"Carbon Dynamics Associated with Even-Aged Forest Management"

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INTRODUCTION

The first section of the paper (A) describes even- and uneven-aged treatments in the context of natural disturbance history by forest type in the U.S. and Canada. This section also introduces variable retention treatments, which maintain growth and reproduction of shade intolerant commercial species. We model variable retention versus clearcutting in an original study and summarize the model results in terms of comparative carbon yields in section D ("An Examination of Silvicultural Influences for the Climate Action Reserve Improved Forest Management Protocol: A Case Study of the Coastal Pacific Northwest Douglas-fir Type").

Throughout the paper, we de-emphasize the use of the terms even- and uneven-aged due to a distinct focus on variable retention two-aged treatments for reproduction of intermediate to shade-tolerant commercial species, and also due to the consideration of rotation length in addition to retention quantity in carbon accounting. The analysis in the paper focuses on the comparative carbon in the above and below ground portions of trees, due to the fact that site preparation treatments including utilization of lying dead wood, soil scarification, and prescribed burning of litter layers can be conducted at various intensities under a number of silviculture treatments.

Our literature review and modeling study combined indicate that four factors are most important in determining the net changes in forest and atmospheric carbon associated with less retention (even-aged) or greater retention (uneven-aged) stand-scale silvicultural treatments, as elaborated in section (B):

(1) *The carbon storage potential based on the pre-treatment land use and productivity has a significant influence on forest carbon.* Any harvestingtreatment will reduce carbon in stocked land versus not harvesting even accounting for in-use forest products pool due to conversion inefficiencies, with this effect particularly pronounced in forests with high initial stocking. Angiosperm stands have higher average wood density than gymnosperm stands so may be preferentially maintained in a mixed wood land scape to maximize carbon. In some but not all cases higher site productivity will translate into higher carbon.

(2) *The quantity of live tree retention significantly determines forest carbon.* Higher retention levels, particularly larger diameter trees, generally result in higher carbon stocks. However, this effect is likely forest-type dependent (see section A of literature review), with less diminishment of carbon stocks in intermediate to shade-tolerant angiosperm forest types. The modeling results of intermediate to intolerant (shade) Douglas-fir showed no impact of silvicultural retention treatment with only rotation period providing a significant difference. One possible explanation is that the Forest Visualization System (FVS) model only allowed spatially dispersed treatments, which may have limited Douglas-fir growth and regeneration compared to aggregated patterns. Aubrey et al., 2008 in the DEMO study, reported increased annual growth efficiency 5 years after dispersed harvesting due to cutting suppressed and codominant trees which was also the cutting rule for our FVS model run, but other studies in

Douglas-fir have not only reported reduced height growth and regeneration as low as 20% retention, but have also reported high retained tree mortality from skid trail damage and wind throw (Drever and Lerzman, 2001 and Newsome et al., 2010). Harmon et al. (2009) modeled Douglas-fir under STANDCARB. Nearly all treatments had asymptotes of carbon stocks from both ecosystem and products at 60-80 year rotation intervals comparable to our study. However, in aggregated spatial configurations (such as patch or group selection cuts) intermediate retention level treatments of 40-60% emerged as maximizing carbon in the 60-80 year interval, an effect we could not demonstrate, whereas in dispersed configurations all treatments were equivalent at this rotation interval, comparable to our modeling study.

(3) *The length of the rotation length (even-aged) or entry period (uneven-aged) also significantly determines forest carbon.* In the stands modeled, longer rotation lengths will generally increase forest carbon stocks at least to 100 years and potentially 200 years. This factor can be more significant than retention levels in intermediate to shade-intolerant gymnosperm and mixed forest types, while retention levels can be more significant in determining total forest carbon in intermediate to shade-tolerant angiosperm forest types. Our modeling suggests that rotation ages linked to annual growth culmination may maximize live tree carbon stocks.

(4) *The quantified effects of silvicultural treatments on total net sequestration or emissions of carbon will depend significantly on how carbon accounting boundaries are drawn, i.e., which carbon pools and downstream effects are included in the analysis.*

In terms of *in situ* ecosystem stocks, the quantity of woody debris recruitment varies temporally between even-aged management and uneven-aged management, irrespective of utilization and site preparation activities.

In terms of forest product accounting, Harmon et al. (2009) modeled Douglas-fir ranging from low utilization crediting (50% of harvested live tree carbon converted to long-lived forest products with losses of 2%/yr), to high utilization crediting (75% of harvested live tree carbon converted to long-lived forest products with losses of 1%/yr). Clearcutting treatments only approached the equivalent carbon levels where retention is high under assumptions of high utilization crediting.

In terms of leakage, Taylor et al. (2008) used the carbon budget model used by the Canadian forest sector (CBM-CFS3) to compare the effect of partial and clearcutting in 100% stocked red spruce forests over a 240 year period. On a per hectare basis, partial cutting removed less than half as much biomass, but occurred twice as frequently as clearcutting.   Over an 80 year clearcut rotation and two partial cutting entries, 85% of merchantable biomass was removed from the clearcut treatments compared to 80% in partial-cut treatments, to account for volume losses due to higher anticipated mortality from wind exposure and logging operations damage in partially-cut treatments. The partial harvest resulted in 49.8 Mg C/ha greater carbon in the forest, but 16.8 Mg C/ha less merchantable timber, which may be considered as 34% leakage, assuming full timber demand inelasticity with reduced supply. The leakage effect could be mitigated by covering more area with the partial harvesting such that the amount of harvested volume is equal.

Finally, life-cycle accounting for brick, concrete, and steel (based on similar volumes and load-bearing capacities of materials) substitution for wood framing materials involves up to 33% more greenhouse gas emissions (Lippke et al., 2004). Accounting for this substitution category would shift silvicultural treatments to low retention and short rotation harvesting and also increase management intensity from natural forests to plantations (Hennigar et al., 2008). Substitution effects are difficult to directly attribute to timber supply due to macroeconomic influences, and also these effects cross greenhouse gas accounting sectoral boundaries.

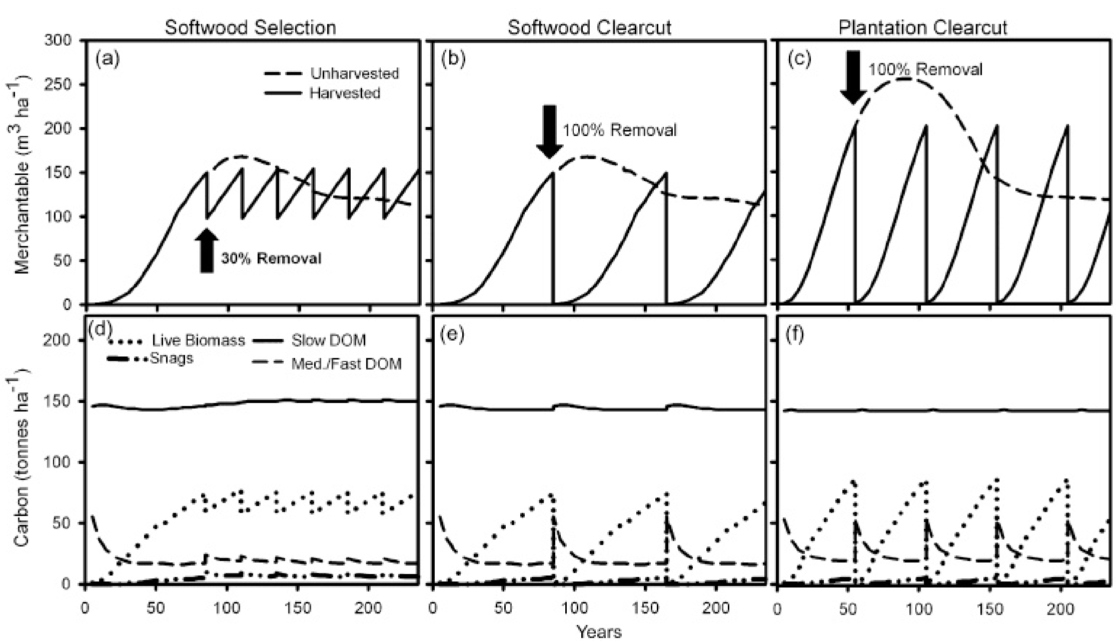

FIGURE ES-1:
Henninger et al., 2008

(A) Variation in the use of even- and uneven-aged management practices by forest type and biogeographic region in North America.

Definitions of silvicultural treatments

Silviculture is defined as the "theory and practice of controlling forest establishment, composition, structure, and growth" (Smith et al., 1997). Silvicultural treatments include both treatments of regeneration to remove the overstory to establish a new tree population (defined as even-aged (1 age class: clearcut and seed tree), multi-aged (2+ age classes: shelterwood) or uneven-aged (3+ age classes: individual tree and group selection)), and intermediate treatments (thinning from above, thinning from below, and geometric thinning) to improve the existing stand's commercial value and regulate its growth. Treatments are applied at the scale of stands and stands are contiguous groups of trees uniform in composition, age class, site quality, and condition. An "age class" and "cohort" are synonymous terms typically used to refer to a 10-20 year aggregation of trees originating from a single natural disturbance or silvicultural regeneration treatment.

Even-aged and uneven-aged regeneration methods

Clearcutting or cleancutting treatments involve removing all vegetation and making all growing space available for new plants. Other even-aged regeneration treatments retain some mature trees for a short period of time, typically 5-15 years, after "seed/establishment cutting" and while regeneration is still flexible and less vulnerable to breakage during the final removal cut. These treatments include seed tree which retains widely scattered trees of the desired species and dominant crown position with a high live crown ratio to provide seed for wind-scattered natural regeneration (which works for light-seeded shade-intolerant species, e.g. pines). Traditional shelterwood cutting typically involves two-stages, an establishment or seed cut and final removal cut (which works for species with advanced regeneration and intermediate tolerance, e.g. oaks), but may also involve multiple removal cuts. Extended and irregular shelterwood treatments involve maintaining mature trees (or reserves) for longer periods, typically up to one additional rotation. The primary distinction between the seed tree and shelterwood regeneration treatments involves the influence of residual trees on microsite conditions (primarily light levels) for regeneration.

Uneven-aged silvicultural treatments include single-tree and group selection and maintain at least three age classes at the stand level. This approach is also referred to as ”continuous cover forestry” treatments since it maintains perpetual forest cover through partial cuts marked by periodic stand “entries” or “cutting cycles” rather than “rotations” used in even-aged management. Single-tree selection is often practiced based on the BDq method (Guldin, 1991) specifying a total basal area (B), maximum diameter class at breast height (D), and target constant ratio between adjacent diameter classes (q); which is conceptually a regulated stand with trees in each age/diameter class represented by equivalent growing space (Nyland, 2001). Group selection occurs with adjacent trees rather than only individuals and can be based on either a BDq or an area control approach of maintaining equal area for each diameter class (generally populous small diameter trees and exponentially fewer large diameter trees). Please see Appendix A (Table A1) for a summary of employment of even- and uneven-aged silvicultural treatments by forest type in the North America.

Disturbance-based forest management as an alternative to traditional even- and uneven-aged silvicultural treatments

In at least two respects, even- and uneven-aged practices both fail to mimic natural disturbances and thus fail to maintain structure to support or "life boat through disturbances" late seral species (Angers et al., 2005): (1) uniformity of application spatially tending toward extremes of low (<10% basal area retained) or high (>70% basal area retained) as opposed to variable retention; and (2) lack of retention of large live standing and dead standing and downed trees. In terms of the first point for example, openings of 20-200 m^2 characterize uneven-aged management in hardwood forests of New England and the Great Lakes, but openings up to 1000 m^2 are necessary for regeneration of intermediate species such as yellow birch (Webster and Lorimer, 2005), openings with a wide range of 10-5000 m^2 occur naturally with windfall disturbances, and openings up to 8000 m^2 are necessary for maintaining early-successional bird species (Hanson and Lorimer, 2007). As an example of the second point, late successional forests in the northern hardwood region are characterized by abundant large live and standing dead trees (>=40 cm dbh), and late-successional spruce-fir forests are characterized by this factor combined with large (>=35 cm dbh) fallen woody debris (Whitman and Hagan, 2007), but this size of debris is uncommon in managed forests. The rationale for natural disturbance-based forest management involves limiting the "manipulation of forest ecosystems...within the [historical range of variability] established by natural disturbance patterns prior to extensive human alteration of the landscape" (Seymour and Hunter 1999). In addition to maintaining habitat for late-successional species, mimicry of natural disturbances may increase provision of ecosystem services including carbon storage (Franklin et al., 2002). Our modeling study examines whether these management practices (particularly variable retention harvesting) affect carbon dynamics over a potential project lifetime of 100 years and we determined that rotation rather than retention is a significant variable for intermediate to intolerant forest types, while the literature review in part (B) suggests that the opposite may be true for intermediate to tolerant forest types.

Disturbance-based forest management or emulation silviculture involves three areas that expand upon conventional silvicultural treatments: (1) variable retention harvesting, a component of disturbance-based forest management involving retention of live trees in multiple rotations to enable structural characteristics of mature forests, a treatment that might be termed “irregular shelterwood with reserves”; targets for maintaining large diameter (>=30 cm dbh) live trees or legacy trees include 14-20 m^2/ha in northern hardwood forests (Nunery and Keeton, 2010) and 6-10 m^2/ha in longleaf pine (Palik et al., 2010) (we mimicked this variable in FVS by establishing basal area retention targets and establishing cutting rules to cut suppressed and codominant trees); and (2) consideration of not only rotation and retention, but also spatial extent or magnitude, which allows for potential landscape rather than only stand scale considerations, such as maintenance of wildlife corridors and stream-side habitats (Lindenmayer and Franklin 2002; Seymour et al. 2002) (this factor was not included in FVS due to spatial limitations of the modeling program therefore potential biodiversity implications were not considered).

Variability in size and frequency of historic disturbance regimes in U.S.

