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Changes in Carbon Storage and Net Carbon Exchange One Year After an Initial Shelterwood Harvest at Howland Forest, ME

NEAL A. SCOTT*
Woods Hole Research Center
P.O. Box 296 Woods Hole
Massachusetts 02536, USA

CHARLES A. RODRIGUES
Department of Plant, Soil, and Environmental Science
University of Maine
Orono, Maine 04469, USA

HOLLY HUGHES
Woods Hole Research Center
P.O. Box 296
Woods Hole, Massachusetts 02536, USA

JOHN T. LEE
Department of Plant Soil, and Environmental Science
University of Maine
Orono, Maine 04469, USA

ERIC A. DAVIDSON
Woods Hole Research Center
P.O. Box 296
Woods Hole, Massachusetts 02536, USA

D. BRYAN DAIL
Department of Plant, Soil, and Environmental Science
University of Maine
Orono, Maine 04469, USA

PHIL MALERBA
International Paper
Augusta, Maine 04330, USA

DAVID Y. HOLLINGER
USDA Forest Service
Northeastern Research Station
Durham, New Hampshire 03824, USA

ABSTRACT / Although many forests are actively sequestering carbon, little research has examined the direct effects of forest management practices on carbon sequestration. At the Howland Forest in Maine, USA, we are using eddy covariance and biometric techniques to evaluate changes in carbon storage following a shelterwood cut that removed just under 30% of aboveground biomass. Prior to harvest, the stand contained about 76 Mg C/ha (30 m^2/ha basal area) in aboveground and belowground live biomass. Harvesting removed about 15 Mg C/ha (SEM = 2.1) and created about 5.3 Mg C/ha (SEM = 1.1) of aboveground and 5.2 Mg C/ha (SEM = 0.7) of root/stump detritus. Leaf-area index (LAI) and litterfall declined by about 40% with harvest. Approximately half of the harvested wood was used for paper products and half for longer-lived wood products. Eddy covariance measurements in a nearby unharvested stand indicated that net ecosystem exchange (NEE) averages about 1.8 Mg C/ha/year of C sequestration. A comparison of NEE at unharvested and harvested stands, both preharvest and postharvest, indicated that NEE declined following the harvest by about 18%, which is less than expected based on basal area and LAI changes. Soil respiration declined slightly (but nonsignificantly, $P = 0.23$) with harvest, suggesting no major soil C loss after harvest. When decay of paper and wood products is included in a preliminary carbon budget, we calculate a postharvest net source of C to the atmosphere for at least 5 years, assuming preharvest growth rates of trees. How quickly the carbon balance becomes positive will depend largely on whether postharvest growth rates increase.

Forests store carbon (C) as they accumulate biomass. The C stored in the vegetation of the world is nearly equivalent to the amount present in the atmosphere as CO_2, and most of the C stored in vegetation is in the woody biomass of forests (Dixon and others 1994; Schlesinger 1991). In addition to storing C, however, many forests are also commercial sources of timber and wood fiber. In most C accounting methodologies, forest harvesting leads to a net transfer of C from the terrestrial biosphere to the atmosphere (Dixon and others 1994; Harmon and others 1990; Houghton and others 1999; Houghton and Hackler 2000). As countries search for methods to help control or mitigate increasing CO_2 emissions, it is critical to understand whether commercial use of forests could be managed to sustain

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*Author to whom correspondence should be addressed, *email:* nscott@whrc.org

or enhance terrestrial C sequestration, rather than cause net release of C to the atmosphere. Can forest management practices be developed that will meet the multiple goals of providing wood and paper products, creating economic returns from natural resources and also sequestering C from the atmosphere?

Terrestrial ecosystems in the northern hemisphere appear to be significant carbon sinks (Gurney and others 2002), and eddy covariance measurements of net C exchange at key forest sites suggest that midlatitude forests in the northeastern United States are accumulating carbon (e.g., Wofsy and others 1993; Hollinger and others 1999). Much of this C accumulates in the live vegetation (Barford and others 2001; Curtis and others 2002). However, none of these research forests are being actively managed for timber production, whereas many forests (nationally and globally) are currently undergoing some form of management. In the United States, about two- thirds of the $\sim 3 \times 10^8$-ha forest resource is classified as "commercial forest" (UN/ECE 1999; USDA-Forest Service 1982) and is potentially subject to harvesting. Management options, including rotation length, thinning, and harvesting intensity, can have a significant impact on net C sequestration in a forest (Hoen and Solberg 1994; Winjum and others 1998; IPCC 2000). In the state of Maine, USA, where our research site is located, about 540,000 acres of forest (3.2% of timberland, 2.7% of the total Maine land area) underwent some form of harvest in 1999 (Maine Forest Service 2000). To understand how forests contribute to regional and global carbon budgets and to the sustainability of current carbon sinks, it is critical to understand how forest management influences C budgets over a range of temporal scales.

In the state of Maine, USA, the use of the shelterwood system has increased dramatically in the last decade. In this commercial harvest system, a portion of the forest basal area is removed several times toward the end of one rotation. Two or three harvests are commonly spaced 5–15 years apart. The shelterwood system encourages natural regeneration by opening up the forest canopy and increasing light penetration to the forest floor. At the end of the rotation, the remaining overstory is removed in the final shelterwood cut, releasing preestablished natural regeneration. Between 1996 and 1999 in Maine, areas of forest harvested with shelterwood cuts increased by over 50%. In 1999, shelterwood cuts were used on 27% of the total harvested Maine forest area (Maine Forest Service 2000). At the Howland Integrated Forest Study (HIFS) Area in Maine, USA, owned and managed by International Paper (IP), shelterwood cuts are used as part of even-aged softwood management strategies. Shelterwood cuts can influence forest C budgets in several ways. After the first cut, growth rates of remaining overstory trees are likely to increase due to reduced resource competition. Available nitrogen (N) pools often increase following harvest (Likens and others 1970); this could stimulate the growth rates of remaining trees following a shelterwood cut. On the other hand, decomposition of slash left on the site from the harvest operation and combustion of wastes created during timber processing will release C to the atmosphere. Wood products (if they are included in the calculation) will persist for years to decades, ultimately releasing C to the atmosphere as they are burned or decay. Changes in soil temperature and moisture will also likely change following a shelterwood cut, altering the rates of soil organic matter decomposition. Dead roots and stumps will also decompose, releasing C to the atmosphere, although some root detritus may become incorporated into soil organic matter. The temporal dynamics of these processes determines whether, and for how long, a forest will continue to sequester C, as well as the rate of C sequestration.

