in CWD following the harvest.

The harvest also significantly reduced the total volume of snags (Figure 3.9). This was probably due to snags being knocked over by loggers, though some snags may have been removed as firewood (L. Hutyra, personal communication). This is consistent with findings that a selective logging operation that removes one-fourth of live trees also removes or knocks down about 70% of existing snags (Holloway *et al*., 2007). As downed logs, this volume of CWD respires more quickly (Erickson *et al.*, 1985, and Mattson *et al.*, 1987), returning the carbon to the atmosphere over a shorter time interval, though respiration may be slower than predicted for downed CWD if the logs in slash piles are not in contact with the ground.

The importance of standing-downed CWD distribution in determining the overall decay rate on a site is evident in the differences in modeled decay rate between the two sites. If only downed CWD is considered, the difference in respiration between the two sites is much smaller than if standing CWD, which is more prevalent on the control site, is considered (Table 3.2a). In addition to the consequences of standing CWD volume for respiration, snags are important for other ecosystem functions, especially for providing suitable wildlife habitat (Conner *et al.*, 1975; DeGraaf and Shigo, 1985; Healy *et al.*, 1989).

4.3 Uncertainty in the CWD respiration model

The between-site differences in modeled decay class-specific respiration rates (Table 3.1) indicate that the CWD respiration model was effective in addressing the effect of differences in micrometeorological conditions on CWD pool dynamics, but there are some weaknesses in its assumptions. The assumption that standing CWD respires at 40% of the rate of downed CWD decay comes from studies in northwestern conifer (Erickson *et al.* 1985) and clear-cut southeastern (Mattson *et al.*, 1987) forests, and should be verified with empirical measurements from stands more similar to the site of this study. The assumption that logs are entirely in contact with the ground and snags and stumps are not in contact with the ground is also problematic. More of a stump or short snag is in contact with the ground than a tall snag, and parts of some logs are held off the ground by other pieces of CWD, especially if they are in slash piles, or are not entirely in contact with the ground because of their shape. For these reasons, there may be more variation in respiration rate among pieces of CWD due to degree of contact with the ground than is represented in the standing-downed difference for which the model accounts.

The rates at which CWD shifted to more decayed classes and from standing to downed pools were calculated by tracking individual pieces between surveys so that new inputs would not be included in the calculated probabilities. For a piece of CWD to be identified in a subsequent survey, an aluminum tag nailed into the wood had to stay attached for two to three years. It appeared that the tags were more likely to fall off of more decayed wood, so the shift rate to more decayed classes may be underestimated. This is less of an issue with the standing to downed shift rate because it was calculated by tracking snags which are almost all in the less decayed classes and thus rarely lose their tags. However, the rate of input to the downed pools from this standing to downed shift accounted only for inputs from falling snags. Downed CWD can also be created by branches falling from live trees, which is difficult to simulate in a model, so inputs to downed CWD may be underestimated.

The models for on-site temperature made the assumption that the relationship between on-site temperature and tower measurements of temperature and incoming radiation in 2004 and 2005, when micrometeorological measurements were available, was the same as in other years. This is more problematic on the harvest site, where it is likely that the light environment changed in the years following the harvest.

It appears that the model overestimated respiration on the harvest site between the harvest and the 2003 CWD survey, as this is the only survey year in which the modeled pool size is smaller than the measured pool size (Figure 3.12). One possible reason for this is that the temperature model made using micrometeorological data from 2004 and 2005 did not produce an accurate representation of the on-site temperatures in the years immediately after the harvest. A more open canopy in these years may have caused drier conditions and less respiration. Also, most of the CWD inputs from the harvest were left in slash piles where many pieces of wood were held off the ground by other pieces and may have decayed more slowly than the model predicts.

Despite these issues, the CWD respiration model was useful in linking small-scale measurement of CWD respiration rate (Liu *et al.*, 2006) to ecosystem-level carbon dynamics. It also allowed for an approximation of annual CWD pool dynamics, as field measurement of this pool was not possible on an annual basis.