North and Keeton (2008) have reviewed natural disturbance history in a variety of temperate forest types, and identified barriers to closely mimicking these disturbances. In the Pacific Northwest, crown fire return intervals tend to be much longer than commercial rotations for timber, varying along precipitation gradients from 200 years in drier central Oregon to 1000 years in wetter coastal Washington. In addition, both fires and wind blowdowns would impact 5,000-10,000 hectares at once under specific environmental conditions, though the impact would be heterogeneous. In the Rocky Mountain and Southeastern pine regions, low-intensity surface fires were historically frequent, but spatially heterogeneous in fashion not typically accounted in current management practices: entire watershed-scale burns of high intensity fires greater than 1,000 HA were infrequent (>250 years), riparian burns of approximately 10 HA were also infrequent (>100 years), midslope surface burns of approximately 100 HA were periodic (>20 years), and ridgetop burns of approximately 5 HA were frequent (>3.5 years). In New England, historic disturbances occur at various spatial and temporal scales from moderate frequency (100 years) and low intensity (<10% of biomass affected) and small extent (0.05 HA) from windthrow and surface fires to lower frequency but higher intensity and extent from ice storms and hurricanes (Seymour et al., 2002). A reconstruction of natural forest disturbances suggests that the current U.S. Forest Service FIA inventory indicates that 4% of the landscape is currently in seedling-sapling stage in Massachusetts, whereas 10% was in this stage in coastal pine-oak forests that dominate the state under historic natural disturbance regimes, and 25% of

Maine forests are in seedling-sapling stage, whereas the historic natural disturbance regime was 3-7% (Lorimer and White, 2003). In summary, although historical natural disturbances provide a useful template for ecological comparison, barriers exist to exact replication of historical disturbances including large-scale disturbances in Pacific Northwest infeasible in checkerboard ownership landscapes, high variability in disturbance intensity from fires in Rocky Mountains and Southeast that would greatly complicate management, and low disturbance history in New England in areas with the most productive commercial forest land.

Case study on variable retention component of natural disturbance based silviculture: DEMO

One of the most intensively researched natural disturbance-based silviculture experiments focused on variable retention is the Demonstration of Ecosystem Management Options (DEMO) study, which consists of six treatments, each 13 ha in size, replicated at six locations (blocks) in Douglas-fir dominated forests in western Washington and Oregon (Hansen et al., 1995, Aubrey et al., 2008). The experiment used a randomized complete block design:

(1) unharvested control—a reference for assessing responses to harvest and natural temporal variation

(2) 75% aggregated retention—all merchantable trees (>18 cm dbh) were harvested from three 1-ha circular gaps (56.4 m radius; 25% of the treatment unit);

(3) 40% aggregated retention—five circular 1-ha aggregates were retained (40% of the treatment unit); all merchantable trees in the surrounding matrix ("clearcut areas") were harvested;

(4) 40% dispersed retention (highest stand retention level to allow morphological development of Douglas-fir)—dominant and co-dominant trees were retained in an even distribution throughout the treatment unit; in each block, the basal area retained was equal to that retained in the corresponding 40% aggregated treatment;

(5) 15% aggregated retention—two circular 1-ha aggregates were retained (15% of the treatment unit); all merchantable trees in the surrounding matrix were harvested;

(6) 15% dispersed retention—dominant and co-dominant trees were retained in an even distribution throughout the treatment unit; in each block, the basal area retained was

equal to that retained in the corresponding 15% aggregated treatment (Aubrey et al., 2008).

Aggregate retention patterns >20% basal area provided greatest benefits in late seral species retention in gymnosperm and mixed forests

Aggregates at 40% retention provided refugia for a variety of late seral species including litter-dwelling arthropods, flying squirrels, shade-tolerant vascular plants, and bryophytes such as mosses and liverworts, compared to dispersed retention treatments (Aubrey et al., 2008). Greater retention of coarse barked hardwood tree species in aggregates also enabled lichen growth. Due to contrasting neighborhood impacts of angiosperm and gymnosperm tree species (such as differences in calcium soil and depth of the duff due to litter quality and decomposition rates), hardwood survival was higher in aggregations. A review of green tree retention in gymnosperm forests suggests that aggregates of greater than at least 20% basal area and at least one hectare size can provide biodiversity benefits, such as favoring bryophytes which require shady, moist habitats (Rosenvald and Lohmus, 2008). Aggregated retention can also allow strategic location of aggregated retention areas as wildlife corridors or streamside protection zones. Higher and aggregated retention patterns were also favored aesthetically, although greater slash (which provides habitat for bryophytes, salamanders and other species) was viewed negatively (Ribe, 2009).

Dispersed retention patterns provided greatest tree growth benefits, but studies vary on appropriate level of retention for Douglas-fir to provide sufficient light

The growth efficiency of dispersed retention at 40% was greatest due both to removal of trees in lower crown classes which may have been deformed or dying, and also wider distribution to fully utilize growing space (Aubrey et al., 2008). In contrast, dispersed trees at 15% retention had high annualized cumulative mortality from windthrow. High rates of mortality due to skidding damage near trails (far exceeding 0.5% estimated background mortality rate) is also likely in dispersed retention treatments during first 5-10 years after felling (Thorpe et al., 2008), but was not reported with DEMO.

Other studies, in contrast to DEMO findings, have demonstrated reduced height growth at 25-40% dispersed retention for intolerant Douglas-fir and larch relative to 0% retention, but not for shade tolerant species such as cedar, fir and spruce (Drever and Lerzman, 2001, Newsome et al., 2010). Harmon et al. (2009) suggests that 20% dispersed retention treatments may diminish Douglas-fir populations to 35% from 70% of live tree biomass. Another risk besides species shifts in partial harvests is spread of *Armillaria ostoyae* and *Inontus tomentosus* root rots, but no evidence of increased incidence was reported during at least first five years of stand development with up to 50% retention (DeLong et al., 2005).

(B) Even and uneven-aged management practices impacts on forest carbon during a potential project lifetime (i.e. 100 years).

Initial conditions of species composition and site quality determine carbon storage potential

Forest types might be targeted for forest carbon projects based on their carbon stocks and predicted response to management interventions. In a boreal mixed wood simulation of 200 years, maximizing carbon involved intensifying harvests in gymnosperm stands via conversion from natural regeneration to plantations to increase forest product carbon storage, while also minimizing harvests in angiosperm stands which have higher wood densities and growth rates (2.5 m3/ha/yr vs. 2.0 m3/ha/yr) to increase forest ecosystem carbon storage (Hennigar et al., 2008).

In addition to forest type, differences in site quality may influence carbon storage potential. Within a given forest type, higher than average site quality may mitigate diminishment of carbon stocks from disturbances by approximately 10-20% due to faster annual live tree growth rates (Keyser, 2010).

Higher live tree retention of larger diameter trees will create higher carbon stocks in intermediate to shade-tolerant angiosperm forests

Starting with an initial condition of a standing forest, any natural or human disturbance that reduces live tree biomass will reduce ecosystem-level carbon stocks, partly due to processing inefficiencies of 33-50% of cut volume (Harmon et al., 2009).

Nunery and Keeton (2010) used the Forest Vegetation Simulator (FVS) to project forest carbon and wood product carbon over 150 years in response to various silvicultural treatments in the northern hardwood forest type. Initial forest conditions were based on data from 32 US Forest Service Forest Inventory and Analysis plots. Simulations of a range of even-and uneven-aged prescriptions demonstrated that greater retention resulted in greater carbon stocks, with stands treated with individual/single tree selection (ITS) storing up to 33% greater carbon than stands treated with clearcutting (see Figure B1). Favoring retention of larger diameter live trees in particular (via high legacy tree retention targets of up to 12 trees/ha of average 41 cm dbh in the single tree selection treatments) increased forest carbon storage and had a more substantial impact than either extension of rotation or cutting cycle length.

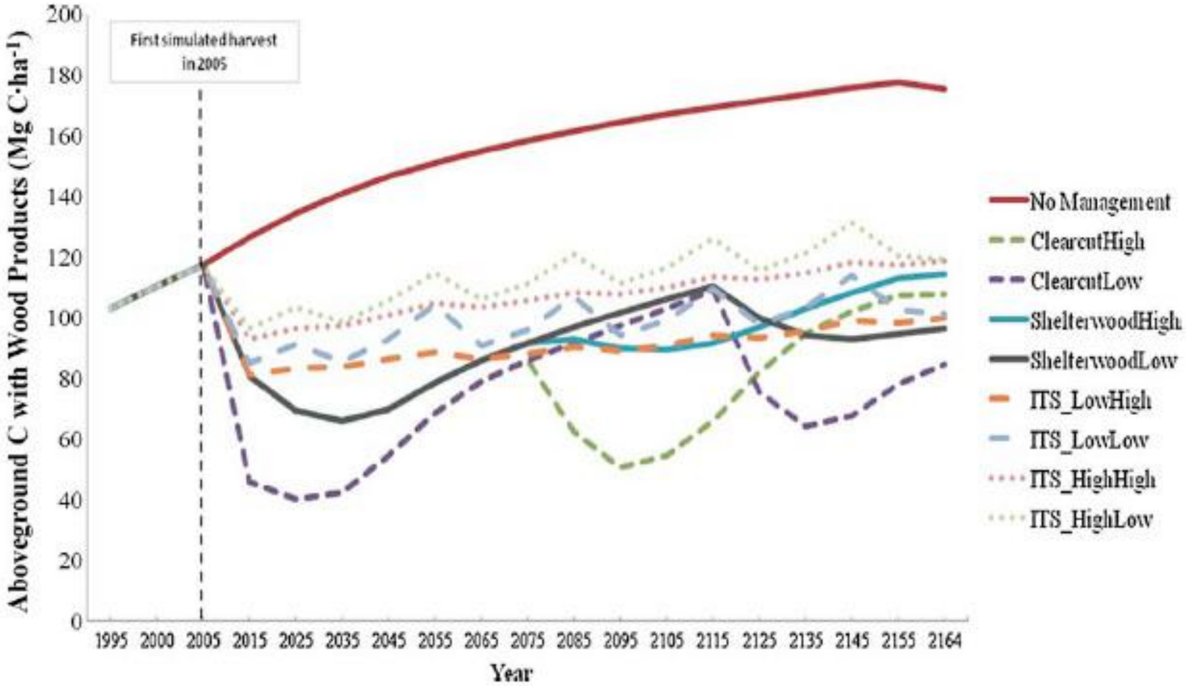

FIGURE B-1 Nunery and Keeton (2010)

Retention of larger diameter trees in particular may have a positive impact on forest carbon stocks (Nunery and Keeton, 2010). Commercial thinning in an even-aged cherry-maple forest showed that thinning primarily overtopped and suppressed trees from below only accelerated stand growth and development, resulting in similar levels of forest ecosystem carbon to untreated stands after 25 years (Hoover and Stout, 2007). However, thinning from above that removed dominants and co-dominants to the same relative density target reduced stand carbon stocks including live tree, dead wood and forest products by one-third over the 25 year period, presumably via stagnation.

Extending rotations increases stand level carbon storage in intermediate to intolerant gymnosperm forests

Extending rotation lengths also increases stand level carbon storage. Boreal and temperate forests can continue storing carbon at an average rate of approximately 2.5 t C/ha/yr up to 200 years (Luyssaert et al., 2008), with aboveground live tree carbon storage peaking at approximately 100 years in boreal pine (Seedre and Chen, 2010), and 200 years in temperate pine (Law et al., 2003).

Harmon et al. (2009) used the STANDCARB model in a Douglas-fir/western hemlock forest type to compare various levels of removal from 20% to 100% in multiple rotation intervals from 20-250 years. For example, a 20% removal (80% retention) treatment stored 2.2 times as much carbon per hectare as a 100% removal at 20 years rotation intervals, but the difference of forest ecosystem plus product carbon between the two treatments became insignificant at a 100 year rotation interval (see Figure B2).

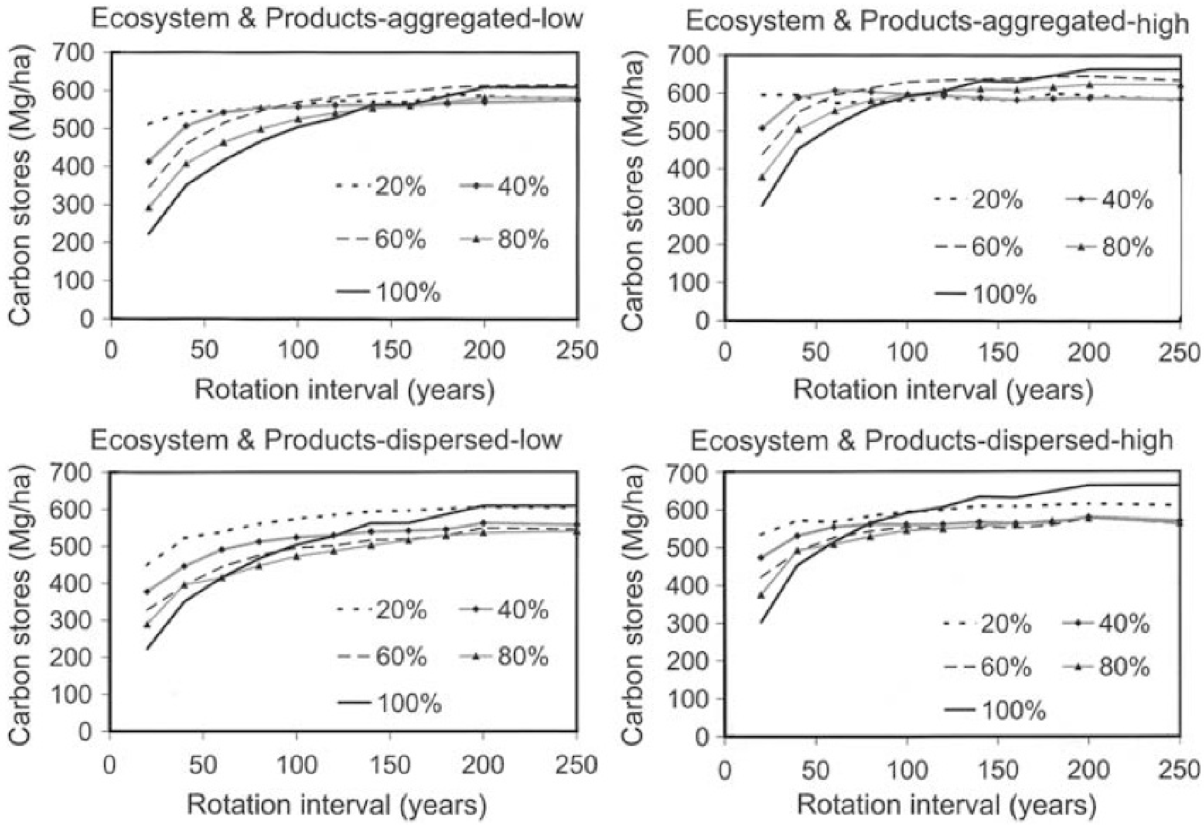

FIGURE B-2: Harmon et al., (2009)

(C)Effect of even-aged management and uneven-aged management on lying dead wood, litter and duff, and soil carbon.