One of the overall objectives of our research at Howland Forest is to evaluate the net C consequences of shelterwood management. We are doing this by measuring changes in C stocks resulting from harvest and then using eddy covariance measurements to quantify the changes in net ecosystem C exchange in both a harvested and a "control" stand. The objective of the work we report on here is to quantify changes in C stocks resulting from an initial shelterwood harvest, explore how these changes in stocks might alter future net ecosystem C storage, and document changes in net ecosystem C exchange 1 year after the initial shelterwood harvest.

Materials and Methods

Site Description

The Howland Forest research site is located about 35 miles north of Bangor, Maine, USA (45°12'N, 68°44'E, 80 masl). The forest is owned, and actively managed, by International Paper, Ltd. Stands in this forest consist primarily of red spruce (*Picea rubens* Sarg.) and eastern hemlock [*Tsuga canadensis* (L.) Carr.], with lesser quantities of other conifers [primarily balsam fir, *Abies balsamea* (L.) Mill., white pine, *Pinus strobus* L., and northern white cedar, *Thuja occidentalis* L.] and hardwoods (red maple, *Acer rubrum* L. and paper birch, *Betula papyrifera* Marsh.). The forest stand around our control eddy covariance tower (control stand; Zone 5 in Figure 1; see Hollinger and others 1999) has a live basal area

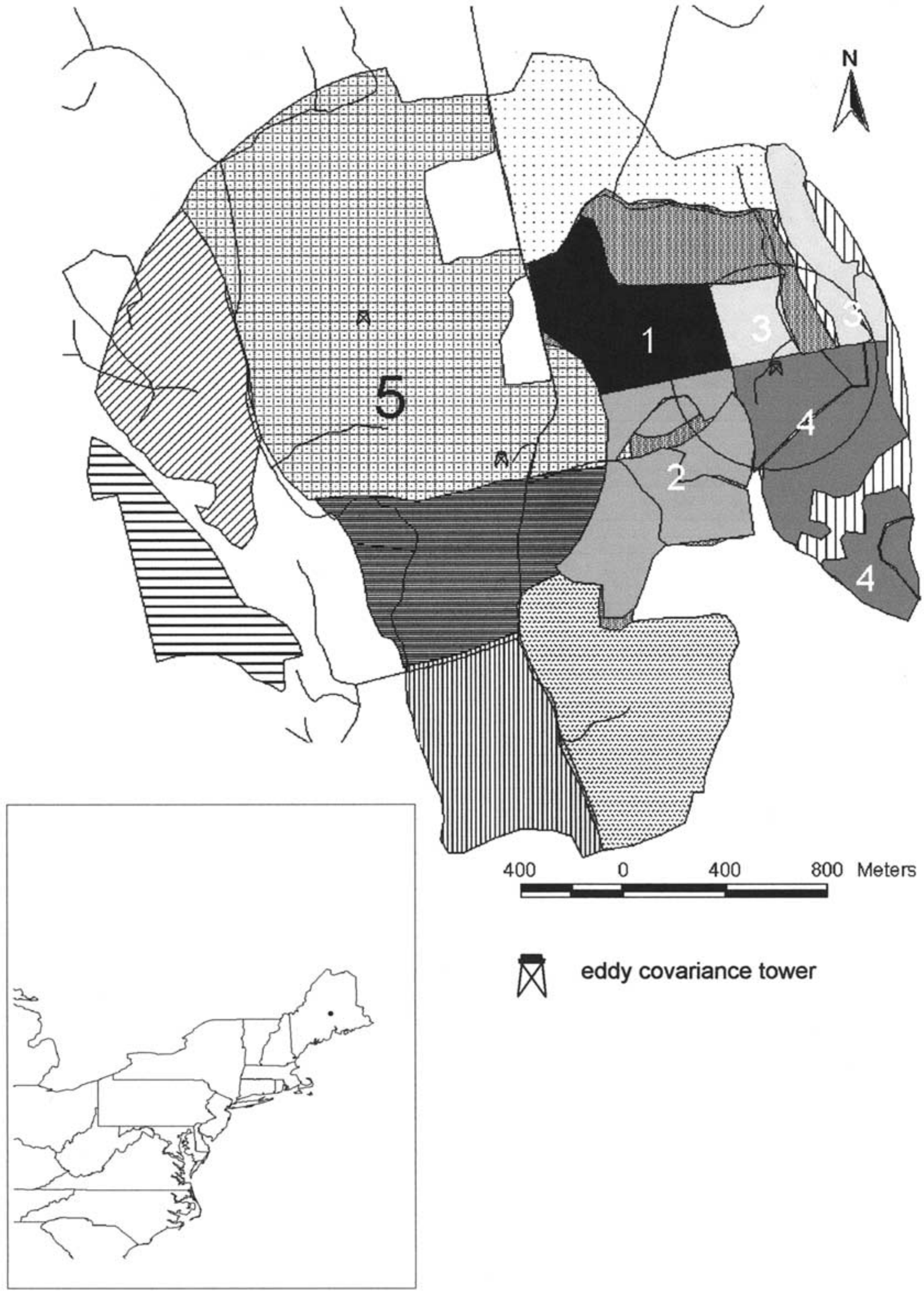

Figure 1. Map of Howland Forest research site. Polygons represent different management units, and the towers (symbol) show the location of our eddy covariance towers. The harvested area is shown as solid polygons (numbers 1–4). Circle around the tower in the harvested stand represents the area within 400 m of the tower, the source region for most of the carbon flux measurements.

of about 43 m^2/ha and leaf area index (LAI) of ~6 m^2/m^2 and is about 20 m in height. This stand was logged (not clear-cut) early in the 1900s, but has been minimally disturbed since that time. The harvested stand (Zones 1–4, Figure 1) had a preharvest live basal area of ~30 m^2/ha and leaf area index (LAI) of ~4 m^2/m^2 and is also about 20 m in height. An eddy covariance tower is located near the middle of the harvested area (Figure 1). The history of the harvested stand prior to 1960 is not known, but there is evidence of light logging activity in parts of the forest in the last 25 years. Topographically, the region varies from flat to gently rolling, with a maximum elevation change of less than 68 m within 10 km. Soils range from well drained to very poorly drained over relatively small areas (Levine and others 1994). Upland soils are fine sandy loams, classified as Aquic Haplorthods. The mean annual temperature is about +5.5°C, and the mean annual precipitation is 1000 mm. The forest stand at the control site is storing about 1.8 Mg C/ha/year (Hollinger and others 1999). See Fernandez and others (1993) for further information on Howland Forest.