4.4 Uncertainty in soil carbon and belowground biomass fluxes

The largest sources of uncertainty in the carbon budget are the estimates of soil carbon uptake and belowground biomass fluxes on the harvest site because they were not measured directly. In a meta-analysis of seven studies on the effect of harvesting on soil carbon in hardwood forests, Johnson and Curtis (2001) found that average soil carbon content decreased slightly after harvest, but this effect was not statistically significant and varied greatly with location and ecosystem type. A study of soil dynamics following a harvest at the Hubbard Brook Experimental Forest in New Hampshire found that harvesting caused significant disturbance and compaction of soil but losses of carbon due to respiration and leaching following the harvest were balanced by inputs from root and leaf litter breakdown (Johnson *et al.,* 1991). These findings support our assumption that the immediate effects of the harvest did not result in a significant flux of carbon into or out of the belowground system. In the years following the harvest, however, the differences in microclimate between the sites (Table 2.1) may have caused differences in soil temperature and moisture. A study conducted at the Harvard Forest found that soil respiration varied spatially and temporally with differences in temperature and moisture (Davidson *et al.*, 1998), so the between-site microclimatic differences may have affected overall soil respiration. Peng and Thomas (2006) found that the effect of selective harvest in a northern hardwood forest on soil respiration was only partially explained by differences in soil temperature and moisture and is also due to changes in belowground biomass. They found that soil respiration increased in the year after selective harvest then declined to below pre-harvest levels and gradually recovered to pre-harvest levels 5-

6 years after logging, resulting in an overall soil CO_2 efflux slightly below that on unharvested sites.

The estimate of belowground biomass using a proportion of aboveground biomass or allometric equations is also problematic as these estimation techniques were developed for unharvested sites. Oak, maple, and beech, can grow from stump sprouts and maple and beech from root suckers (Tirmenstein, 1991a; Tirmenstein, 1991b; Coladonato 1991), so the mortality of a tree may not necessarily result in the morality of its entire root system. For this reason, allometric equations have been found to systematically underestimate root biomass in young stands (Park *et al.,* 2007). Logging operations can also damage the roots of unharvested trees (Vasiliauskas, 2001), leading to unaccounted for belowground mortality. The best way to improve the certainty of the carbon budgets would be to do a more complete belowground carbon budget including measurement of litter inputs and soil respiration or by doing a ^{14}C labeling study, as was done for the control site (Gaudinski *et al*., 2000).

4.5 Carbon sequestration in wood products

The estimated contribution of wood products to the carbon dynamics of the harvest is highly dependent on which details are considered in the calculations. Because so much firewood was removed from the site, consideration of the fossil fuel offset (5.0 MgC/ha) for this wood has a large effect on the overall carbon balance. If a larger proportion of the wood from this harvest had been sold as sawtimber, whether or not emissions from fossil fuels used in harvesting and processing wood products was considered would have been very important, as this flux was equal to nearly half (46%)

of the carbon stored in wood products. Other accounting details, such as fossil fuel offsets from using wood in place of materials such as steel and concrete, which require more energy for production (Werner *et al.*, 2005; Gustavsson and Sathre, 2006), were not considered in this study could further alter the conclusions made about carbon storage in wood products.

Under the assumptions used in this study, 2.0 MgC/ha were stored in wood products from removed sawtimber after fossil fuel emissions in harvesting and processing were accounted for. Though nearly all of this carbon remained in use or in landfills over the time period considered in this study, wood products are not a permanent carbon sink. As they are disposed, a significant portion of their carbon will be returned to the atmosphere as landfill gases over the next 100-200 years (Figure 3.16). If landfill gas capture and combustion are not used, this is more damaging from a climate change perspective than simply burning the wood because of the global warming potential of the methane released.

4.6 Source-sink dynamics of the harvest site

The carbon budgets for the harvest and control sites indicate that in the first five years, harvesting does not increase carbon uptake. Even after considering wood products as a sink and applying fossil fuel offsets, uptake on the harvest site will have to be greater than uptake on the control site by an average of 0.4 MgC ha^{-1} yr^{-1} over the next 20 years for the harvest site to be a carbon sink of a magnitude similar to the control site on a 25-year time scale. The future carbon dynamics of the site will depend on the balance between uptake in new growth as the stand transitions from early- to mid-successional and CWD respiration as the inputs from the harvest move into more decayed classes. While the harvest site was a smaller net carbon sink than the control site cumulatively since the harvest, the site recovered quickly to a net annual carbon flux that was similar to the control site just three years after the harvest.

4.7 Management implications

Though the data from this study do not suggest that New England forests should be selectively logged with the goal of increasing carbon sequestration, they do show that these forests are resilient and, with proper management, could provide a sustainable source of wood products for the Northeast without seriously compromising carbon uptake. Currently, only 2% of the wood and paper products used in Massachusetts are produced locally, while the majority of Massachusetts forests are not harvested to their sustainable yield. Increased local production could alleviate logging pressure in other parts of the country and the world, decrease fossil fuel emissions used in transporting wood, and provide local economic opportunities, especially in small towns (Berlik *et al.,* 2002).