Lying dead wood

Lying dead wood or downed necromass includes branches, twigs, and boles that can be divided into fine woody debris and coarse woody debris based on size class. U.S. Forest Service Forest Inventory and Analysis (FIA) establishes size classes based on diameter at mid-point of segment length. Fine woody debris includes: <0.64, <2.5, and

2.5-7.6 cm size classes based on fire fuel classes (0.25-1, 1-5, 5-10 cm based on Intergovernmental Panel on Climate Change (IPCC) 2006 Guidelines for National Greenhouse Gas Inventories definition). Coarse woody debris includes: 7.6-22.6 and >22.6 cm size classes (10-20, >20 cm size based on IPCC Guidelines). In addition to size, woody debris is typically characterized by species name and decomposition state on an index from 1-5, with 1 being newly fallen to 3 being branchless to 5 being nearly fully decomposed (Maser et al., 1979). Rates of decomposition for downed wood are typically 1-5%/yr in gymnosperm and mixed forest high latitude zones with low temperatures unfavorable to decomposition (Laiho and Prescott, 2004) 5-10%/yr in angiosperm temperate forests (Harmon and Hua, 1991, Gough et al., 2007) and 10+%/yr in wet lowland tropical forests. High lignin content and high volume to surface ratios are additional factors that may slow decomposition. Laiho and Prescott (2004) estimate that 15 cm average diameter gymnosperm logs in direct contact with ground in Alberta, Canada, would decompose completely in 35-45 years.

Table B.1 in Appendix B shows 5-15% of aboveground ecosystem carbon stored in lying dead wood in New England northern hardwood forest, 8% in mid-Atlantic oak-hickory forest, and 5-10% in Rocky Mountain western conifer forest (Bradford et al., 2009), which exceeds required carbon accounting in shrubs and herbaceous understory and matches accounting for standing dead wood. Similar results were found in a boreal forests (Hagemann et al., 2009) and moist and montane neotropical forests (Delaney et al., 1997). The inclusion of aboveground stumps as coarse woody debris in managed forests can further increase coarse woody debris stocks (North et al., 2009).

Woody debris accumulation typically follows a U-shaped curve during even-aged forest stand development, with the largest volumes encountered during the stand initiation stage, followed by decomposition so that levels drop to a minimum during stem exclusion stage, then rebuilding again with contribution of detritus from single tree and group senescence during the understory re-initiation stage (Oliver and Larson, 1996). Woody debris recruitment (both standing and downed) is determined not only by stand development stage, but also by type, frequency and intensity of past disturbances, both human and natural (Pyle et al., 2008). In a boreal black spruce-balsam fir forest for example, total aboveground biomass of fallen necromass was 54 t C/ha immediately after commercial clearcut harvesting, over eight times greater than biomass in snags, then dropped to 10 tons 17 years after harvest and reached minimum of 2.5 tons 35 years after harvest, then rose again to 24 tons at old growth stage over 100+ years in chronosequence (Hagemann et al., 2009).

Even-aged management is likely to negatively influence standing but not necessarily downed wood stocks in the short term, depending on tree utilization for biomass, firewood, or pulp markets, and also subsequent slash treatment[5]. Eight years post-harvest in Missouri Ozark Mountains, snag biomass in clearcuts was less than one-tenth the level in unharvested control and single-tree selection. In contrast, downed

[5] In terms of natural forest management, snags are widely recognized as an important component of wildlife habitat. . Lohr et al. (2002) demonstrated the importance of snags by removing them in 40- to 50-year old loblolly pine plantations in South Carolina, which negatively affected total bird species diversity and dramatically reduced woodpecker abundance. However, fallen woody debris removal also caused declines in populations of nearly all bird guilds including weak excavators, secondary cavity nesters, and neotropical migrants who used debris primarily for feeding on insects.

woody debris biomass was two times the level in clearcut as in control and single-tree selection (Li et al., 2007). Over time, the paucity of retention of both live and dead standing trees in clearcuts with slash treatment will reduce the recruitment of downed dead wood. Over 30 years after clearcutting treatments in the northern Rocky Mountains, fallen coarse woody debris biomass was 700% lower in second growth clearcut than in old growth forests (1 t C/ha vs. 7 t C/ha), even though other biomass stocks were only approximately 100% lower including live trees (25 t C/ha vs. 47 t C/ha), coarse roots (6 t C/ha vs. 13 t C/ha), and forest floor (5 t C/ha vs. 8 t C/ha) (Bisbing et al., 2010).

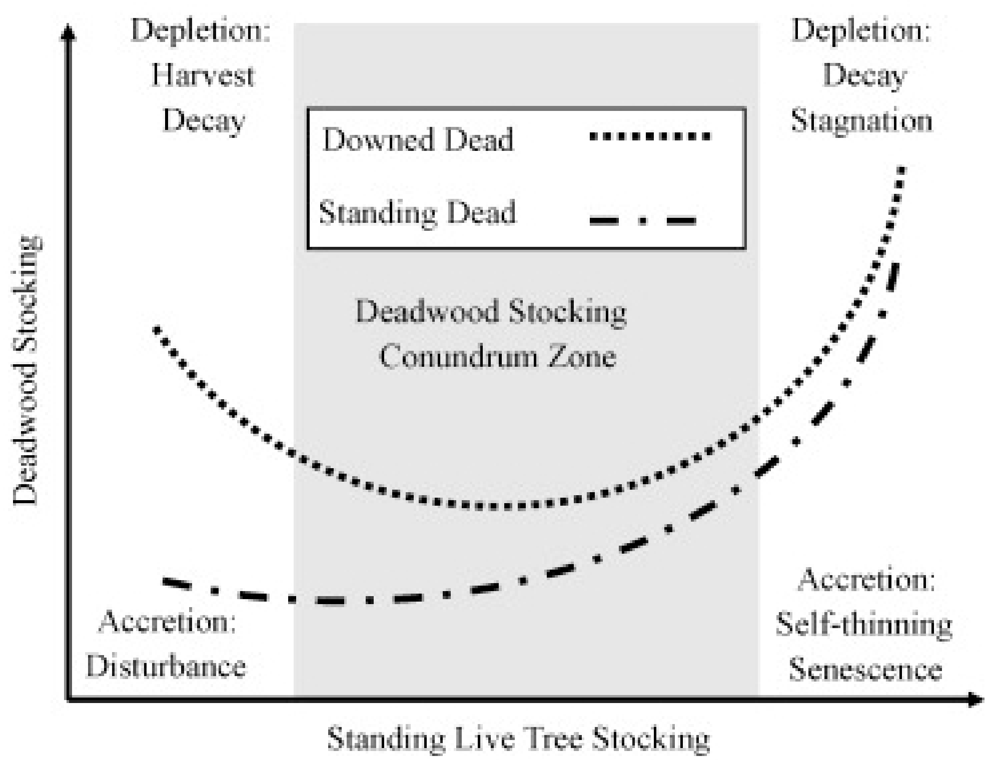

FIGURE C-1: (Woodall and Westfall, 2009).

Litter and duff

Litter, on top of duff, constitutes recognizable non-woody plant parts, such as leaves and flowers (O1 organic matter layer) (6-10 mm in diameter based on IPCC standards). Duff constitutes partly decomposed, relatively uncompacted organic material above mineral soil (O2 organic matter layer) (2-6 mm diameter based on IPCC standards). Together duff and litter constitute the forest floor.

In boreal forests, the quantity of carbon in forest floors may be particularly high, constituting an equivalent of 40% of aboveground carbon stocks, exceeding many other stocks outside of live trees, due to low rate of decomposition (Kranabetter, 2009; Table C.1). Similar ratios are found in gymnosperm high alpine regions (Bradford et al., 2009). The forest floor carbon stocks might also account for bryophytes such as sphagnum moss and buried coarse woody debris in boreal and wet temperate forests, which can more than double the proportion of total ecosystem carbon stored in the forest floor (Goulden et al., 2008). Regeneration harvests promote increased decomposition of organic matter due to increased temperatures and soil surface exposure, thus reducing forest floor depth, but this impact may be offset by the organic matter contribution to soil from logging slash (Johnson and Curtis, 2001). In temperate forests, the forest floor generally constitutes the equivalent of 10-30% of aboveground carbon stocks as shown Table C.1 (Johnson et al., 2010). In tropical forests the forest floor generally constitutes <10% of ecosystem biomass due to rapid decomposition rates (Delaney et al., 1997). Changes in the forest floor carbon pool, though uncorrelated with live tree retention as explained below, can significantly influence total forest carbon stocks.

Logging operations (such as dispersed ground-based skidding without snow cover) rather than silvicultural treatment are anticipated to have the greatest impact on forest floor carbon by causing physical disturbance and accelerating forest floor decomposition (Jandl et al. 2007). In addition, broadcast burning can result in losses of litter carbon up to 60% during the first five years post burn, without evident re-accumulation by year 20 (Kranabetter and Macadam, 2007). Prescribed burning for crown fire control in Rocky Mountain conifer forests resulted in emissions of 4.5-18 t C/ha, largely from forest floor emissions, and resulted in emissions of 14.5 t C/ha in northern California (North et al., 2009). Thinning small to medium sized trees to concentrate diameter increment in larger trees can partially offset these emissions over 10-20 years (Hurteau and North, 2009), but these emissions must be counted at the time they occur under accrual accounting principles. Peat soil draining may also significantly reduce forest floor carbon.

Soil Carbon

Mineral soil carbon is a significant component of forest ecosystem carbon stocks, typically constituting between one-third and two-thirds of total ecosystem biomass and typically outweighing aboveground tree biomass in young forests. However, neither forest age nor natural or human disturbance appear to impact soil carbon in a consistent manner. For example, in an Ozark oak forest uneven-aged management and clearcutting did not result in significant soil carbon change relative to unharvested control (Li et al., 2007). Thinning and regeneration harvests have also had no consistent impact on soil carbon in boreal forests (Martin et al., 2005). Over 100 years or longer, modeling in boreal forests suggests that whole tree harvests (removing entire trees from a stand rather than only the boles without dispersion of tops and branches back into the stand) will diminish soil organic matter and nitrogen content, thus diminishing net primary productivity by up to one-third (Peng et al., 2002). However, field studies in northern New Hampshire and Maine temperate forests with a short time of 15 years have not demonstrated reductions in soil carbon under whole tree harvesting treatments (McLaughlin and Philips, 2006). Indeed, over a 2 year period, whole tree harvesting and clearcutting in Douglas-fir forests in Washington increased mineral soil carbon (>50 cm depth) relative to levels under initial forest cover conditions by increasing root decomposition via higher temperatures (Slesak et al., 2009). In contrast to peat drainage in tropical forests (Page et al., 2002), peat drainage in temperate and boreal forests may not necessarily involve net greenhouse gas emissions as drainage minimizes sedge growth, and sedges are a main exporter of methane to the atmosphere from methanogenic microbes, though a time lag may be involved which will need to be accounted until forest carbon stocks reach a density over 100 m3/ha which reduces sedge growth (Minkkinen et al., 2002). Soil carbon is not anticipated to change substantially as a result of most project activities, except in cases of direct soil disturbance such as plowing, ripping, furrowing, and subsoiling which are widely documented to reduce forest ecosystem carbon (Dias et al., 2007). Research on potential negative soil carbon impacts from multiple rotation whole tree harvests and also peat drainage in boreal and temperate forests should continue to be monitored.

(D) A Case Study of the Carbon Stocks in Various Management Regimes in a Coastal Pacific Northwest Douglas-Fir Forest Type

Introduction

The carbon sequestration and storage implications of silviculture for North American forests have been studied for a number of geographical areas, forest types, and life cycle components. Past studies have not, however, addressed in detail the question of how quantification results differ depending on the scope of the analysis (i.e., different sources, sinks, and effects considered). The study presented here examines the net sequestration of carbon under a range of silvicultural practices quantified according to Version 3.2 of the CAR forestry protocol for improved forest management.

Methods

The Pacific Northwest Cascade mixed forest (McNab et al. 2005, M242), Oregon and Washington Coast Range was studied. Initial starting data were from the USDA Forest Service, Forest Inventory Analysis (FIA) program (FIA 2010). The growth simulator used was the Forest Vegetation Simulator (FVS), Pacific Northwest variant (PN) version 07/02/2010 (Keyser 2010).

The carbon estimates were based on the FIA volume and biomass functions for above-ground live tree components (FIA 2009a, b), the function given in the protocol for below-ground live tree biomass (Cairns et al. 1997), and other components were estimated using the FVS Fire and Fuels Extension (FFE) carbon tracking (Rebain 2010).

The following parameters were used to screen plots, producing a similar set of data for analysis.

(3) Assessment Area: M242A

(4) Forest Type: 201, Douglas-fir

(5) Maximum Slope: 50%

(6) Landowner Code: 46, private

(7) Maximum Elevation: 3000 feet (914 meters)

(8) Ground Land Class: 120, timberland

(9) Physiographic Class: Mesic

There were 310 FIA plots that met these criteria. The CAR Appendix F assessment area criteria for this region were 24 tonnes per acre (59.3 tonnes per hectare) live above-ground C for low site and 39 (96.3 tonnes per hectare) tonnes per acre live above-ground C for high site. Table D-1 shows the resulting FIA plot attributes.

Table D-1. Plot characteristics for all trees on the plots.