The shelterwood system at HIFS involves three entries into the forest, each one 10–15 years apart, with about one-third of the basal area removed each time. Harvesting is accomplished using mechanical harvesters with wood forwarding (carriage) that avoids dragging logs on the ground, reducing soil disturbance and compaction significantly. Soil disturbance is further reduced by placing slash in tracks and then driving over the slash as the harvesting equipment moves through the forest. The goal of the shelterwood system is to promote seedling establishment and development, so that by the final harvest there is a well-developed stand of seedlings and saplings.

Preharvest Stand Measurements

Prior to harvest, we surveyed the forest to obtain preharvest estimates of total biomass (standing dead and live), down and dead wood (DDW), and leaf-area index. For biomass estimates, we located 48 circular plots (14.6 m in diameter) on transects radiating out from the eddy covariance towers (both control and harvested) every 30°; plots were located 50, 100, 200, and 400 m from the tower. This plot size is identical to the subplot size used in the current Forest Inventory and Analysis (FIA) program (Birdsey 1992; Birdsey and Heath 1995). In each plot, we marked individual trees, noted species and condition (live or dead), and measured the diameter at breast height (DBH). Diameter measurements were converted to biomass using species-specific allometric equations (Young and others 1980).

Along six transects (240°, 270°, 300°, 330°, 0°, 30°), we established plots for DDW measurements exactly 22.9 m due east of the biomass survey plots located at 100 and 200 m. These transects were selected because they represent the area most commonly upwind of the eddy covariance tower; plots were half the size of the biomass plots. In these plots, we collected all down and dead wood that was > 0.5 cm in diameter and divided that material into pieces < 5 cm in diameter and > 5 cm in diameter. All material was weighed wet in the field on a suspended spring scale (100-lb capacity) and then subsampled. Subsamples were dried at 60°C to a constant mass. For very large pieces of DDW (i.e., ones that would not fit into the plastic container used to weigh the samples), we estimated DDW volume by measuring the total length and diameter at each end of the log. Using an average diameter, we estimated the volume assuming that it was a cylinder. Subsamples of the logs were removed (intact) for density determinations in the lab. First, we measured the subsample volume by wrapping the wood in plastic film and submerging it in water to measure displacement. The sample was then dried at 60°C to a constant mass. All DDW was removed from the plots so that the same plots could be resurveyed after harvest to quantify DDW production resulting from the harvest.

Harvest Procedures

Mechanized harvesting of hardwoods and softwoods (cut to length and forwarded) in the experimental area (Zones 1–4; Figure 1) began in November 2001, starting in the southernmost part of the area (Figure 1). Harvesting in the Howland township area (Zones 1 and 3, Figure 1) started early in 2002. The southeast section of the area (Zones 2 and 4) required the construction of a winter road that was completed in January 2002. Mechanized harvesting was largely completed by the end of February. Chain-saw harvesting of larger trees, mostly pine, was carried out for 2 weeks, ending in mid-March. Based on our plot survey, the diameters of stems removed ranged from 6.9 to 45.8 cm (mean = 23.9 cm, median = 22.1 cm).

C Removals with Harvest

Throughout the harvest, the logging contractors provided information on wet mass (truck weights) of logs removed from the forest by species and the destination of each load of wood. Throughout the harvest, we subsampled logs of all species for moisture content in proportion to their abundance; a total of 108 wood samples were collected during the harvest. After 2.5–5 cm of the log end was removed, a thin (< 2.5 cm) cross section was taken from each sample log. These wood moisture samples were removed from logs harvested on the same day. Larger samples were cut into pie-shaped wedges to allow for faster drying. In the lab, wet sample weights were recorded, and the samples were dried at 60°C to a constant weight (~2–4 weeks for most samples). Moisture contents proved to be relatively constant throughout the harvest, so single moisture values were used for mixed hardwoods, spruce, hemlock, mixed spruce/fir, and white pine (42.1%, 50.3%, 51%, 50.8%, and 58.1%, respectively). We assumed 50% C to convert dry mass to C.

To estimate the size of the harvested area, we georeferenced a harvest map into a Geographic Information System (GIS) using ArcGIS 8.3 software from Environmental Research Systems Institute (ESRI). ArcGIS allows the transformation of digitized raster information to map coordinates. Each cell in the digitized map is "warped" to real-world coordinates (in this case, UTM zone 19N) using an affine transformation. The affine transformation uses at least three control points, or points in the raster image that can also be located within the spatial data of the GIS. Harvest polygons were demarcated using the ArcScan extension to ArcGIS software. The ArcScan extension allows raster features in the georeferenced harvest maps to be converted to vector features in the GIS dataset. After the harvest maps were vectorized and coregistered to other Howland GIS data, we calculated the area of each of the harvest polygons (Zones 1–4, Figure 1).

We also estimated carbon removals by resurveying our 48 biomass plots (see preharvest stand measurements) arrayed around the tower. In each plot, we noted which stems had been removed during the harvest (each stem location was mapped prior to harvest). Based on the DBH of the harvested tree measured prior to harvest, and species-specific allometric equations for stem mass (Young and others 1980), we were able to estimate C removals independently of the logging contractors' data.

Dead-Wood Production

Given the high spatial variability of postharvest slash and the potential importance of DDW decay to the postharvest forest C balance, we quantified slash production in several ways. First, we resampled the 11 DDW plots that had been cleared of material preharvest (one plot location was "lost" during harvest). As with the preharvest measurements, DDW was divided into two categories: < 5 cm and > 5 cm in diameter. We did not separate foliage detritus produced during the harvest from the other < 5-cm fraction material. Second, we used the resurvey data from 48 plots (see previous section), along with species-specific allometric equations (Young and others 1980) to estimate branch, foliage, and stump/coarse root detritus produced during the harvest.