Because carbon uptake rates in mid-successional New England hardwood forests do not appear to decline with stand age (Peterson, 2005; Urbanski *et al.*, in press), the carbon sequestration potential of managed forests could probably be improved by increasing the amount of time between harvests. This allows the forest to grow and store carbon, offsetting short-term losses associated with harvesting. It is also possible that carbon sequestration in managed forests could be increased through changes to the species selection of harvests. Red oak and white pine, which are highly productive as large canopy trees, are preferentially harvested because of their economic value (Kittredge *et al.*, 2003). This approach to harvesting selects for the less-frequently cut species (Oliver and Larson, 1996), in this case, less productive species such as red maple, birch, and hemlock. Using wood in place of other construction materials such as steel and concrete has been suggested as a way that New England forest management could contribute to emissions reduction (D. Foster, personal communication), though the potential release of carbon contained in the wood to the atmosphere after disposal and emissions in processing must be considered. Forest management with this goal must be done with a well-designed management plan that is careful to avoid further increasing the preferential harvest of red oak and white pine, as these species are valuable as construction material.

Increasing time between harvests and cutting less productive species would decrease short-term profit from forest harvesting and increase the value of future harvests, making these difficult strategies to encourage because most landowners in Massachusetts are not in a position to benefit from trading short-term profits for long-term benefit. Most private forest owners only own their land for about 20 years (Foster *et al.*, 2005), and development is encroaching on many forested areas (McDonald *et al.,* 2006), giving landowners the option of selling land to developers in the near future. In this situation, creating an economic incentive for management plans that increase carbon sequestration, such as incorporation of forest offsets in a cap-and-trade program, may be the best way to encourage the participation of land owners.

Sampson (2004) outlined several models for incorporating carbon sequestration from forestry and agriculture into cap-and-trade programs. Of these, the "business as usual" (BAU) model seems to be the best structure for incorporating forestry practices in a landscape where the forests will take up carbon in the absence of management. Under the BAU model, a baseline is established as the carbon uptake that would occur under normal growth conditions. Offsets or emissions are then credited or debited as the change from this baseline. Modifying a harvest plan to minimize the reduction of carbon uptake could then be an economically viable option, even if it decreased the profit from the harvested wood, because it would reduce the emissions credited to the land owner.

More study of the effects of specific management techniques on carbon uptake in different types of forests would be needed before a program like this could be implemented. While this could be a useful tool in influencing land owners, it shouldn't be the only oversight on forest harvesting. Carbon sequestration is just one of a group of goals for forest management that address concerns from the local to the global scale.

5. Conclusions

Five years after selective logging, the harvest site is a small cumulative net carbon sink. If the use of wood removed in the harvest is considered, the system is a carbon sink in relation to the atmosphere but stored much less carbon over the five year period than the control site. The future carbon dynamics of the harvest site will depend strongly on the trajectories of growth and woody debris respiration, and biometric measurement of these pools should continue to investigate this. The overall certainty of these conclusions would be improved by a more careful treatment of the belowground carbon budget. The data from this study do not suggest that more intensive forest management should be initiated in New England with the goal of increasing carbon sequestration unless further measurement of this site shows an increase in uptake over the next 5-10 years. However, carbon uptake rates on the harvest site recovered quickly after the harvest, suggesting that these forests are resilient and could provide a sustainable source of wood products as part of a management plan that considers carbon sequestration among a broad spectrum of goals for forest management in New England.

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Appendix A Allometric equations for calculating the weight (wt) of live trees and whole tree snags from DBH or basal diameter (BD). Weight is in pounds and diameter is in inches unless otherwise noted.

Species	Equation	Source
American Beech (*Fagus grandifolia*)	$\ln(wt) = 1.3303 + 2.2988 \ln(DBH)$	Young et al. 1980
Black Birch (*Betula lenta*)	$wt = 1.6542 (DBH)^{2.6606}$	Brenneman et al.1978
Black Cherry (*Prunus serotina*)	$wt = 1.8082 (DBH)^{2.6174}$	Brenneman et al.1978
Gray Birch (*Betula populifolia*)	$\ln(wt) = 1.0931 + 2.3146 \ln(DBH)$	Young et al. 1980
Highbush Blueberry* (*Vaccinium corymbosum*)	$wt = 95.143 (BD)^{3.706}$ Based on wt in g and BD in cm	Telfer 1969
Eastern Hemlock (*Tsuga canadensis*)	$\ln(wt) = 0.6803 + 2.3617 \ln(DBH)$	Young et al. 1980
Eastern White Pine (*Pinus strobus*)	$\ln(wt) = 0.4080 + 2.4490 \ln(DBH)$	Young et al. 1980
Hawthorn (*Crataegus spp.*)	$\ln(wt) = -2.48 + 2.4835 \ln(DBH)$ Based on wt in kg and DBH in cm	Jenkins et al. 2003
Northern Red Oak (*Quercus rubra*)	$wt = 2.4601 (DBH)^{2.4572}$	Brenneman et al.1978
Northern Wild Raisin** (*Viburnum cassinoides*)	$wt = 29.615 (BD)^{3.243}$ Based on wt in g and BD in cm	Telfer 1969
Red Maple (*Acer rubrum*)	$\ln(wt) = 0.9392 + 2.3804 \ln(DBH)$	Young et al. 1980
Red Pine (*Pinus resinosa*)	$\ln(wt) = 0.7157 + 2.3865 \ln(DBH)$	Young et al. 1980
Spirea (*Spirea spp.*)	$wt = 36.648 (BD)^{2.579}$ Based on wt in g and BD in cm	Telfer 1969