Category	No. of Plots	Basal Area (sq. ft per acre)	Basal Area (sq. m per hectare)	Trees per Acre	Trees per Hectare	Avg. Diam. (QMD) (in)	Avg. Diam. (QMD) (cm)	Aboveground Live Tree C (tonnes/acre)	Aboveground Live Tree C (tonnes/hectare)
Below CP Stocking (39), High Site	213	68.7	15.8	343.0	847.3	6.1	15.4	14.2	35.0
Below CP Stocking (24), Low Site	17	38.8	8.9	230.0	568.1	5.6	14.1	10.1	25.1
Above CP Stocking (39), High Site	69	199.7	45.8	266.0	656.9	11.7	29.8	60.9	150.4
Above CP Stocking (24), Low Site	11	166.8	38.3	201.4	497.4	12.3	31.3	53.7	132.6

In order to estimate the relative carbon stocks produced based on the CAR forestry protocol, 100-year simulations of growth, harvest and regeneration were produced. Each simulation started with the FIA measured data assuming a start year of 2010. Plots were grown in 5-year increments. If a harvest occurred in a decade then it was scheduled to occur mid-period so that 5 years of growth would occur before and after the harvest. Regeneration harvests (clearcut, variable retention) were followed by planting of Douglas-fir on a 12x12 foot (3.7 m) spacing, which produced 304 trees per acre (751 trees per hectare). Where the stand age was available for the plot, it was used as a criterion for implementing a regeneration harvest. Otherwise a zero age was assigned. Each regeneration cycle was repeated. Considering a 50-year rotation for example, if a stand was 30 years of age it was harvested in decade three and again 50 years later in decade eight in the FVS projections. A range of silviculture was considered including a no-harvest scenario; clearcut at ages 30, 50, 70 and 90; and dispersed variable retention (VR) leaving 15% and 40% of basal area at ages 30, 50, 70 and 90. Each plot was considered independently as a project. This allowed an analysis to identify factors that were influential on carbon stocking under the Climate Action Reserve protocol. Grouping plots into projects would have been more realistic from a project development perspective, but would have reduced the ability to draw inferences with the subsequent analysis. When combining all the plots it was possible to examine other factors such as site quality and starting inventories. The example in Appendix C was for carbon yields and not reductions, which requires a projection of project activity and baseline to calculate gross reductions. Gross reductions were estimated using a linear programming optimization library (GIPALS, 2010) as part of Spatial Informatics

Group, LLC carbon analysis software. Reductions for each plot were calculated by maximizing the gross CRTs. Baselines were allowed to select any combination of prescriptions while project activities were run using the following scenarios.

(1) Grow with no harvest.

(2) Clearcut at 30 years.

(3) Clearcut at 50 years.

(4) Clearcut at 70 years.

(5) Clearcut at 90 years.

(6) Dispersed Variable Retention (15% retention) at 30 years.

(7) Dispersed Variable Retention (15% retention) at 50 years.

(8) Dispersed Variable Retention (15% retention) at 70 years.

(9) Dispersed Variable Retention (15% retention) at 90 years.

(10) Dispersed Variable Retention (40% retention) at 30 years.

(11) Dispersed Variable Retention (40% retention) at 50 years.

(12) Dispersed Variable Retention (40% retention) at 70 years.

(13) Dispersed Variable Retention (40% retention) at 90 years.

Reductions were calculated based on the CAR improved forest management equation 6.1 in version 3.2 of the forestry protocols. This incorporated onsite and offsite carbon, a 20% leakage deduction, secondary effects estimates and negative carryovers. Buffer pool reductions were not applied as they would have been the same across all plots and would not have contributed information to the analysis. If the project activity landfill pool was less than the baseline landfill pool then the difference was subtracted from the reductions. If offsite carbon reductions were greater than onsite carbon reductions then an adjustment was made disallowing the difference in credit, which is consistent with the protocol.

Data Analysis

The age class distribution of the plots was plotted to inform the context of the harvest scheduling. The simulations described above resulted in 4,030 data points for analysis. Two response variables were analyzed:

1. required carbon pools, and
2. all carbon pools but soil (inclusion of fallen woody debris, shrubs, and forest floor in-situ carbon pools).

The data was analyzed using recursive partitioning (Breiman et al. 1984; Everitt and Hothorn 2006; Murthy 1998). The Rpart library (Therneau et al. 2007), version 3.1-36, was used for the modeling analysis. Recursive partitioning on the data was conducted using default parameters. Pruning of the resulting trees was conducted automatically using the cost-complexity prune function (Breiman et al. 1984) to minimize the cross-validated prediction error (Everitt and Hothorn 2006).

The independent variables considered were as follows:

- Silvicultural method + Rotation Age: Clearcut (30, 50, 70, 90), VR-15 (30, 50, 70, 90), VR-40 (30, 50, 70, 90), No-harvest
- Silvicultural method: Clearcut, VR-15, VR-40, No-harvest
- Rotation Age: 30, 50, 70, 90, NA(no-harvest)
- Common Practice Site Class: Low or High
- Quadratic Mean Diameter (QMD) of the initial stand
- Gross thousand cubic foot volume of the initial stand
- Site Class based on FIA classes: 1-7
- Percent slope: 0-100%
- Aspect: 0 - 359 degrees

- Age of initial stand

The site classes and common practice site class produced the same results as did the QMD and volume of initial stands. The carbon pool contributions were estimated for each scenario that was identified as statistically significant. This was done for the analysis conducted on all the pools but soil and was reported as percent contribution. Two additional analyses were performed to compare the impacts of the protocol rules relative to a straight averaging of carbon yields over the 100-year period for the projected project activity. This was done for the required pools and all the pools except soil. The landfill pool was included.

Results

The age class distribution is shown in Figure D-1. Private forestlands in this region are clearly distributed in younger age classes, which will have a bearing on the results of the analysis given that we are considering a 100-year harvest schedule. Tables D-2 and D-3 (metric) shows the average carbon stocks in order of importance with factor II being a subset of factor I, etc. Unharvested stands produced the highest average carbon stocking (248 t/a) and rotation age 30 produced the lowest average carbon (95 t/a).

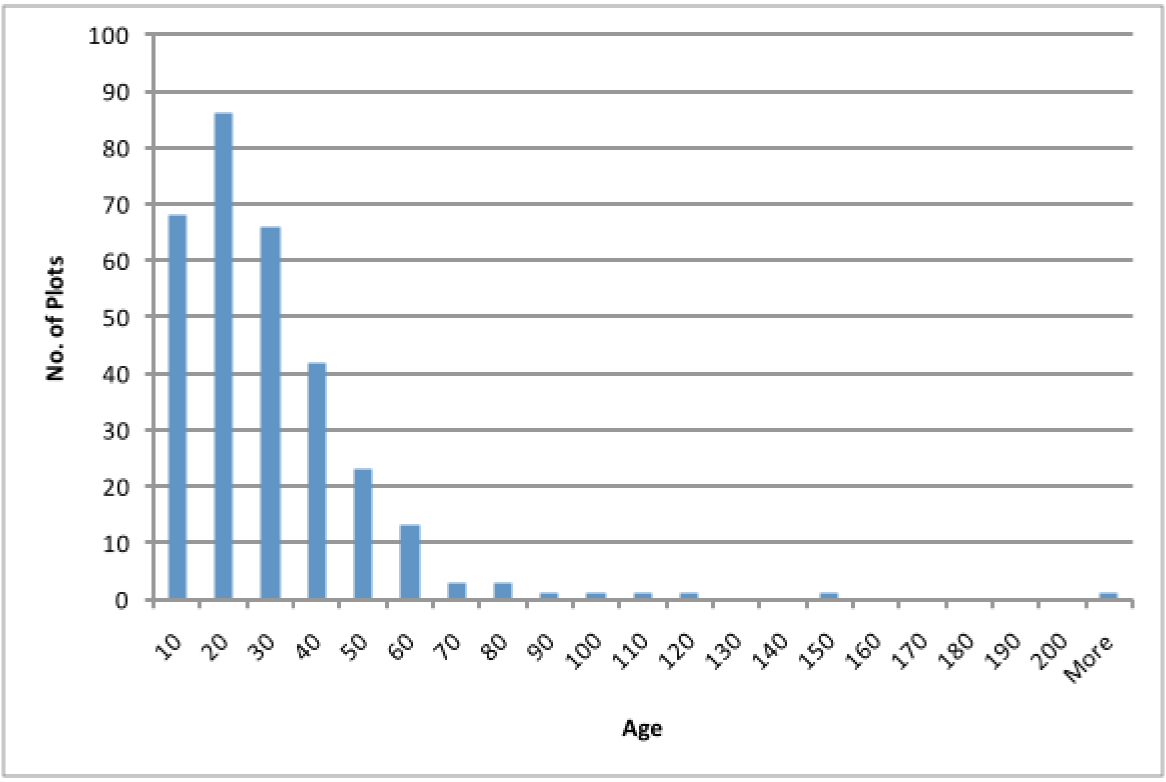

Figure D-1. Age class distribution of plots.

Rotation ages of 70 and 90 produced the highest stocking, where harvesting occurred, with an average of 161 t/a. Higher quality sites (FIA site class 1-3) produced higher stocking on average than lower quality sites: 167 t/a versus 98 t/a. Higher sites with rotation ages of 70 produced higher carbon stocking on average than 90 year rotations: 186 t/a versus 148 t/a. Rotation ages of 50 produced average carbon stocks of 130 t/a.

Table D-2. Average carbon stocks from project activity (carbon tonnes/acre) for the required pools only.

Factor			Average Stocks		
I	II	III	I	II	III
No harvest			248		
	Higher Inititial Vol.			267	
	Lower Initial Vol.			194	
Rotation Age 70 and 90			161		
	Lower Site			98	
	Higher Site			167	
		Rotation Age 70			186
		Rotation Age 90			148
Rotation Age 50			130		
Rotation Age 30			95		

Table D-3. Average carbon stocks from project activity (carbon tonnes/hectare) for the required pools only.

Factor			Average Stocks		
I	II	III	I	II	III
No harvest			613		
	Higher Inititial Vol.			659	
	Lower Initial Vol.			479	
Rotation Age 70 and 90			398		
	Lower Site			242	
	Higher Site			412	
		Rotation Age 70			459
		Rotation Age 90			366
Rotation Age 50			321		
Rotation Age 30			235		

Tables D-4 and D-5 (metric) show the resulting average carbon stocks considering all the carbon pools but soil. The general patterns are the same.

Table D-4. Average carbon stocks from project activity (carbon tonnes/acre) for the all pools but soil.

Factor			Average Stocks		
I	II	III	I	II	III
No harvest			317		
	Higher Initital Vol.			342	
	Lower Initial Vol.			247	
Rotation Age 70 and 90			206		
	Lower Site			131	
	Higher Site			214	
		Rotation Age 70			233
		Rotation Age 90			195
Rotation Age 50			166		
Rotation Age 30			114		

Table D-5. Average carbon stocks from project activity (carbon tonnes/hectare) for the all pools but soil.

Factor			Average Stocks		
I	II	III	I	II	III
No harvest			783		
	Higher Initital Vol.			845	
	Lower Initial Vol.			610	
Rotation Age 70 and 90			509		
	Lower Site			324	
	Higher Site			529	
		Rotation Age 70			576
		Rotation Age 90			482
Rotation Age 50			410		
Rotation Age 30			282		

Table D-6 shows the contribution of each required and optional carbon pool towards average carbon stocks. The live tree pool was the largest pool in all cases followed by the standing dead for the required pools. The in-use wood products pool ranged between 15% and 24%, but was not all countable towards reductions.

Table D-6. Contribution of each carbon pool as a percent of the total average stocking.

Description	Required Pools				Optional Pools		
	Live Tree	Standing Dead	In-Use	Landfill	Shrubs/Forbs	Down Dead	Floor
No Harvest	64%	14%	0%	0%	0%	17%	4%
Rot. Ages 70 and 90	41%	11%	15%	10%	0%	17%	6%
Rot. Ages 70 and 90, Lower Site	38%	10%	16%	11%	0%	17%	8%
Rot. Ages 70 and 90, Higher Site	41%	12%	15%	10%	0%	17%	6%
Rot. Age 70, Higher Site	49%	11%	14%	10%	0%	16%	6%
Rot. Age 90, Higher Site	37%	13%	18%	12%	0%	20%	7%
Rot. Age 50	35%	10%	24%	16%	0%	15%	8%
Rot. Age 30	49%	4%	24%	16%	0%	7%	11%

Discussion

Carbon stocks produced using the CAR forestry protocol for improved forest management were clearly influenced by presence or absence of harvesting activities. Preservation scenarios were optimal for carbon stocks, but we did not consider leakage, land-use change nor natural disturbance risk from wind and fire. For example, in terms of land-use change, if a working forest was converted to a "no harvest" management then leakage would emerge as a larger factor. Next to presence and absence of harvesting, the next most influential factor appeared to be the age of the harvested trees. For the forest type modeled, Northwest Coast Douglas-fir, a rotation age of around 70 appeared optimal, probably due to culmination of mean annual increment. Shorter rotations of 50 and 30 years produced the lowest stocks on average in Douglas-fir, with a reduction of over two-thirds compared to a 70 year rotation. A rotation of 90 years reduced carbon stocks compared to 70 years. Projections of coastal redwood would be longer to maximize carbon based on its later culmination of biomass, likely over 100 years; and projections of loblolly pine would be shorter based on its earlier culmination.

The third substantial finding was that site class and initial stocking were influential in determining carbon stocks for 70 year rotations; and initial stocking was significant for 90 year rotations. Lower site class reduced carbon stocks by nearly 60% by lowering growing space potential. Initial stocking levels had a similar but less profound impact of approximately 30% likely due to reversal of biomass volume from high initial to lower residual levels with harvesting (in line with our initial finding that all harvesting reduces ecosystem level carbon in a standing forest).

A complex array of factors combine when considering silviculture within a 100-year harvest schedule with fixed rules for the various on and offsite carbon pools. The use of realistic inventory data representative of a region (FIA) in a simulation context (FVS) is a useful means of understanding this complexity in a real-world context and identifying important variables over time.

This analysis shows that:

1) all harvesting reduces forest carbon by over 40% when starting with a standing forest;

2) rotation length overrides retention due to silvicultural treatment as the significant factor influencing forest carbon and the age of maximum mean annual increment for the dominant species likely determines the rotation interval for maximizing forest carbon; variable retention treatments had no significant impact on carbon stocks when considering all forest carbon stocks, and may have had negative impacts due to leakage not considered in our analysis;

3) higher site quality classes will enable greater carbon reductions over time and lower initial stocking levels will allow less diminishment from harvesting, together increasing potential carbon stocks.

A 100-year harvest schedule is necessary for baseline calculations in a Climate Action Reserve improved forest management project. The project activity could, however, be as short as one year with subsequent monitoring. Project length, initial inventory conditions, economics, or other factors could influence optimal carbon reductions for a given project. We modeled dispersed variable retention silviculture in this paper as a means of characterizing silviculture less intense than clearcutting but still actively managed. While difficult to model accurately due to competition and regeneration factors, other forms of silvicultural methods such as aggregated variable retention, group selection or single tree selection may interact with the other factors analyzed in this paper in different ways in this and other forest types.