During the harvest, almost all of the slash was placed in logging tracks (~4 m wide) through the forest to reduce soil disturbance by the harvesting equipment. We were concerned that a small number of systematically located plots (11) might not adequately measure this nonrandomly distributed slash. We therefore established six plots in logging tracks along transects at 270° and 300° from the eddy flux tower. On each transect, rectangular plots (16.7 m^2) were placed on the track located closest to a point 75, 150, and 300 m from the tower. Slash produced from the harvest was removed from the plots and sorted into three size classes: < 1 cm (foliage and fine branches), 1–5 cm, and > 5 cm in diameter. All DDW was weighed using a hanging scale in the field and subsampled for moisture determination. Subsamples were dried at 60°C to a constant mass. We scaled up these estimates of slash biomass using estimates of total area covered by the logging tracks (based on track width and spacing).

Leaf-Area Index and Litterfall

With expected removal of about 30% of aboveground biomass at harvest, we expected a proportional decrease in LAI. This is important, as LAI is a key factor influencing canopy C uptake. We measured the preharvest LAI with a LICOR LAI-2000 (LI-COR Corp., Lincoln, Nebraska, USA) along transects radiating out at 30° intervals from each tower (similar to those used for biomass). LAI measurements were made using the remote above/below mode, where one unit is placed out in the open (no canopy), and the other unit is used beneath the canopy. The two units were synchronized to obtain measurements at the same time under identical sky conditions. In the forest, LAI readings were made by holding the sensor level about 2 m above the ground. The 2001 measurements were made in August along seven transects (NNE to SSW). Measurements were made at 50-m intervals, with 10 measurements collected along each transect (total of 70 measurements). Postharvest measurements were collected mid-September to early October (before leaf fall) in 2002.

We collected litterfall in the harvested stand by placing 10 litter traps (0.14 m^2) randomly in 2 areas: one south and the other west–southwest of the tower. Traps were spaced at least 5 m apart and were emptied six times at ~2-week intervals during the summer of 2002. In September and October, collections were less frequent (every 3 weeks). In 2001 (preharvest), traps were deployed after September 27. Traps were subsequently emptied weekly during October 2001, and then once in mid-November. Preharvest and postharvest comparisons of litterfall were, therefore, made for the month of October only.

Soil Respiration

We measured soil respiration rates at two sites (same sites as the litterfall measurements) in each of the control and harvested stands. At each site (~15 m^2 per site), eight polyvinyl chloride (PVC) rings (10 cm tall by 25 cm in diameter) were driven about 1 cm deep into the mineral soil. On each sampling date, soil respiration was measured by placing vented chambers (10 cm tall) over the rings for 5 min and circulating chamber air (0.5 L/min) to a LiCor 6252 infrared gas analyzer (LiCor Corp., Lincoln, Nebraska, USA). Detailed information on sampling protocols and flux calculations are provided by Savage and Davidson (2001, 2003). Measurements were made weekly during the summer and less frequently during the autumn and spring.

Eddy Covariance Measurements of Net Ecosystem CO_2 Exchange

Methods and theory for measuring net ecosystem CO_2 exchange (NEE) via eddy covariance have been described previously (e.g., Hollinger and others 1999). Briefly, continuous flux measurements have been made since 1996 at the control site on a 30-m walk-up tower. The flux system consists of a LiCor LI-6262 closed path infrared gas analyzer and an ATI model K sonic anemometer (Applied Technologies Inc., Longmont, Colorado, USA). Data are recorded at 5 Hz on personal computers using a variant of the flux program originally developed by McMillen (1988). Measurements of NEE during a calm night are problematic, so data collected at night when turbulence is low (below a friction velocity or u^* threshold value of 0.2) are rejected. For these nights, nightime NEE values are estimated from a simple model (Lloyd and Taylor 1994) based on air temperature.

Similar methods are used to measure NEE at the harvest site, but some changes in instrumentation were required because we did not have power at the site. We used a LI-COR Li- 7500 open-path infrared gas analyzer and Campbell CSAT sonic anemometer (Campbell Scientific Inc., Logan, Utah, USA) mounted on a crank-up instrument elevator (instruments can be serviced and leveled on the ground). Power for the system comes from 320 W of solar panel feeding into a charger and 500 A-hr of deep-cycle battery storage. All raw data were transmitted by radiofrequency (rf) modem back to our control tower and archived on a computer.

Unfortunately, this radio-telemetry link was compromised by precipitation events. This, and occasional power system problems, resulted in a lower return of data than we anticipated during the preharvest calibration year. Fluxes were calculated according to standard methods (e.g., Hollinger and others 1999) with density and other corrections (Webb and others 1980; Massman and Lee, 2002).

Statistical Analysis

Soil respiration rates were compared between the harvested and control areas for 2001 and 2002 growing seasons using repeated measures analysis of variance; treatment-by-year interactions were used to examine harvest effects on soil respiration. To examine changes in net C exchange for the control and harvested stands preharvest and postharvest via eddy covariance, we compared fluxes from both towers in 2000 and 2001 using Reduced Major Axis (RMA) regression analysis (Hammer and others 2001); we compared slopes (Sokol and Rohlf 1981) of the regression relationship for each year to examine changes due to harvest. We used RMA analysis because there are errors in both the dependent and independent variables (harvest flux and control flux, respectively).

Results

Preharvest C Pools

In the harvested stand, total biomass (live and dead, above- and belowground) was about 81 Mg C/ha (SEM = 4.8) prior to harvest and dominated by hemlock (about 45%), spruce (20%), and red maple (15%) (Table 1). The stand basal area (total) was 31.6 m^2/ha; the hemlock basal area was two-fold higher than spruce. These results are lower than values obtained around our control tower (120 Mg C/ha; 48 m^2/ha basal area), which is dominated (slightly) by spruce; live biomass in the control stand was about 90% of total (Table 1). Total C storage in soils to 1 m depth is about 110 Mg C/ha (Fernandez and others 1993). Standing-dead biomass in the harvested stand was $<$ 5% of total biomass (Table 1), and DDW, estimated from 11 plots, was 16.1 Mg C/ha (SEM = 3.9) (Table 1). Over 90% of the DDW was in pieces larger than 5 cm in diameter.