White Ash (*Fraxinus americana*)	$wt = 2.3626\ (DBH)^{2.4798}$	Brenneman et al.1978
White/Paper Birch (*Betula papyrifera*)	$\ln(wt) = 0.4792 + 2.6634 \ln(DBH)$	Young et al. 1980
White Oak (*Quercus alba*)	$wt = 1.5647\ (DBH)^{2.6887}$	Brenneman et al.1978
White Spruce† (*Picea glauca*)	$\ln(wt) = 0.8079 + 2.3316 \ln(DBH)$	Young et al. 1980
Witch Hazel (*Hamamelis virginianaia*)	$[0.001*10^{(-3.037 + 2.900 \log10(BD*10))}] - [0.001*10^{-2.0729 + 2.162 \log10(BD*10))}]$	Telfer 1969
Yellow Birch (*Betula alleghaniensis*)	$\ln(wt) = 1.1297 + 2.3376 \ln(DBH)$	Young et al. 1980

*Equation calculated for *Vaccinium spp.*
**Equation calculated for *Viburnum lantanoides*
† Equation calculated for *Picea spp.*

Equation for northern red oak also used for black oak (*Quercus velutina*) and American chestnut (*Castanea dentata*). Equation for red maple also used for striped maple (*Acer pensylvanicum*).

Appendix B WD measurement and calculation information

B.1 CWD Classifications (Harmon and Sexton, 1996)

Classification	Description
Log	horizontal (< 45° off the ground), ≥ 7.5 cm in diameter at larger end, ≥ 1 m long
Log Snag	standing (≥ 45° off the ground) bole, ≥ 7.5 cm in diameter at the base, ≥ 1 m in height
Stump	standing (≥ 45° off the ground) bole, ≥ 7.5 cm in diameter at the base, ≤ 1 m in height
Whole Tree Snag	standing (≥ 45° off the ground) bole, ≥ 7.5 cm in diameter at the base, retains the structure of a live tree, twigs and branches still intact

B.2 Volume Equations (Harmon and Sexton, 1996)

Logs (Newton's Formula): $V = L(A_b + 4A_m + A_t)/6$

Log Snags (Frustrum of a Cone): $V = L(A_b + \mathrm{sqrt}(A_bA_t) + A_t)/3$

Stumps (Frustrum of a Cone): $V = L(A_b + \mathrm{sqrt}(A_bA_t) + A_t)/3$

Where: V = volume in m^3 L = length in m A_b = Cross-sectional area at the larger end/base in m^2

A_m = Cross-sectional area at the mid-point in m^2 A_t = Cross-sectional area at the smaller end/top in m^2

FWD Volume per Area: $V = A_c / (8\ L)$

Where: V = volume per area in m^3/m^2 A_c = Cross-sectional area of FWD on transect segment L = transect segment length in m

B.3 Physical Descriptions of WD Decay Classes (Harmon and Sexton, 1996)

Decay Class	Description
I	Solid wood, recently fallen, bark and twigs present.
II	Solid wood, bark and branches still present.

III	Wood not solid, some sloughing of sapwood, but nail must still be pounded into log.
IV	Sapwood sloughing, nail may be forcibly pushed into log.
V	Heartwood friable, barely holding shape, nail may be easily pushed into log.

B.4 WD Density Values (Liu *et al.,* 2006)

Decay Class	Bulk Density (g/cm^3)
I	0.47
II	0.41
III	0.31
IV – V	0.23

Appendix C Parameters and conversion factors used in wood products accounting.