APPENDIX A

Variation in the use of even- and uneven-aged management practices by forest type and biogeographic region in North America.

Silviculture practices vary by region. Table A1 displays this variation and compares the dominant silviculture activity to the natural disturbance most commonly associated with the forest type.

TABLE A1: Common management practice by forest type in U.S. (Barrett, 1995)

Note the following trends:

*Uneven-aged treatments generally have frequent entry cycles compared to even-aged rotation lengths to achieve similar volume targets;

*Uneven- and 2-aged treatments are primarily employed in forests with windthrow and surface fire natural disturbance history, whereas even-aged treatments are primarily employed in forest types with crown fire disturbance history;

*Note that uneven-aged silvicultural treatments are not widely employed in reality. In areas with high landownership by nonindustrial private/family forest owners such as U.S. Northeast and Southeast (Best et al., 2001), exploitative partial harvesting to maximize current harvest economic value without consideration of future stand growth and regeneration is commonplace (Munsell and Germain, 2007), typically involving "high grading" (selective cutting of individual trees of highest commercial value in terms of size, species, and quality).

Forest type and composition	Major natural disturbances	Silvicultural practices typically employed*	Estimated rotation period (range from low site index or pulpwood to high site index or sawlogs)
Uneven-aged (3+ cohorts) regeneration treatments customary			
New England and Great Lakes: northern hardwood (sugar maple-american beech-yellow birch)	Individual or small patch disturbances from wind throw. Infrequent hurricanes in New England and fires in Great Lake states.	Single tree and group selection used for shade-tolerant and intermediate species that require approx. 80% canopy cover for regeneration; herbicide control or mechanical brushing necessary to reduce competing striped maple and hobblebush regeneration Two-staged shelterwood (irregular shelterwood including initial cut and delayed overstory removal) also commonly employed, particularly with cherry component	20-40 year cutting cycles
Great Lakes and Southeast: Bottomland Hardwoods Cottonwood-Willow-Birch (Early Seral), Elm-Ash (Mid Seral), Oak-Hickory (Late Seral)	Individual or small patch disturbances from wind throw	Single tree and group selection used for shade-tolerant and intermediate species	20-40 year cutting cycles
Southeast: Longleaf-slash pine	Surface fires	Clearcut, seed tree, and two-staged shelterwood within 10 years with soil scarification and hardwood control. Group and individual tree selection periodically used Prescribed fire periodically used.	30-60 years 20-40 year cutting cycle

Forest type and composition	Major natural disturbances	Silvicultural practices typically employed*	Estimated rotation period (range from low site index or pulpwood to high site index or sawlogs)
Southeast: Loblolly-shortleaf pine	Crown and surface fires	Clearcut, seed tree, and two-staged shelterwood within 10 years with soil scarification and hardwood control Group and individual tree selection periodically used Prescribed fires periodically used.	30-60 years 20-40 year cutting cycle
Rocky Mountains: ponderosa pine (occasionally seral)-grand fir-Douglas-fir	Surface fires, root rot in Douglas-fir, western spruce budworm in Douglas-fir, mountain pine bark beetle, dwarf mistletoe	Single tree and group selection used for ponderosa pine regeneration along with prescribed fire to control more shade-tolerant fir populations Shelterwood to regenerate shade-tolerant but rot-susceptible fir	20-40 year cutting cycles
Even-aged (2 cohorts) regeneration treatments customary			
Mid-Atlantic and Great Lakes: Oak-pine (xeric), oak-hickory (mesic), oak-	Crown and surface fires; deer browsing	Two to three stage shelterwood to promote combination of advanced and new regeneration often combined with herbicide control of oak competitors in mesic sites Clearcutting favors non-oak vegetative competition Unthinned stands of average site quality (18 m height @ 50 years) can grow at annual rate of 3.4 m3/ha/yr	80-120 years

Forest type and composition	Major natural disturbances	Silvicultural practices typically employed*	Estimated rotation period (range from low site index or pulpwood to high site index or sawlogs)
New England, Mid-Atlantic, and Great Lakes: White and red pine (seral to hardwoods)	Crown and surface fires; white pine blister rust	Up to three-staged shelterwood with one-third removal in establishment, one-third removal after 5 years in intermediate, and one-third removal after 15 years in final harvest.	60-100 years
New England and Great Lakes: red spruce-balsam fir	Fir has easiest reproduction but most susceptible to spruce bud worm. Acidic precipitation has also affected high-elevation spruce-fir via interference with calcium-mediated winter hardening off process	Clearcut and seed tree cut with soil scarification and hardwood control Shelterwood harvests periodically used; typically 40% basal area establishment cut favoring spruce since more difficult to regenerate, then 100% removal harvest in year 10	30-60 years
Pacific coast: Douglas-fir	Windthrow, crown fire	Clearcut, seed tree, two-staged shelterwood and irregular shelterwood with soil scarification and hardwood control. Thinning at 20-40% to produce highest rates of diameter growth in individual trees; thinning at 40-60% to produce highest volume stands	40-120 years
Even-aged (1 cohort) regeneration treatments customary			
Great Lakes: Jack pine (xeric, infertile soils; seral to spruce-fir)	Crown and surface fires; white pine blister rust	Clearcut, seed tree with approximately 20 tree/ha retention, Mineral soil scarification required along with slash treatment for adequate stocking. Prescribed fire periodically used. Herbicide control of hardwood regeneration required in mesic sites.	60-100 years

Forest type and composition	Major natural disturbances	Silvicultural practices typically employed*	Estimated rotation period (range from low site index or pulpwood to high site index or sawlogs)
Great Lakes and Rocky Mountains: Aspen	Crown fires	Coppicing with sprout regeneration	30-60 years
Rocky Mountains: Lodgepole pine	Dwarf mistletoe, mountain pine bark beetle, crown fires	Clearcut Slash treatment to prevent Ips. beetle outbreak	60-80 years
Pacific coast: Western hemlock-sitka spruce (seral) occasionally fir and redwood associates	Windthrow, spruce weevil, variable fire intensity	Clearcut and seed tree; susceptible to blowdown and mechanical damage	40-80 years
Pacific coast: Redwood	Windthrow, flooding, landslides, variable fire intensity	Clearcut and seed tree	40-80 years
Pacific coast: True/silver fir-mountain hemlock (seral)	Windthrow, shoestring fungus and Annosus root disease, variable fire intensity	Clearcut and seed tree; susceptible to blowdown and mechanical damage	100-150 years
Upland Alaska: white and black spruce (birch and aspen first seral, poplar and alder second seral, white spruce third seral and black spruce fourth seral)	Crown fires, spruce bark beetle and budworm, moose browsing	Clearcut and seed tree	100-200 years
Coastal Alaska: Sitka spruce and western hemlock (alder first seral, hemlock second seral, spruce and red cedar third seral)	Crown fires, windthrow, spruce budworm	Clearcut and seed tree	50-150 years

APPENDIX B

(B) Relative carbon stocks by carbon pools in various forest types

TABLE B1: Representative summary of measured carbon stocks in various forest types.

Note the following trends:

*aboveground tree biomass (live and dead) increases with age and is highest in moist temperate forests, such as Pacific Northwest;

*belowground biomass varies from 12-66% of aboveground tree biomass and is greatest in young forests and in dry forests as a proportion of aboveground biomass;

*lying necromass varies from 2.5-20% of aboveground tree biomass and has low correlation with forest age and type since it is strongly influenced by disturbances, generally following a U-shaped curve from high densities at early and late successional stages;

*forest floor (C) ranges widely from <10% to >50% of aboveground tree carbon and is positively related to decomposition recalcitrance of tree material and low temperatures that retard decomposition, thus highest levels are in boreal and montane gymnosperm forest types;

*and mineral soil constitutes 33-66% of forest ecosystem carbon and often represents the largest carbon pool in young forests before aboveground stocks accumulate.

Forest type & data source	Age @ measure-ment	Flux/flow (NPP Mg C/ ha/yr)	Above-ground tree carbon stocks (Mg C /ha)	Below-ground stocks (Mg C/ha)	Lying necromass stocks (Mg C/ha)	Forest floor stocks (Mg C/ha)	Mineral soil stocks (Mg C/ha) (20-100 cm depth)
Boreal forests: Canada & northern U.S.							
Black spruce (Hagemann et al., 2010)	1 yr. post-clearcut		<1		26	31	122
	17 yr. post-clearcut		<1		9	47	208
	35 yr. post-clearcut		9		1	48	186
	21 yr. post-fire		0		14	33	168
	100 yr. old-growth		47		6	40	163
Black spruce-lowland sphagnum, Canada (Gulden et al., 2008)	120		20-60			32-58	150-250
Aspen, USA (Gower et al., 1997)	60		57-93			16-19	32-92
Jack pine, USA (Gower et al., 2001)	65		29-35			11.5-14.6	14.2-25.8
Boreal moist (Keith et al., 2009)	100+ yr. old-growth		64				
Boreal dry (Keith et al., 2009)	100+ yr. old-growth		49				
New England & Great Lakes							

Forest type & data source	Age @ measure-ment	Flux/flow (NPP Mg C/ ha/yr)	Above-ground tree carbon stocks (Mg C /ha)	Below-ground stocks (Mg C/ha)	Lying necromass stocks (Mg C/ha)	Forest floor stocks (Mg C/ha)	Mineral soil stocks (Mg C/ha) (20-100 cm depth)
Northern hardwood, New Hampshire USA (Fahey et al., 2005)	70-100	4.8	95	25	13	30	127
Northern hardwood Michigan, USA (Gough et al., 2008)	85		76	23	4		80
White pine, Ontario, Canada (Peichl and Arain, 2006)	15		40	5			30
	30		52	9			34
	65		100	19			37
Mid-atlantic							
Oak-hickory, Tennessee, USA (Malhi et al., 1999)	55-100	6.5-8.0 (Hanson et al., 2003)	79		10	8-27	7-55
Southeast							
Loblolly pine, USA (Hamilton et al., 2002)	15	7.1	15	10			
Rocky Mountains							
Ponderosa pine, USA (Law et al., 2003)	20	2.1	6				99
	70	4.0	53	17	10		76
	100	4.9	102	33	20		102
	250	3.3	134	42	14		64

Forest type & data source	Age @ measure-ment	Flux/flow (NPP Mg C/ ha/yr)	Above-ground tree carbon stocks (Mg C /ha)	Below-ground stocks (Mg C/ha)	Lying necromass stocks (Mg C/ha)	Forest floor stocks (Mg C/ha)	Mineral soil stocks (Mg C/ha) (20-100 cm depth)
Cool temperate dry forest (Keith et al., 2009)	100+ yr. old-growth		176				
Pacific Northwest							
Fir-spruce-cedar, Oregon, USA (Smithwick et al., 2002)	150		120			10	37
	700		628			19	366
Cool temperate moist forest (Keith et al., 2009)	100+ yr. old-growth		377				
Temperate Mexico							
Montane pine-oak, Mexico (Ordonez et al., 2008)	unknown		92-113	24-29	3-4		93-116
Cool temperate montane forest (Keith et al., 2009)	100+ yr. old-growth		147				
Tropical natural species plantations: Costa Rica							
Monocultures (Piotto et al., 2010)	15	2.5-5.5	31-117				
Mixed species	15	3.6-8.2	56-134				
Tropical natural forests: Venezuela							
Dry tropical (500-1500 mm/yr) (Delaney et al., 1997)	100+		67-72	32-35	2-2.7	5.2	196-271

Forest type & data source	Age @ measure-ment	Flux/flow (NPP Mg C/ ha/yr)	Above-ground tree carbon stocks (Mg C /ha)	Below-ground stocks (Mg C/ha)	Lying necromass stocks (Mg C/ha)	Forest floor stocks (Mg C/ha)	Mineral soil stocks (Mg C/ha) (20-100 cm depth)
Transitional/ seasonal tropical (1500-2000 mm/yr)			80-256	13-41	1.2-4.3	1.6-4	73-153
Moist lowland (2000+ mm/ yr)			165-210	26-34	10.9-27.7	1.8-2.7	132-232
Lower montane			119-260	26-57	10.3-38.3	2.3-4.6	186-319
Upper montane			147-167	32-37	6.7-27.7	2.4-2.9	237-276
Subtropical exotic species plantations: Australia							
Radiata pine plantation, Australia (Ryan et al., 1996)	20	9	59	12			

APPENDIX C

(C) A Case Study of the Carbon Stocks in Various Management Regimes in a Coastal Pacific Northwest Douglas-Fir Forest Type

This appendix provides information on an example plot and how it was processed. Table C-1 shows an example of the yields produced by applying the suite of silvicultural prescriptions to a plot.

A summary of the total carbon yields for one example plot is shown in Figure C-2. Table C-2 shows the average carbon yield over the 100-year period. The effect of silviculture for the example plot shows that any harvesting reduces carbon. The average yield for the clearcuts was 330 t/a (815 t/h), 359 t/a (887 t/h) for VR 15%, 379 t/a (936 t/h) for VR 40% and 775 t/a (1913 t/h) for the "no harvest" grow scenario. Looking across silviculture and considering age of trees when harvested yields an average of 214 t/a (529 t/h) at 30 years, 296 t/a (731 t/h) at 50 years, 434 t/a (1,072 t/h) at 70 years and 480 t/a (1,186 t/h) at 90 years. In this example for one plot, harvest versus no harvest appears to be the biggest factor for carbon yields, followed by stand age at harvest rather than retention via silvicultural treatment. Note that this example is carbon yield, not Climate Action Reserve creditable reductions (since baseline and leakage are not considered), and illustrates how the yields were constructed for a single plot.

Figure C-3 shows the baseline elements for the example plot. The silviculture selected for the baseline will consist of whatever combinations are required to most closely model the long-term common practice average, 24 tonnes per acre (59.3 t/h) in aboveground live tree carbon in this case. The onsite carbon for the baseline was averaged as was the offsite in-use and landfill pools.

Table C-1. Plot 3121 example of yields generated for suite of silvicultural prescriptions.