Harvest C Removals

Based on the wood mass trucked from the forest and wood moisture content, we estimate that harvesting of the entire area (~152 ha) removed about 19 Mg C/ha. Based on resurveying 45 biomass plots and use of species-specific allometric equations for stem mass, we estimated timber removals of 13.0 Mg C/ha (SEM = 1.7 Mg C/ha) by the shelterwood cuts. On a mass basis,

Table 1. Carbon pools in both the harvested and control stand preharvest, and changes in carbon pools associated with the harvest activity

Carbon pool	Control stand	Harvested stand (preharvest)	Harvest carbon fluxes
Live basal area (m^2/ha)	43 (2.4)	30 (1.7)	—
Live biomass (Mg C/ha)	109 (6.6)	77.3 (4.7)	—
Standing dead (Mg C/ha)	10.8 (1.2)	3.3 (0.8)	—
Down-dead (Mg C/ha)	4.1[a]	16.1 (3.9)	—
Soil[b]	110	—	—
Wood removal (Mg C/ha)[c]	—	—	14.9 (2.1)
Aboveground slash (Mg C/ha)[d]	—	—	5.3 (1.1)
Belowground slash (Mg C/ha)[e]	—	—	5.2 (0.7)

Note: Data given as mean (SEM).
[a]Data from Davidson (unpublished data).
[b]Data from Fernandez and others (1993) (to 1 m depth).
[c]Estimated based on plot resurvey, not including plots 50 m from tower (see text).
[d]Based on measurements of down-dead wood in the logging-track plots.
[e]Estimated from resurvey of 48 plots and allometric equations (Young and others 1980).

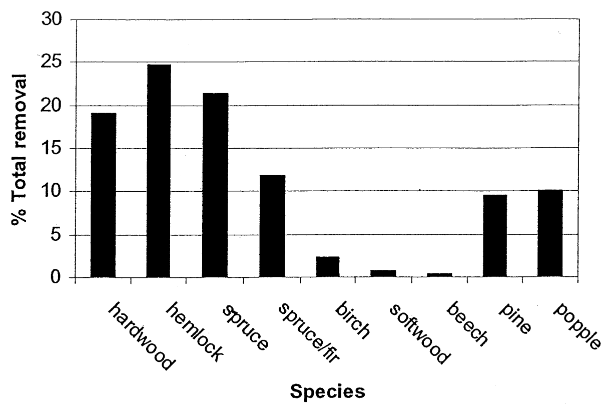

Figure 2. Biomass removals by species as a percent of the total biomass removed during shelterwood harvest.

Table 2. Wood products produced during shelterwood harvest and estimates of the half-life of (use of) these products

Product	Wet mass (tons)	% Total	Half-life (years)
Boltwood[a]	232	2	20
Chipwood[b]	364	3	3.5
Ground wood	199	2	3.5
Logs	4770	40	45
Pulp	4270	36	3.5
Stud	463	4	45
"Tree length"[c]	1510	13	3.5

[a]Short logs (Stokes and others 1989) (i.e., not cut to specified length); used for pulp or lumber.
[b]Small pieces of wood used to make pulp (Stokes and others 1989).
[c]Bole left intact after removal of nonmerchantable limbs and top (Stokes and others 1989); Can be used for either pulp or lumber, but here used primarily for pulp
Source: Skog and Nicholson (1998).

hemlock was the dominant species removed, followed by spruce and miscellaneous hardwoods (Figure 2). Removal of different species generally followed their preharvest relative abundance.

Based on our observations, we were concerned that the logging crew did not harvest as intensively around our eddy covariance tower as in the remainder of the harvest area to avoid hitting the tower. To examine this question, we compared C removals as a function of distance from the tower. Plots located 50 m from the tower had two-fold lower rates of C removals compared to plots located at 100, 200, and 400 m from the tower [7.6 Mg C/ha (SEM = 2.0) within 50 m versus 14.9 Mg C/ha (SEM = 2.1) at > 100 m]. Although analysis of variance did not indicate a significant distance effect ($P = 0.26$), we believe that the area around the tower was underharvested based on discussions with the logging contractor. We therefore believe that excluding the 50-m plots provides a better average estimate of C removals, and we calculate wood C removals from the stand as 14.9 Mg C/ha (SEM = 2.1 Mg C/ha, $n = 34$) (Table 1); the 95% confidence interval (4.3 Mg C/ha) overlaps the removal estimate based on log mass removed from the forest (19 Mg C/ha).

Wood Products

Logs were taken to 19 different sawmills over the course of the harvest. While a variety of wood products were produced (Table 2), about half of the wood was used to produce paper products, with the remainder going into longer-lived wood products (Table 2). Paper products were assigned a half-life of about 3.5 year, whereas the wood going into e.g. lumber has a much longer half-life (estimated at 45 years) (Skog and Nicholson 1998).

Down-Dead Wood Production

We quantified DDW production during the harvest in three ways. First, resampling of the 11 plots cleared of preharvest DDW suggest the harvest created about 4.8 Mg C/ha (SEM = 1.9) of DDW (all size classes). Second, postharvest-plot surveys and allometric equations (Young and others 1980) were used to calculate the biomass of branch, foliage, and coarse roots/stumps produced by the harvest. These results suggest detrital inputs of 2.9 Mg C/ha (SEM = 0.4), 2.2 Mg C/ha (SEM = 0.3), and 5.2 Mg C/ha (SEM = 0.7), respectively (Table 1). The sum of branch and foliage detritus (5.1 Mg C/ha) based on allometry of removals was very close to our estimate of slash production from cleared–plot measurements (4.8 Mg C/ha). Finally, measurements of slash in six logging-track plots gave much higher estimates of DDW per unit area (35.3 Mg C/ha; SEM = 7.4), which was expected, given that the debris were concentrated on the tracks during harvest. To compare this estimate to the others, we multiplied the logging-track DDW concentration (35.3 Mg C/ha) by the proportion of the total area (152 ha) covered by the tracks (15%, or 22.8 ha), giving an estimate of 5.3 Mg C/ha (SEM = 1.1) (Table 1). Some new DDW will likely be added to the current pool size in the future as small trees knocked over during harvest die; this will be measured in later surveys. The close agreement of these estimates is reassuring and suggests that future predictions of the contribution of DDW decay to net C storage should be accurate if decay-rate constants are accurate.