C.1 Conversion factors

Input	Value	Source
Gasoline consumption, chainsaw	0.275 l/m^3	Raymer (2006)
Diesel consumption, harvester	1.1 l/m^3	Raymer (2006)
Diesel consumption, tractor	2.0 l/m^3	Raymer (2006)
Diesel consumption, heavy transport	0.45 l/km	Raymer (2006)
Maximum load, heavy transport	27,500 kg	Raymer (2006)
Distance from logging site to sawmill	13.3 km	
Green density, Birch	0.90 g/cm^3	Liu (2006)
Green density, Oak	0.93 g/cm^3	Liu (2006)
Green density, Maple	0.88 g/cm^3	Liu (2006)
Green density, Hemlock	0.70 g/cm^3	Liu (2006)
Green density, All species	0.90 g/cm^3	Liu (2006)
Wood density, Red Oak	0.56 g/cm^3	Jenkins et al. (2003)
Wood density, White Oak	0.60 g/cm^3	Jenkins et al. (2003)
Wood density, Black Oak	0.56 g/cm^3	Jenkins et al. (2003)
Wood density, White Ash	0.55 g/cm^3	Jenkins et al. (2003)
Wood density, Hemlock	0.38 g/cm^3	Jenkins et al. (2003)
Wood density, White Pine	0.34 g/cm^3	Jenkins et al. (2003)
Wood density, Black Birch	0.60 g/cm^3	Jenkins et al. (2003)
Wood density, White Birch	0.48 g/cm^3	Jenkins et al. (2003)
Wood density, Beech	0.56 g/cm^3	Jenkins et al. (2003)
Wood density, Red Maple	0.49 g/cm^3	Jenkins et al. (2003)
Weighted average biomass, firewood	772 kgC/cord	Kuhns and Schmidt (2003)
Efficiency, oil-fueled central heating	80%	DOE (2005)
Efficiency, catalytic wood stove	70%	DOE (2005)
Emissions, fuel oil (No. 2)	21.0 kgC/million BTU	Aubé (2001)
Emissions, diesel	0.746 kgC/l	CORRIM (2001)
Emissions, gasoline	0.643 kgC/l	CORRIM (2001)
Heat content, Beech	32300 BTU/kgC	Kuhns and Schmidt (2003)
Heat content, Birch	30600 BTU/kgC	Kuhns and Schmidt (2003)
Heat content, Hemlock	31500 BTU/kgC	Kuhns and Schmidt (2003)
Heat content, Maple	30500 BTU/kgC	Kuhns and Schmidt (2003)
Heat content, Red/Black Oak	30700 BTU/kgC	Kuhns and Schmidt (2003)
Heat content, White Ash	30700 BTU/kgC	Kuhns and Schmidt (2003)
Heat content, White Oak	30500 BTU/kgC	Kuhns and Schmidt (2003)
Heat content, White Pine	31100 BTU/kgC	Kuhns and Schmidt (2003)
Heat content, weighted average	31000 BTU/kgC	
Carbon content, paper	44%	Kubeczko (2001)
Energy use, paper production	38,600 BTU/kg paper	EPA (2007)
Production-related emissions (for 1 kgC in sawtimber)		
Construction beams	0.255 kgC	Werner et al. (2005)
Floor joists	0.483 kgC	Werner et al. (2005)

Laminated timber board	0.134 kgC	Werner et al. (2005)
Rough panels and supports	0.0214 kgC	Werner et al. (2005)
Furniture	0.603 kgC	Werner et al. (2005)
Recycling rate, paper products	50.0%	EPA (2005)
Recycling rate, wood	9.4%	EPA (2005)
Percent of MSW burned for energy	13.8%	EPA (2005)

C.2 HARVCARB equations for product retirement from end-use pools (Row and Phelps 1996)

Original expected median life in use (H) is adjusted to account for the proportion of each use recycled (R) to give adjusted expected median life in use (L):

$$L = H / (1 - R)$$

The equation used to find the pattern of product retirement (P) depends on the time in the sink (T). When T exceeds L:

$$P = 0.5 / [1 + 2 (\ln(T) - \ln(L))]$$

When T is between ½ L and L, P is interpolated between 0.5 and:

$$P = 1 - 0.5 / (1 + 2 \ln(L))$$

When T is less than ½ L, P is interpolated between 1 and P at T = ½ L.

C.3 Statistics for the Martone Landfill in Barre, MA, used as inputs for LandGEM v3.02 (DEP, 2006).

Open Date 1968
Scheduled Close Date 12/31/2010

Annual Acceptance (Tons)

Year	Acceptance
1990	62400
1991	62400
1992	93600
1993	93600
1994	97499
1995	96393
1996	94141
1997	48904

Year	Acceptance
1998	53686
1999	96575
2000	97471
2001	93578
2002	93390
2003	93522
2004	93534
2005	93557