		Decade										
Prescription	**Variable**	**0**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Grow	Trees per Acre	584	535	486	441	401	366	335	308	284	263	244
	Basal Area (sq. ft/ac.)	229	280	317	344	364	379	390	399	406	411	416
	Avg. Diameter (in)	8.5	9.8	10.9	12.0	12.9	13.8	14.6	15.4	16.2	16.9	17.7
	Board Feet per Acre	34,536	49,177	62,061	72,832	83,034	91,004	99,853	107,285	113,923	120,226	126,451
	Harvested BF/Ac		-	-	-	-	-	-	-	-	-	-
	Cubic Feet per Acre	6,610	9,264	11,469	13,281	14,877	16,164	17,459	18,455	19,427	20,224	20,933
	SDI	448	518	561	588	603	612	616	617	615	613	609
	Live Tree AG C (tonnes/ac.)	54.4	74.3	91.7	106.8	118.9	129.2	137.5	144.6	150.6	155.6	160.1
	Live Tree BG C (tonnes/ac.)	14.5	19.1	23.0	26.4	29.0	31.2	33.0	34.4	35.7	36.8	37.7
	Offsite In-Use (tonnes/ac.)		-	-	-	-	-	-	-	-	-	-
	Dead BG C (tonnes/ac.)		0.6	1.7	2.7	3.6	4.3	4.9	5.2	5.5	5.6	5.8
	Snags C (tonnes/ac.)	0.1	4.3	8.3	12.2	16.1	19.6	21.7	23.7	25.5	27.1	28.7
	Down Dead C (tonnes/ac.)	12.7	13.2	15.5	18.7	21.8	25.4	29.8	33.9	37.8	41.4	44.9
	Floor C (tonnes/ac.)	12.4	13.3	13.5	13.7	13.8	14	14.1	14.3	14.4	14.5	14.7
	Shrub/Herb C (tonnes/ac.)	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
	Total C (tonnes/ac.)	94.3	125.0	154.0	180.7	203.4	223.9	241.2	256.3	269.7	281.2	292.1
	Total CO2e (tonnes/ac.)	346.2	458.7	565.1	663.1	746.5	821.6	885.1	940.7	989.8	1032.0	1071.8
	Offsite Landfill (tonnes/ac.)	-	-	-	-	-	-	-	-	-	-	-
Clearcut Age 30	Trees per Acre	584	23	324	321	43	343	339	80	379	374	353
	Basal Area (sq. ft/ac.)	229	0	6	80	0	8	83	0	10	86	226
	Avg. Diameter (in)	8.5	0.0	1.8	6.8	0.0	2.1	6.7	0.0	2.2	6.5	10.8
	Board Feet per Acre	34,536	-	-	-	-	-	173	-	-	161	24,790
	Harvested BF/Ac		42,553	-	-	11,560	-	-	11,361	-	-	-
	Cubic Feet per Acre	6,610	-	-	-	-	-	31	-	-	29	5,285
	SDI	448	0	22	171	0	27	179	0	32	187	402
	Live Tree AG C (tonnes/ac.)	54.4	0.0	0.0	10.0	0.0	0.0	10.4	0.0	0.0	10.7	40.2
	Live Tree BG C (tonnes/ac.)	14.5	0.0	0.0	3.3	0.0	0.0	3.4	0.0	0.0	3.4	11.1
	Offsite In-Use (tonnes/ac.)		13.4	-	-	4.1	-	-	3.8	-	-	-
	Dead BG C (tonnes/ac.)		16.4	10.6	6.9	13	8.4	5.5	12.1	7.8	5.1	3.4
	Snags C (tonnes/ac.)	0.1	1.6	1	0.7	0.6	0.5	0.4	0.3	0.3	0.2	1.4
	Down Dead C (tonnes/ac.)	12.7	6	4.9	4.3	7.2	5	4	6.8	4.5	3.5	4.1
	Floor C (tonnes/ac.)	12.4	12.1	11.7	12	11.8	11.5	11.8	11.6	11.4	11.7	12.6
	Shrub/Herb C (tonnes/ac.)	0.2	1.1	0.7	0.2	1.1	0.6	0.2	1.1	0.5	0.2	0.2
	Total C (tonnes/ac.)	94.3	50.6	28.9	37.4	37.8	26.0	35.6	35.7	24.5	34.8	73.0
	Total CO2e (tonnes/ac.)	346.2	185.6	106.1	137.3	138.7	95.4	130.8	130.9	89.9	127.7	268.1
	Offsite Landfill (tonnes/ac.)		8.9	-	-	2.7	-	-	2.5	-	-	-
Clearcut Age 50	Trees per Acre	584	23	324	321	305	268	33	334	330	313	274
	Basal Area (sq. ft/ac.)	229	0	6	80	220	318	0	7	83	224	322
	Avg. Diameter (in)	8.5	0.0	1.8	6.8	11.5	14.7	0.0	2.0	6.8	11.5	14.7
	Board Feet per Acre	34,536	-	-	-	24,300	38,625	-	-	133	24,895	39,546
	Harvested BF/Ac		42,553	-	-	-	-	43,761	-	-	-	-
	Cubic Feet per Acre	6,610	-	-	-	5,409	8,887	-	-	24	5,491	8,918
	SDI	448	0	22	171	381	500	0	25	178	389	507
	Live Tree AG C (tonnes/ac.)	54.4	0.0	0.0	10.0	39.7	70.7	0.0	0.0	10.5	40.4	71.2
	Live Tree BG C (tonnes/ac.)	14.5	0.0	0.0	3.3	11.0	18.3	0.0	0.0	3.4	11.2	18.4
	Offsite In-Use (tonnes/ac.)		13.4	-	-	-	-	18.2	-	-	-	-
	Dead BG C (tonnes/ac.)		16.4	10.6	6.9	4.5	3.9	29.8	19.3	12.5	8.2	6.3
	Snags C (tonnes/ac.)	0.1	1.6	1	0.7	1.6	7.1	8.7	6.5	4.9	4.5	9.6
	Down Dead C (tonnes/ac.)	12.7	6	4.9	4.3	4.9	7.6	17.8	12.6	10.4	10.7	13.1
	Floor C (tonnes/ac.)	12.4	12.1	11.7	12	12.8	13.5	12.3	11.9	12.2	13.1	13.8
	Shrub/Herb C (tonnes/ac.)	0.2	1.1	0.7	0.2	0.2	0.2	1.1	0.7	0.2	0.2	0.2
	Total C (tonnes/ac.)	94.3	50.6	28.9	37.4	74.7	121.3	87.9	51.0	54.0	88.2	132.6
	Total CO2e (tonnes/ac.)	346.2	185.6	106.1	137.3	274.3	445.0	322.4	187.2	198.3	323.7	486.7
	Offsite Landfill (tonnes/ac.)		8.9	-	-	-	-	12.1	-	-	-	-
Clearcut Age 70	Trees per Acre	584	535	486	19	320	317	301	265	226	194	20
	Basal Area (sq. ft/ac.)	229	280	317	0	6	79	218	317	369	396	0
	Avg. Diameter (in)	8.5	9.8	10.9	0.0	1.9	6.8	11.5	14.8	17.3	19.3	0.0
	Board Feet per Acre	34,536	49,177	62,061	-	-	-	24,308	38,401	52,955	67,297	-
	Harvested BF/Ac		-	-	67,991	-	-	-	-	-	-	74,000
	Cubic Feet per Acre	6,610	9,264	11,469	-	-	-	5,425	8,869	11,318	13,468	-
	SDI	448	518	561	0	22	170	378	498	544	559	0
	Live Tree AG C (tonnes/ac.)	54.4	74.3	91.7	0.0	0.0	10.1	39.9	70.9	93.7	110.1	0.0
	Live Tree BG C (tonnes/ac.)	14.5	19.1	23.0	0.0	0.0	3.3	11.0	18.3	23.5	27.1	0.0
	Offsite In-Use (tonnes/ac.)		-	-	21.1	-	-	-	-	-	-	26.9
	Dead BG C (tonnes/ac.)		0.6	1.7	24.9	16.1	10.5	6.8	5.4	6.1	7.5	40.3
	Snags C (tonnes/ac.)	0.1	4.3	8.3	7.9	5	3.5	3.5	8.1	15.3	23.2	19.6
	Down Dead C (tonnes/ac.)	12.7	13.2	15.5	9.5	9.3	9	9.8	12.6	17.7	23.2	38.3
	Floor C (tonnes/ac.)	12.4	13.3	13.5	12.4	12	12.2	13	13.7	14.2	14.5	13.1
	Shrub/Herb C (tonnes/ac.)	0.2	0.2	0.2	1.1	0.7	0.2	0.2	0.2	0.2	0.2	1.1
	Total C (tonnes/ac.)	94.3	125.0	154.0	76.9	43.1	48.8	84.2	129.2	170.7	205.8	139.3
	Total CO2e (tonnes/ac.)	346.2	458.7	565.1	282.0	158.2	179.1	309.1	474.2	626.3	755.4	511.3
	Offsite Landfill (tonnes/ac.)		-	-	14.0	-	-	-	-	-	-	17.9

Clearcut Age 90	Trees per Acre	584	535	486	441	401	16	317	314	298	261	222
	Basal Area (sq. ft/ac.)	229	280	317	344	364	0	6	81	222	320	371
	Avg. Diameter (in)	8.5	9.8	10.9	12.0	12.9	0.0	1.9	6.9	11.7	15.0	17.5
	Board Feet per Acre	34,536	49,177	62,061	72,832	83,034	-	-	131	24,930	38,672	53,005
	Harvested BF/Ac		-	-	-	-	87,080	-	-	-	-	-
	Cubic Feet per Acre	6,610	9,264	11,469	13,281	14,877	-	-	38	5,584	8,973	11,409
	SDI	448	518	561	588	603	0	21	172	383	500	546
	Live Tree AG C (tonnes/ac.)	54.4	74.3	91.7	106.8	118.9	0.0	0.0	10.3	40.6	71.5	94.3
	Live Tree BG C (tonnes/ac.)	14.5	19.1	23.0	26.4	29.0	0.0	0.0	3.3	11.2	18.5	23.6
	Offsite In Use (tonnes/ac.)		-	-	-	-	26.2	-	-	-	-	-
	Dead BG C (tonnes/ac.)		0.6	1.7	2.7	3.6	30.7	19.9	12.9	8.4	6.4	6.9
	Snags C (tonnes/ac.)	0.1	4.3	8.3	12.2	16.1	14.2	9.8	6.7	6.1	10.6	18.4
	Down Dead C (tonnes/ac.)	12.7	13.2	15.5	18.7	21.8	12.7	13.3	13.9	14.8	17.4	21.8
	Floor C (tonnes/ac.)	12.4	13.3	13.5	13.7	13.8	12.7	12.3	12.5	13.4	14.1	14.5
	Shrub/Herb C (tonnes/ac.)	0.2	0.2	0.2	0.2	0.2	1.1	0.7	0.2	0.2	0.2	0.2
	Total C (tonnes/ac.)	94.3	125.0	154.0	180.7	203.4	97.6	56.0	59.9	94.7	138.7	179.7
	Total CO2e (tonnes/ac.)	346.2	458.7	565.1	663.1	746.5	358.2	205.5	219.7	347.7	508.9	659.6
	Offsite Landfill (tonnes/ac.)		-	-	-	-	17.5	-	-	-	-	-
VR 15% Age 30	Trees per Acre	584	29	330	324	46	346	340	81	380	373	354
	Basal Area (sq. ft/ac.)	229	38	50	102	36	48	102	36	48	99	207
	Avg. Diameter (in)	8.5	15.5	5.3	7.6	12.0	5.0	7.4	9.0	4.8	7.0	10.4
	Board Feet per Acre	34,536	9,553	12,384	15,042	12,098	14,444	16,705	14,200	16,000	17,513	34,746
	Harvested BF/Ac		34,135	-	-	7,100	-	-	7,439	-	-	-
	Cubic Feet per Acre	6,610	1,644	2,036	2,406	1,904	2,283	2,556	2,150	2,370	2,562	5,712
	SDI	448	58	118	209	62	115	211	68	117	209	374
	Live Tree AG C (tonnes/ac.)	54.4	11.4	14.3	22.3	12.8	14.8	22.7	13.6	14.9	21.9	43.4
	Live Tree BG C (tonnes/ac.)	14.5	3.6	4.5	6.6	4.1	4.6	6.7	4.3	4.6	6.5	11.9
	Offsite In Use (tonnes/ac.)		11.0	-	-	1.9	-	-	2.2	-	-	-
	Dead BG C (tonnes/ac.)		13.7	8.9	5.8	10.9	7.1	4.6	9.9	6.4	4.2	2.9
	Snags C (tonnes/ac.)	0.1	1.6	1	1	1	0.8	0.8	0.9	0.7	0.7	1.8
	Down Dead C (tonnes/ac.)	12.7	11.1	9	7.7	6.5	5.2	4.6	4.7	3.6	3.2	3.8
	Floor C (tonnes/ac.)	12.4	12.2	11.9	12	11.8	11.6	11.8	11.6	11.4	11.6	12.3
	Shrub/Herb C (tonnes/ac.)	0.2	1	0.6	0.2	1.1	0.5	0.2	1.1	0.5	0.2	0.2
	Total C (tonnes/ac.)	94.3	65.6	50.1	55.7	50.1	44.6	51.4	48.3	42.2	48.3	76.3
	Total CO2e (tonnes/ac.)	346.2	240.7	184.0	204.3	183.8	163.5	188.6	177.3	154.7	177.2	280.0
	Offsite Landfill (tonnes/ac.)		7.3	-	-	1.3	-	-	1.5	-	-	-
VR 15% Age 50	Trees per Acre	584	29	330	324	308	277	37	338	332	315	281
	Basal Area (sq. ft/ac.)	229	38	50	102	210	297	36	46	102	213	302
	Avg. Diameter (in)	8.5	15.5	5.3	7.6	11.2	14.0	13.4	5.0	7.5	11.1	14.0
	Board Feet per Acre	34,536	9,553	12,384	15,042	29,883	48,604	13,607	15,076	17,226	35,469	50,283
	Harvested BF/Ac		34,135	-	-	-	-	40,106	-	-	-	-
	Cubic Feet per Acre	6,610	1,644	2,036	2,406	5,281	9,072	2,075	2,301	2,557	5,923	9,150
	SDI	448	58	118	209	369	476	58	110	209	375	485
	Live Tree AG C (tonnes/ac.)	54.4	11.4	14.3	22.3	43.7	68.6	13.5	15.1	22.8	45.1	70.9
	Live Tree BG C (tonnes/ac.)	14.5	3.6	4.5	6.6	12.0	17.8	4.2	4.7	6.7	12.3	18.4
	Offsite In Use (tonnes/ac.)		11.0	-	-	-	-	15.1	-	-	-	-
	Dead BG C (tonnes/ac.)		13.7	8.9	5.8	3.9	3.4	24.2	15.7	10.2	6.8	5.3
	Snags C (tonnes/ac.)	0.1	1.6	1	1	2.1	6.6	7.9	5.4	4	3.8	7.9
	Down Dead C (tonnes/ac.)	12.7	11.1	9	7.7	7.7	9.7	12.5	10.5	9.7	10.5	12.9
	Floor C (tonnes/ac.)	12.4	12.2	11.9	12	12.6	13.2	12.3	11.9	12.1	12.7	13.4
	Shrub/Herb C (tonnes/ac.)	0.2	1	0.6	0.2	0.2	0.2	1.1	0.6	0.2	0.2	0.2
	Total C (tonnes/ac.)	94.3	65.6	50.1	55.7	82.2	119.6	90.9	63.8	65.8	91.4	129.0
	Total CO2e (tonnes/ac.)	346.2	240.7	184.0	204.3	301.6	438.8	333.5	234.2	241.3	335.6	473.3
	Offsite Landfill (tonnes/ac.)		7.3	-	-	-	-	10.1	-	-	-	-
VR 15% Age 70	Trees per Acre	584	535	486	24	324	319	303	272	237	207	24
	Basal Area (sq. ft/ac.)	229	280	317	37	48	104	211	298	350	381	35
	Avg. Diameter (in)	8.5	9.8	10.9	16.8	5.2	7.7	11.3	14.2	16.5	18.4	16.4
	Board Feet per Acre	34,536	49,177	62,061	11,324	13,467	16,296	34,764	49,733	57,177	68,800	15,313
	Harvested BF/Ac		-	-	58,020	-	-	-	-	-	-	60,206
	Cubic Feet per Acre	6,610	9,264	11,469	1,791	2,120	2,550	5,890	9,112	11,481	13,463	2,259
	SDI	448	518	561	55	113	212	369	476	528	549	53
	Live Tree AG C (tonnes/ac.)	54.4	74.3	91.7	12.7	15.0	23.5	45.2	70.6	92.1	108.4	14.8
	Live Tree BG C (tonnes/ac.)	14.5	19.1	23.0	4.0	4.6	6.9	12.3	18.3	23.1	26.7	4.6
	Offsite In Use (tonnes/ac.)		-	-	18.3	-	-	-	-	-	-	22.1
	Dead BG C (tonnes/ac.)		0.6	1.7	22	14.3	9.3	6.2	4.9	5.3	6.4	34.2
	Snags C (tonnes/ac.)	0.1	4.3	8.3	8.1	5.6	4.3	4.5	8.3	14.1	20.8	18.7
	Down Dead C (tonnes/ac.)	12.7	13.2	15.5	16.1	13.9	12.6	12.8	15	19.4	24	25.2
	Floor C (tonnes/ac.)	12.4	13.3	13.5	12.5	12.1	12.3	12.9	13.5	13.9	14.3	13
	Shrub/Herb C (tonnes/ac.)	0.2	0.2	0.2	1.1	0.6	0.2	0.2	0.2	0.2	0.2	1.1
	Total C (tonnes/ac.)	94.3	125.0	154.0	94.8	66.1	69.1	94.1	130.8	168.1	200.8	133.7
	Total CO2e (tonnes/ac.)	346.2	458.7	565.1	348.0	242.7	253.5	345.3	480.1	616.8	737.0	490.6
	Offsite Landfill (tonnes/ac.)		-	-	12.2	-	-	-	-	-	-	14.7