Leaf-Area Index and Litterfall

Leaf-area index was measured preharvest and postharvest on seven transects radiating out from the eddy covariance tower. Based on these data, LAI was 3.5 (0.2) m^2/m^2 prior to harvest and 2.1 (0.4) m^2/m^2 after harvest, which is about a 40% reduction. Litterfall was 854 kg/ha prior to harvest in October 2001 and was 464 kg/ha in October 2002, which is about a 46% reduction following harvest.

Soil Respiration

Based on the 2001 and 2002 growing season (July to October) data (six dates each year), soil respiration may have decreased slightly at the harvested site following the harvest in 2002 (Figure 3). Average summer soil respiration was 140 (SEM = 11) and 135 (SEM = 11) mg C m^{-2} hr^{-1} in the harvested (preharvest) and control areas, respectively, in 2001. This relative ranking reversed after the harvest in 2002, with mean summertime respiration rates of 128 (SEM = 8) and 141 (SEM = 12) mg C/m^2/hr in harvested and control areas, respectively. However, the harvest treatment-by-year interaction was not statistically significant ($P = 0.23$) in a repeated measures design; therefore, this small difference may be fortuitous. Continued measurements will reveal whether this trend becomes statistically significant in future years.

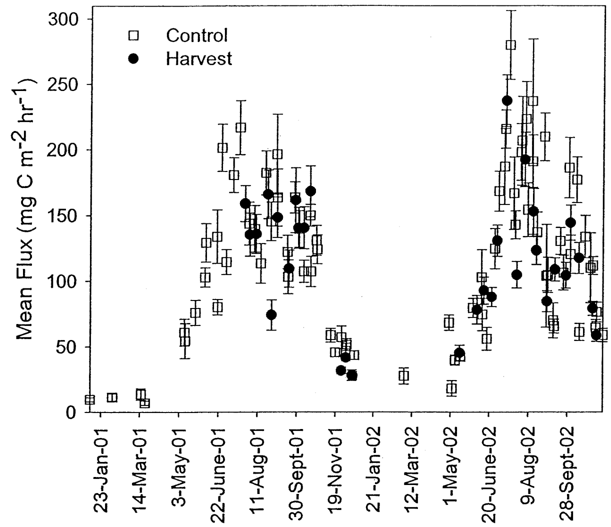

Figure 3. Soil respiration in the control and harvested stands (2001–2002). Each point represents the mean (and SEM) of 16 chambers. Preharvest is 2001 and postharvest is 2002.

Net C Exchange from Eddy Covariance

Based on mean half-hourly flux values from July to August 2002, daytime C uptake in the harvested stand was about 63% of that in the control stand (−5.9% versus −9.3 μmol/m^2/s, respectively). Nocturnal respiration was also lower (55% of control) in the harvested stand than the control over the same time period (3.0 versus 5.5 μmol/m^2/s, respectively). Integrated over the 2-month period, net ecosystem C storage was ~51 and 96 g C/m^2 in the harvested and control areas, respectively. Although we clearly saw differences in NEE in the harvested and control stands after harvest, NEE could have differed in these stands preharvest due to differences in species composition, biomass, and basal area. To determine whether the observed differences in NEE resulted from the harvest, we compared the relationship between NEE in the control and harvested stands in both 2001 and 2002 (Figure 4). Prior to harvest, half-hourly NEE values were almost identical for the two towers (Figure 4a; slope = 0.92, SE_{slope} = 0.04). After harvest, however, NEE measured at the harvest tower was lower than at the control tower (Fig-

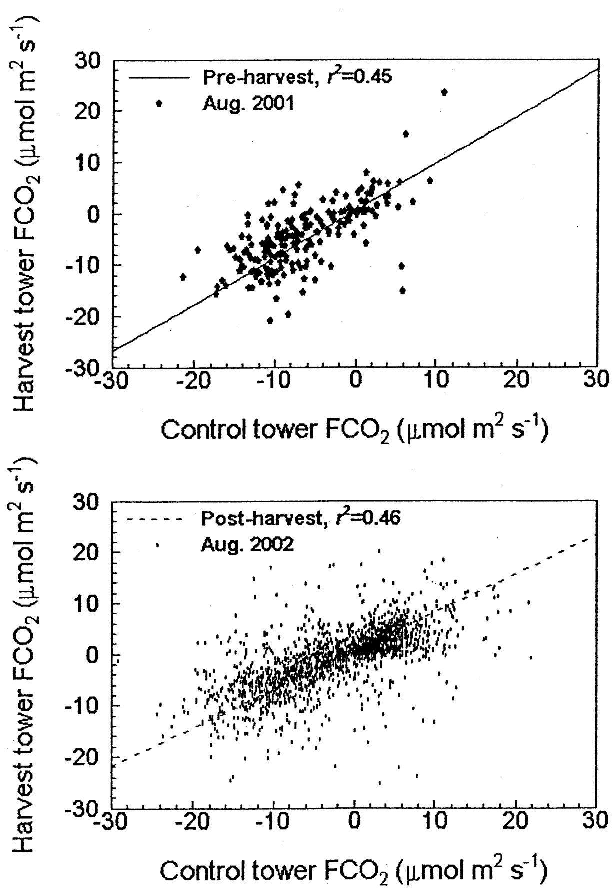

Figure 4. Comparison of half-hourly net ecosystem exchange of CO_2 by eddy covariance for identical time periods on the control tower and the harvest tower in 2001 (preharvest) and 2002 (postharvest) in August. The slope of the regression line for 2002 data (postharvest) is lower (slope = 0.75) than the slope for 2001 (preharvest; see text) (slope = 0.92).

ure 4b; slope = 0.75, SE_{slope} = 0.01); the slopes of these lines differed significantly (t_{2388} = 13.1, $P \ll 0.001$). Comparison of the slopes for the two regression lines (preharvest and post-harvest; Figure 4) suggests that NEE was significantly reduced by the harvest ($P <$ 0.001), due either to changes in C uptake, C loss via respiration, or both.