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VR 40% Age 90	Trees per Acre	584	535	486	441	401	26	327	321	309	288	261
	Basal Area (sq. ft/ac.)	229	280	317	344	364	96	107	127	179	248	304
	Avg. Diameter (in)	8.5	9.8	10.9	12.0	12.9	26.0	7.7	8.5	10.3	12.6	14.6
	Board Feet per Acre	34,536	49,177	62,061	72,832	83,034	31,849	37,045	42,025	45,466	56,454	69,625
	Harvested BF/Ac		-	-	-	-	56,920	-	-	-	-	-
	Cubic Feet per Acre	6,610	9,264	11,469	13,281	14,877	5,021	5,784	6,413	6,929	8,981	11,700
	SDI	448	518	561	588	603	121	217	248	324	415	480
	Live Tree AG C (tonnes/ac.)	54.4	74.3	91.7	106.8	118.9	32.5	36.5	40.4	50.4	64.3	79.5
	Live Tree BG C (tonnes/ac.)	14.5	19.1	23.0	26.4	29.0	9.2	10.2	11.2	13.6	16.8	20.3
	Offsite In Use (tonnes/ac.)		-	-	-	-	18.7	-	-	-	-	-
	Dead BG C (tonnes/ac.)		0.6	1.7	2.7	3.6	23.4	15.3	10	6.7	4.9	4.4
	Snags C (tonnes/ac.)	0.1	4.3	8.3	12.2	16.1	14.3	10.3	7.9	7.1	8.7	12
	Down Dead C (tonnes/ac.)	12.7	13.2	15.5	18.7	21.8	21.2	20.1	19.4	19.2	19.8	22.3
	Floor C (tonnes/ac.)	12.4	13.3	13.5	13.7	13.8	12.8	12.5	12.6	12.8	13.2	13.6
	Shrub/Herb C (tonnes/ac.)	0.2	0.2	0.2	0.2	0.2	0.8	0.6	0.3	0.2	0.2	0.2
	Total C (tonnes/ac.)	94.3	125.0	154.0	180.7	203.4	132.9	105.5	101.8	109.9	127.9	152.4
	Total CO2e (tonnes/ac.)	346.2	458.7	565.1	663.1	746.5	487.9	387.0	373.5	403.5	469.4	559.2
	Offsite landfill (tonnes/ac.)		-	-	-	-	12.5	-	-	-	-	-

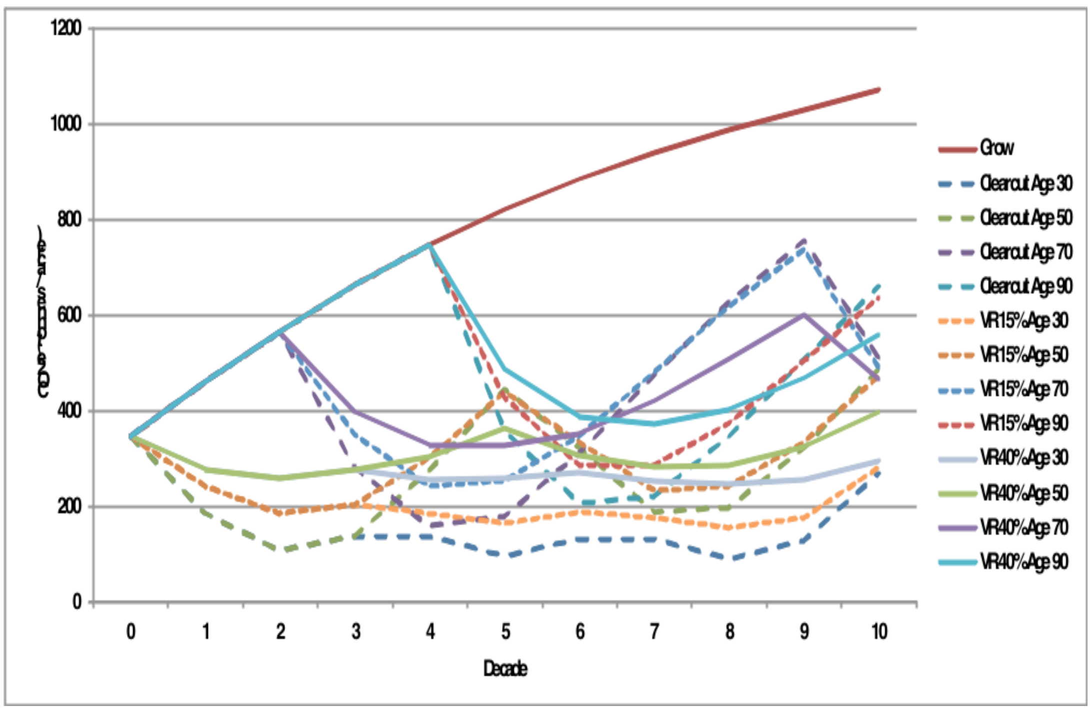

Figure C-2. Total carbon (CO2e) yields (without landfill) for example plot 3121 by silvicultural method.

Table C2. Example of average carbon yield by silviculture for plot 3121.

Prescriptin	Avg. CO2e (tonnes /acre)	Avg. CO2e (tonnes /hectare)
Grow	775	1913
Clearcut Age 30	160	394
Clearcut Age 50	274	677
Clearcut Age 70	424	1048
Clearcut Age 90	462	1140
VR 15% Age 30	209	516
VR 15% Age 50	303	748
VR 15% Age 70	444	1097
VR 15% Age 90	481	1188
VR 40% Age 30	273	675
VR 40% Age 50	312	771
VR 40% Age 70	433	1070
VR 40% Age 90	496	1226

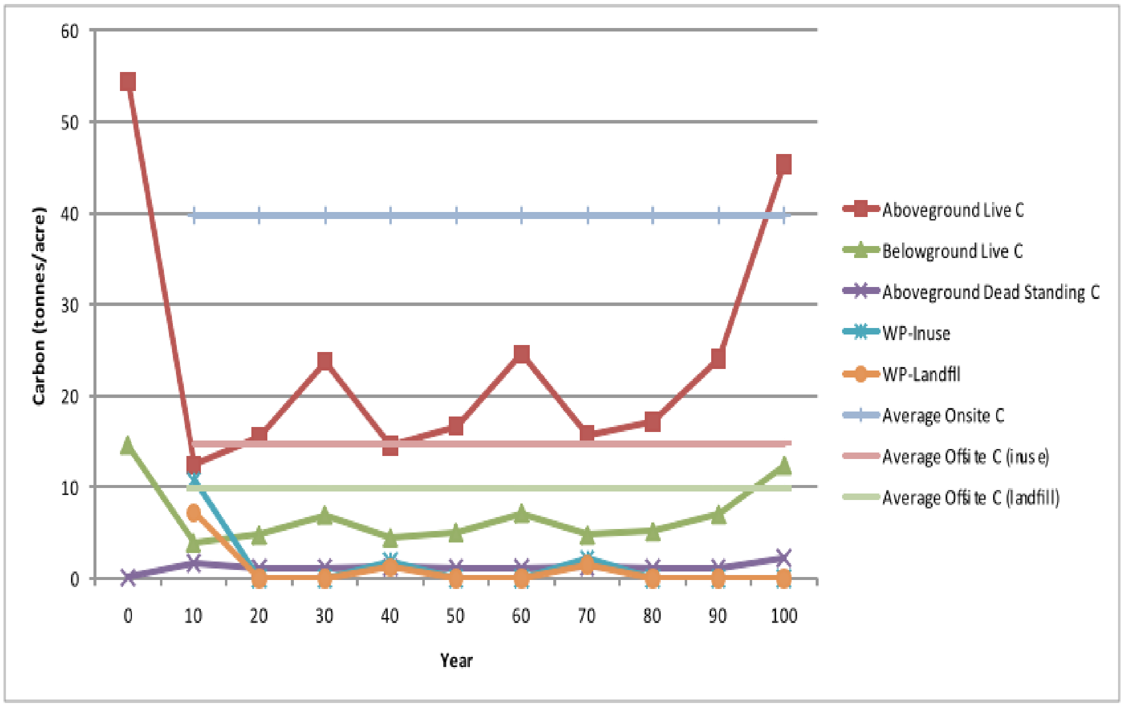

Figure C-3. Baseline pools and averages for example plot 3121.

Figure C-4. Project activity pools for example plot 3121 with clearcut treatments on a 70 year rotation.

WORKS CITED

2010. GIPALS32. Optimalon Software, Kitchener, Ontario.

Angers, et al. 2005. Comparing composition and structure in old-growth and harvested (selection and diameter-limit cut) northern hardwood stands in Quebec. Forest Ecology and Management 217:275-293.

Aubrey, K.B., C.B. Halpern, and C.E. Peterson. Variable-retention harvests in the Pacific Northwest: A review of short-term findings from the DEMO study 2008. Forest Ecology and Management 258:398-408.

Barrett, J.W., ed. 1995. Regional Silviculture of the United States, 3rd ed. New York, NY: John Wiley and Sons.

Best, C. and Wayborn, L.A. 2001. America's Private Forests: Status and Stewardship. Washington, D.C.: Island Press.

Bisbing, S.M., Alaback, P.B., and Deluca, T.H. 2010. Carbon storage in old-growth and second growth fire-dependent western larch forests of the Inland Northwest, USA. Forest Ecology and Management 259:1041-1049.

Borjesson, P. and Gustavsson, L. 2000. Greenhouse gas balances in building construction: wood versus concrete from forest and land-use perspectives. Energy Policy 28:575-588.

Boston, K. and Bettinger, P. 2006. An economic and landscape evaluation of the green-up rules for California, Oregon, and Washington (USA). Forest Policy and Economics 8:251–266.

Bradford, J.B. et al. 2009. Detrital carbon pools in temperate forests. Canadian Journal of Forest Research 39:802-813.

Bradford, J.B. and Kastendick, D.N. 2010. Age-related patterns of forest complexity and carbon storage in pine aspen-birch ecosystems of northern Minnesota, USA. Canadian Journal of Forest Research 40:401-409.

Breiman, L., J. Friedman, C.J. Stone, and R.A. Olshen. 1984. Classification and regression trees. Chapman & Hall/CRC, Boca Raton, FL.

Cairns, M.A., S. Brown, E.H. Helmer, and G.A. Baumgardner. 1997. Root biomass allocation in the world's upland forests. Oecologia 111:1-11.

Chiang, J-M, and Brown, K.J. 2010. The effects of thinning and burning treatments on within-canopy variation of leaf traits in hardwood forests of southern Ohio. Forest Ecology and Management 260:1065-1075.

Davis, S.C. et al. 2009. Forest carbon sequestration changes in response to forest harvest. Forest Ecology and Management 258:2101-2109.

Delaney M., S. Brown, A.E. Lugo, A. Torres-Lezama, and N. Bello Quintero. 1997. The distribution of organic carbon in major components of forests located in five life zones of Venezuela. Journal of Tropical Ecology 13:697-708.

DeLong, D.L. et al. 2005. Survival and growth response of seedlings in root disease infected partial cuts in the Interior Cedar Hemlock zone of southeastern British Columbia. Forest Ecology and Management 206:365–379.

Dias, A.C., Luis, A., and Capela, I. 2007. Carbon dioxide emissions from forest operations in Portuguese eucalypt and maritime pine stands. Scandinavian Journal of Forest Research 22:422-432.

Drever, C.R. and Lertzman, K.P. 2001. Light-growth responses of coastal Douglas-fir and western redcedar saplings under different regimes of soil moisture and nutrients. Canadian Journal of Forest Resources 31: 2124–2133.

Ericsson, E. et al. 2007. Integrated carbon analysis of forest management practices and wood substitution. Canadian Journal of Forest Resources 37:671-681.

Everitt, B.S., and T. Hothorn. 2006. A handbook of statistical analyses using Rpart. Chapman & Hall/CRC, Taylor & Francis Group, Boca Raton, FL. 275 p.

Fahey, T.J. et al. 2005. The biogeochemistry of carbon at Hubbard Brook. Biogeochemistry 75:109-176.

FIA. 2009a. *Regional biomass equations used by FIA to estimate bole, bark and branches* (dated 13 May 2009). USDA Forest Service, Pacific Northwest Research Station.

FIA. 2009b. *Volume estimation for the PNW-FIA Integrated Database* (dated 13 May 2009). USDA Forest Service, Pacific Northwest Research Station.

FIA. 2010. Forest Inventory and Analysis Program. USDA Forest Service, Washington, D.C.

Foley, T.G., Richer, D.B., and Galik, C.S. 2009. Extending rotation age for carbon sequestration: A cross-protocol comparison of North American forest offsets. Forest Ecology and Management 259:201-209.

Franklin, J. et al., 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management 155:399-423.

Gough, C.M., et al. 2008. Controls on annual forest carbon storage: Lessons from the past and predictions for the future. Bioscience 58:609-622.

Gough, C.M. et al. 2010. Wood net primary production resilience in an unmanaged forest transitioning from early to middle succession. Forest Ecology and Management 260:36-41.

Goulden, M.L. et al. 1998. Sensitivity of boreal forest carbon balance to soil thaw. Science 279:214-217.

Gower, S.T. et al. 2001. Net primary production and carbon allocation patterns of boreal forest ecosystems. Ecological Applications 11:1395-1411.

Gulden, J.M. 1991. Uneven-aged BDq regulation of Sierra Nevada mixed conifers. Western Journal of Applied Forestry 6:27-32.

Hagemann, U., Moroni, M.T., and Makeschin, F. 2009. Deadwood abundance in Labrador high-boreal black spruce forests. Canadian Journal of Forest Resources 39:131-142.

Hagemann, U. et al. 2010. Comparing measured and modeled forest carbon stocks in high-boreal forests of harvest and natural-disturbance origin in Labrador, Canada. Ecological Modeling 221:825-839.

Hamilton, J.G. et al. 2002. Forest carbon balance under elevated CO2. Oecologia 131:250-260.

Hansen, A.J. et al. 1995. Alternative silvicultural regimes in the Pacific Northwest: Simulations of ecological and economic effects. Ecological Applications 5:535-554.

Hanson, J.J. and Lorimer, C.G. 2007. Forest structure and light regimes following moderate wind storms: Implications for multi-cohort management. Ecological Applications 17:1325-1340.

Hanson, Jacob J., and Craig G. Lorimer. 2007. Forest structure and light regimes following moderate wind storms: Impliactions for multi-cohort management. Ecological Applications 17:1325–1340

Harmon ME, Harmon JM, Ferrell WK, Brooks D. 1996. Modeling the accumulation of forest products in the Pacific North- west. Clim Change 33:521–50.

Harmon, M.E. and Hua, C. 1991. Coarse Woody Debris Dynamics in Two Old-Growth Ecosystems BioScience 41: 604-610

Harmon, M.E., Moreno, A., and Domingo, J.B. 2009. Effects of partial harvest on the carbon stores in the Douglas-fir/western hemlock forests: A simulation study. Ecosystems 12:777-791.

Harmon, M.E. and Marks, B. 2002. Effects of silvicultural practices on carbon stores in Douglas-fir – western hemlock forests in the Pacific Northwest, U.S.A.: results from a simulation model. Canadian Journal of Forest Resources 32: 863–877.

Hennigar, C.R., MacLean, D.A., and Amos-Binks, L.J. 2008. A novel approach to optimize management strategies for carbon stored in both forests and wood products. Forest Ecology and Management 256:786-797.

Hoover, C. and S. Stout. 2007. The Carbon Consequences of Thinning Techniques: Stand Structure Makes a Difference. Journal of Forestry:266-270.

Hurteau, M. and North, M. 2009. Fuel treatment effects on tree-based forest carbon storage and emissions under modeled wildfire scenarios. Frontiers of Ecology and Environment 7(8): 409–414.

Jandl et al. 2007. How strongly can forest management influence soil carbon sequestration? Geoderma 137:253-268.

Johnson, D.W. and P.S. Curtis. 2001. Effects of forest management on soil carbon and nitrogen storage: Meta analysis. Forest Ecology and Management 140:227-238.

Keith, H., Mackey, B.G., and Lindenmayer, D.B. 2009. Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon dense forests. PNAS 106:11635-11640.

Keyser, C.E. 2010. Pacific Northwest Coast (PN) Variant Overview – Forest Vegetation Simulator. Internal Rep. U. S. Department of Agriculture, Forest Service, Forest Management Service Center. 49.

Keyser, T.A. 2010. Thinning and site quality influence aboveground tree carbon stocks in yellow-poplar forests of the southern Appalachians. Canadian Journal of Forest Resources 40:659-667.

Kranabetter, JM. 2009. Site carbon storage among productivity gradients of a late seral southern boreal forest. Canadian Journal of Forest Research 39:1053-60.

Kranabetter, JM and Macadam, AM. 2007. Changes in carbon storage of broadcast burn plantations over 20 years. Canadian Journal of Soil Science 87:93-102.

Laiho, R. and C. Prescott. 2004. Decay and nutrient dynamics of coarse woody debris in northern gymnosperm forests. Canadian Journal of Forest Resources 34:763-777.

Law, B.E. et al. 2003. Changes in carbon storage and fluxes in a chronosequence of ponderosa pine. Global Change Biology 9:510-524.

Li, Q. et al. 2007. Effects of timber harvest on carbon pools in Ozark forests. Canadian Journal of Forest Research 37:2337-2348.

Lippke, B., Wilson, J., Perez-Garcia, J., Bowyer, J. and Meil, J. 2004. CORRIM: Life-Cycle Environmental Performance of Renewable Building Materials. Forest Products Journal 54(6); 8–19

Lohr S.M., Gauthreaux S.A., Kilgo J.C.. 2002. Importance of coarse woody debris to avian communities in loblolly pine forests. Conserv. Biol. 16: 767–777.

Lorimer, C.G. and White, A.S. 2003. Scale and frequency of natural disturbances in the northeastern U.S.: Implications for early successional forest habitats and regional age distributions. Forest Ecology and Management 185:41-64.

Luyssaert, s. Et al. 2008. Old growth forests as global carbon sinks. Nature 455:213-215

Malhi, Y., Baldocchi, D.D. and Jarvis, P.G. 1999. The carbon balance of tropical, temperate and boreal forests. Plant, Cell and the Environment 22:715-740.

Martin, J.L. et al. 2005. Carbon pools in a boreal mixed-wood logging chronosequence. Global Change Biology 11:1883-1894.

Maser, C., R.G. Anderson, K. Cromack Jr., J.T. Williams and R.E. Martin. 1979. Dead and down woody material. In J.W. Thomas, ed. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Washington, D.C.:USDA Forest Service Agricultural Handbook No. 553.

McLaughlin, J.W. and S.A. Philips. 2006. Soil carbon nitrogen and base cation cycling 17 years after whole-tree harvesting in a low-elevation red spruce-balsam fir forested watershed in central Maine, USA. Forest Ecology and Management 222:234-253.

McNab, W.H., D.T. Cleland, J.A. Freeouf, J.E. Keys Jr., G.J. Nowacki, and C.A. Carpenter. 2005. Description of ecological subregions: sections of the conterminous United States [CD-ROM]. U.S. Department of Agriculture, Forest Service. 80.

Minkkinen, K., Korhonen, R., Savolainen, I. and Laine, J. 2002. Carbon balance and radiative forcing of Finnish peatlands 1900-2100- impact of forestry drainage. Global Change Biology 8:785-799.

Mitchell, A.K. et al. 2007. Regenerating montane conifers with variable retention systems in a coastal British Columbia forest: 10-year results. Forest Ecology and Management 246:240-250.

Munsell, J.F. and R.H. Germain. 2007. Woody biomass energy: An opportunity for silviculture on NIPF in New York. Journal of Forestry 105:398-402.

Murthy, S.K. 1998. Automatic construction of decision trees from data: a multi-disciplinary survey. Data Mining and Knowledge Discovery 2:345-389.

Neilson, E.T. et al. 2007. Spatial distribution of carbon in natural and managed stands in an industrial forest in New Brunswick, Canada. Forest Ecology and Management 253:148–160

Neilson, E.T. et al. 2008. Optimal on- and off-site forest carbon sequestration under existing timber supply constraints in northern New Brunswick. Canadian Journal of Forest Resources 38:2784-2796.

Newsome, T.A. 2010. Ten-year regeneration responses to varying levels of over-story retention in two productive southern British Columbia ecosystems. Forest Ecology and Management 260:132–145.

Noormets, A., Chen, J. and Crow, T.R. 2007. Age-Dependent changes in ecosystem carbon fluxes in managed forests in northern Wisconsin, USA. Ecosystems 10:187-203.

Nunery, J.S. and Keeton, W.S. 2010. Forest carbon storage in the northeastern United States: Net effects of harvesting frequency, post-harvest retention, and wood products. Forest Ecology and Management 259:1363–1375.

Oliver, D.C. and B.C. Larson. 1996. Forest Stand Dynamics. New York, NY: Wiley.

Ordonez, J.A.B. et al. 2008. Carbon content in vegetation, litter, and soil under 10 different land-use and land-cover classes in the Central Highlands of Michoacan, Mexico. Forest Ecology and Management 255:2074-2084.

Peichl, M. and Arain, A.A. 2006. Above- and belowground ecosystem biomass and carbon pools in an age-sequence of temperate pine plantation forests. Agricultural and Forest Meteorology 140:51-63.

Peng, C. et al. 2002. Effects of harvesting regimes on carbon and nitrogen dynamics of boreal forests in central Canada: A process model simulation. Ecological Modelling 155:177-189.

Perez-Garcia, J. et al. 2005. An assessment of carbon pools, storage, and wood products market substitution using life-cycle analysis results. Wood and Fiber Science 37:140-148.

Piotto, D. et al. 2010. Silvicultural and economic aspects of pure and mixed native tree species plantations on degraded pasturelands in humid Costa Rica. New Forests 39:369-385.

Puettmann, M.E. et al. 2010. Cradle-to-gate life-cycle inventory of U.S. wood products. Wood and Fiber Science 42:15-28.

Rebain, S.A. 2010. The Fire and Fuels Extension to the Forest Vegetation Simulator: Updated Model Documentation. Internal Rep. U. S. Department of Agriculture, Forest Service, Forest Management Service Center. 361.

Ribe, R.G. 2009. In-stand scenic beauty of variable retention harvests and mature forests in the U.S. Pacific Northwest: The effects of basal area, density, retention pattern, and down wood.   Journal of Environmental Management 91:245-260

Rosenvald, R. and A. Lohmus. 2008. For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects 2008. FEM 255:1-15.

Ryan, M.G. et al. 1996. Foliage, fine-root, woody-tissue and stand respiration in Pinus radiata in relation to nitrogenn status. Tree Physiology 16:333-343.

Seedre, M. and Chen, H.Y.N. 2010. Carbon dynamics of aboveground live vegetation of boreal mixedwoods after wildfire and clearcutting. Canadian Journal of Forest Resources 40:1862-1869.

Seymour, R.S. and Hunter, M.L. 1998. Principles of Ecological Forestry. New York, NY: Wiley.

Siitonen, J. et al. 2000. Coarse woody debris and stand characteristics in mature managed and old-growth boreal forests in southern Finland. Forest Ecology and Management 128:211-225.

Slesak, R.A., Schoenholtz, S.H., and Harrington, T.B. 2010. Soil respiration and carbon responses to debris and competing vegetation. Forest, Range and Wildland Soils 74:936-946.

Smith, D.M. et al. 1997. The Practice of Silviculture: Applied Forest Ecology. New York, NY: John Wiley and Sons.

Smithwick, E.A. et al. 2002. Potential upper bounds of carbon stores in forests of the Pacific Northwest. Ecological Applications 12:1303-1317.

Taylor, A.R., Wang, J.R., and Kurz, W.A. 2008. Effects of harvesting intensity on carbon stocks in eastern Canadian red spruce forests: An exploratory analysis using the CBM- CFS3 simulation model. Forest Ecology and Management 255:3632-3641.

Therneau, T.M., B. Atkinson, and B. Ripley. 2007. The Rpart package.

Thorpe, H.C., Thomas, S.C. and Caspersen, J.P. 2008. Tree mortality following partial harvests is determined by skidding proximity. Ecological Applications 18:1652-1663.

Webster, C.R. and Lorimer, C.G. 2005. Minimum opening sizes for canopy recruitment of midtolerant tree species: A retrospective approach. Ecological Applications 15:1245-1262.

Whitman, A.A. and J.M. Hagan. 2007. An index to identify late-successional forest in temperate and boreal zones. Forest Ecology and Management 246:144-154.

Woodall, C.W. and Westfall, J.A. 2009. Relationships between the stocking levels of live trees and dead tree attributes in forests of the United States. Forest Ecology and Management 258:2602-2608