Discussion

Several studies have examined the carbon consequences of forest management, but most have focused on management that includes clear-felling, and few have measured the whole-ecosystem C consequences of different management practices. In France, Kowalski and others (2003) used eddy covariance to measure changes in NEE following clear-felling of a 50- year-old pine (*Pinus pinaster*) forest. This forest went from storing about 6 Mg C/ha/year preharvest to losing about 3 Mg C/ha/year after harvest. They observed a larger decline in C uptake (about 60%) compared to the decline in total respiration (about 30%). In boreal forests of Canada, a 1-year-old clear-cut aspen stand was also a net C source compared to a mature, intact nearby aspen stand (NEE of 1.6 g C/m^2/day versus −3.8 g C/m^2/day measured over 1 week during the growing season) (Amiro 2001). Hoen and Solberg (1994) compared the economic efficiency of carbon capture for different management practices, including clear-felling and various thinning regimes, and found that fertilization enhanced economic returns in terms of carbon capture and storage the most, but they did not evaluate the whole-ecosystem response to the different management practices. Several studies have demonstrated the carbon-storage benefits of longer rotations (e.g., Plantinga and Birdsey 1994; Boscolo and Buongiorno 1997).

Little work has examined the carbon consequences of other management options such as the shelterwood system. Our preliminary eddy covariance data from the first postharvest growing season indicates about an 18% reduction in NEE following a shelterwood cut at Howland Forest (Figure 4). This reduction in NEE is smaller than the proportion of biomass harvested (about 30%), suggesting that per unit basal area, net carbon uptake has not declined as much as expected. Stimulated tree growth following shelterwood cuts has been observed previously in northern forests (Hannah 1988). However, a longer-term database of eddy covariance measurements will be needed to see if the trend applies on an annual basis and how it varies from year to year.

Our estimated changes in net C sequestration resulting from shelterwood harvesting assume no net change in soil C storage over the 30-year period. Soil C stocks in the control stand at Howland Forest are almost as large as vegetation C (about 110 Mg C/ha), so even small changes in soil C storage resulting from harvest could influence whole-ecosystem C sequestration. At a deciduous forest site in Massachusetts, USA, radiocarbon-based estimates of soil C accumulation in a ~70-year-old forest recovering from hurricane damage are between 0.1 and 0.3 Mg C/ha/year (Gaudinski and others 2000). In the control stand at Howland Forest (Hollinger and others 1999; Savage and Davidson 2001), preliminary radiocarbon results suggest low net soil C (forest floor and mineral soils) accumulation rates of 0–0.3 Mg C/ha/year (J Gaudinski, personal communication). Whereas postharvest net ecosystem C exchange estimates (Arneth and others 1998) and simulation studies (e.g., Scott and others 2003) suggest short-term soil C losses after clear-cutting, a synthesis of results from several sites suggests that over longer time scales, forest harvesting (in this case, clear-cutting) has no net effect on soil C storage (Johnson and Curtis 2002; Johnson and others 2002), although the range of responses was large. Although our soil respiration results include both autotrophic and heterotrophic respiration, the fact that we did not see a large increase in soil respiration after harvest suggests that there was no major stimulation of soil C decomposition (i.e., no dramatic net soil C change after harvest). Based on the low net C accumulation rates at Howland Forest, reviews of empirical studies, and our soil respiration results, we believe that our initial assumption of little or no soil C changes following harvest is valid. Future work at the site will examine key processes controlling soil C dynamics following shelterwood cuts.

Differences in management history can influence the distribution of C among detritus pools. Around our long-term monitoring tower (control; Figure 1, Zone 5), DDW was only about 4 Mg C/ha (Table 1), whereas in the shelterwood-cut stands, we found about 16 Mg C/ha. In contrast, the preharvest standing-dead biomass C pool around the harvest tower was relatively small (Table 1) compared to that around the control tower, which contained about 10 Mg C/ha, or almost 10% of the total standing biomass (live and dead) C pool (Table 1). This difference in the distribution of dead wood likely reflects the fact that no logging has occurred in our control stand for about 100 years, whereas the harvested area had likely experienced some light-intensity harvesting in the past 30 years. Disturbance due to harvesting is likely to decrease the size of the standing-dead C pool while increasing the amount of C contained in DDW due to slash production and knocking over standing-dead stems.

Simulated Changes in Net C Balance

We use postharvest C pools to predict how the net C balance of the forest might change following harvest, including both decay and growth processes. Specifically, we are interested in estimating how long it might take before the system becomes a net C sink and how much growth enhancement would be required to increase C sequestration over a 30-year period. To do this, we developed a simple model to predict the net C consequences of this shelterwood harvest.

First, we developed equations to simulate the decay of both wood products and debris produced during the harvest (Figure 5, top). For wood products, we estimated decay rates of two classes of products; paper and long-lived wood products (e.g., lumber) (Table 2 and Figure 5, top). We also estimated decay rates for foliage, branches, and coarse roots/stumps (Table 3). Fine roots, which constitute about 16% of total root biomass (Vogt 1991), were estimated to be 10% of the combined root/stump mass (5.2 Mg C/ha) and given a decay half-life of 3.5 years (Berg 1984). Combined C losses were then simulated for 30 years postharvest (Figure 5, middle). Including wood products in the C budget of the forest leads to greater (over twofold) C losses during the first 10 years after harvest. To calculate changes in total ecosystem C storage, we assumed that ecosystem C storage decreased with harvest from 1.8 Mg C/ha/year (the long-term average around our control tower) in proportion to biomass removals and that it will recover to preharvest levels (linearly) in 10 years (Figure 5, bottom). In boreal forests of Canada, NEE in a partially logged and then burned forest recovered to near preburn levels in 10 years (Amiro 2001); we therefore believe that this is a reasonable assumption for Howland Forest.

Based on simulations (Figure 5, middle and bottom) that include both slash and wood products, the harvested stand is a net C source until about 5 years after harvest (Figure 5, bottom). Most of the early C losses are due to the decay of paper products and foliage detritus (Figure 5, top), each with relatively short half-lives (Tables 2 and 3). Only the decay of longer-lived timber products and stump/coarse roots contributes significantly to net C storage over longer (decadal) time scales (Figure 5, top and middle). When wood products are not included in the calculation, the harvested stand never becomes a net C source. With no harvest and assuming C accumulation at preharvest rates (1.8 Mg C/ha/year), over 30 years this forest would accumulate 54 Mg C/ha. With the harvest and offsite decay of products and assuming no stimulation of growth (Figure 5, bottom), over the 30-year period we estimate that C storage in the forest plus products will increase by 34 Mg C/ha. For net C storage in a harvested stand (including its wood products) to equal C storage in an unharvested stand at this study area over 30 years, we estimate that C accumulation in the ecosystem (mostly as tree growth) would have to increase in the harvested stand by about 40% relative to preharvest growth rates. Increased C accumulation could be caused by a release from competition in the current stock of trees or by increased abundance of faster-growing species such as white pine.

Although our simulations suggest that net C sequestration recovers quickly after a shelterwood cut, whether shelterwood cuts optimize C sequestration and wood production will require direct comparison of different management regimes for this region (IPCC 2000). Once we have verified the net C consequences of shelterwood cuts at Howland Forest, we can then compare this management regime to others (e.g., clear-

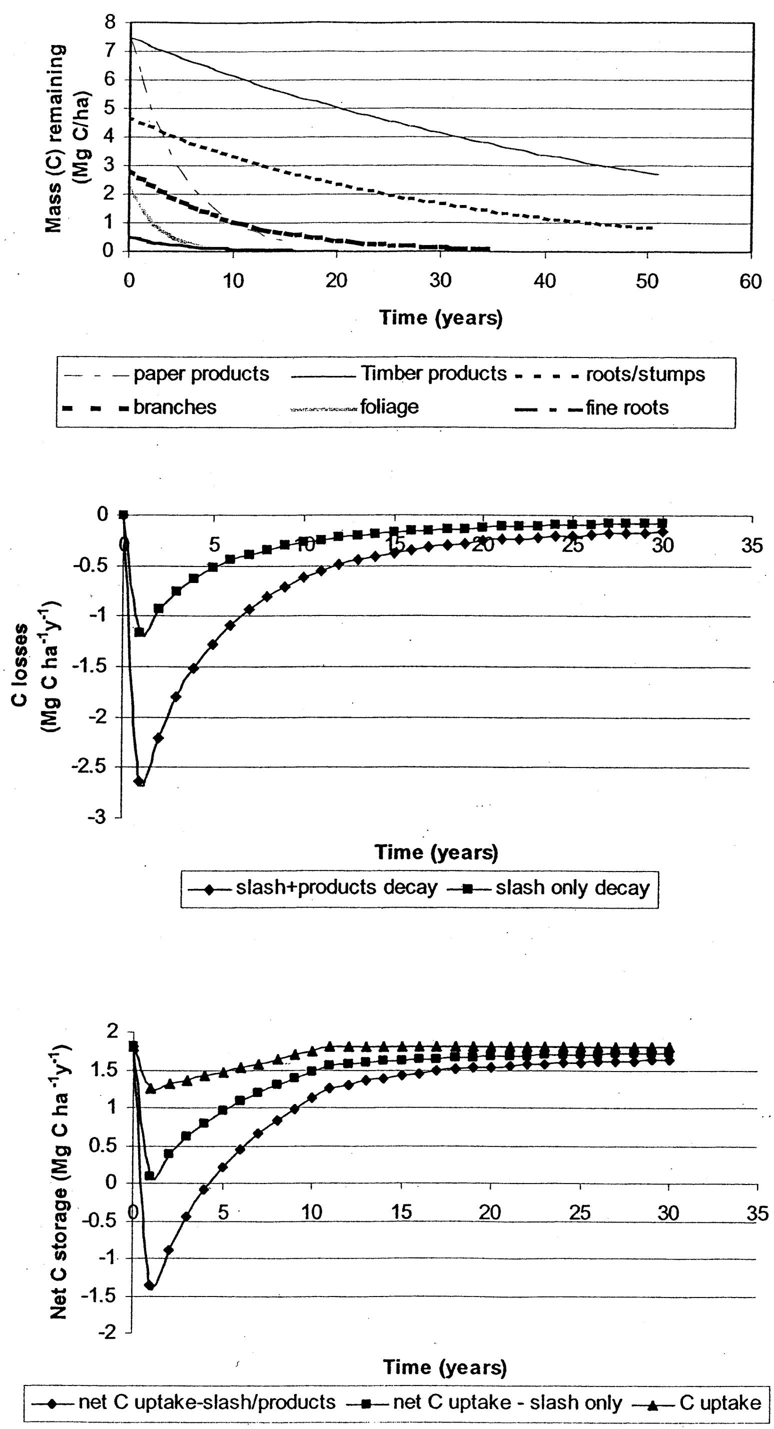

Figure 5. Simulated post-harvest C fluxes: Top: Decay of wood products and detritus produced from the harvest; middle: C losses due to wood product and slash decay; bottom: net C uptake as the difference between product and detritus decay and tree growth after harvest.

cutting) that are used in this region. This comparison will provide a rigorous assessment of whether forest management practices can be modified to optimize the use of forest resources in Maine (USA) for maximal economic and environmental benefit. It is clear that management affects carbon storage in forests, and the potential to manage carbon sequestration exists. However, analysis of changes in carbon storage due to management must include all the ecosystem carbon pools and wood products, and must be measured over adequate time scales to include the contribution of carbon pools with relatively long turnover times.

Table 3. Estimates of harvest debris pools and their half-life used to calculate C losses from heterotrophic decay of slash after the harvest

Detritus type	Pool size (Mg C/ha)	Half-life (years)	Refs.
Stump/coarse root	4.7	20	Chen and others 2001; Fahey and others 1988
Branches	2.9	7	Stone and others 1998; Perala and Alban 1982[a]
Foliage	2.2	2	Keenan and others 1996 Aber and Melillo 1980
Fine root	0.5	3.5	Berg 1984

[a]Used to derive estimates of DDW decay. Perala and Alban estimated the decay of slash (branches and foliage) to have a half-life of 3–4 years, whereas the smallest woody debris (< 20 cm) has a half-life of 10 years. Our half-life estimate of 7 years is in between these estimates.

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