# CARBON STOCKS OF WESTERN REDCEDAR AND WESTERN HEMLOCK STANDS IN CANADA'S INLAND TEMPERATE RAINFORESTS

by

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# Abstract

This study was conducted to fill a knowledge gap in forest ecosystem carbon (C) stocks for managed/harvested and unmanaged old-growth stands of Interior Cedar Hemlock (ICH) biogeoclimatic zone forests, also known as 'Inland Temperate Rainforests' (ITRs), in central British Columbia (BC). Carbon stocks of live tree (with accounting for heart-rot and hollow) and dead organic matter (snag, coarse woody debris, and forest floor excluding mineral soil) were quantified in three study sites designated as ICHwk3 or ICHvk2. The C stocks were evaluated among stands treated with three different harvesting methods: clearcutting (CC), group retention (GR, 30% retention), group selection (GS, 70% retention), and uncut (UN, 100% retention). Mean total forest C stocks (excluding mineral soil) in oldgrowth stands were  $455 \pm 156$  Mg C ha<sup>-1</sup> ( $\pm 95\%$  confidence interval) similar to regional average forest C stocks for the Pacific Northwestern USA, demonstrating the important C reservoirs in this ecosystem despite the high incidence of heart-rot in cedar. Live-tree and dead-organic-matter C stocks accounted for 76 and 24% of the total C, respectively. Tree biomass allometric equations were the largest contributor to total uncertainty in live-tree C stocks. This indicates the need to develop ITR-specific tree allometric equations for more accurate assessment of live-tree C. Old ITRs were found to be vulnerable to intensive harvesting (CC and GR), losing the total C stocks of 78 and 65% below those of uncut oldgrowth stands. In contrast, low-intensity harvesting (GS) reduced the C stocks of only 13% below those of uncut old-growth stands and thus provides a good compromise between forest harvesting to provide wood products and maintenance of forest C stocks at all sites. High spatial variability observed in the total forest C stock also stresses the importance of recognizing landform-related productivity gradients (i.e., toe slopes) in the old ITRs. In the face of continued greenhouse gas accumulation in the atmosphere and attendant climate

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change, and the related need to move toward low-carbon economies, managing old ITRs to both conserve and sequester C may call for an approach that has an added benefit of conserving both structural and biological diversity in the remaining old-growth ITRs.

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# Glossary

CC:	Clear-cut						
CV:	Coefficient of variation						
CWD:	Coarse woody debris						
CWH:	Coastal Western Hemlock						
DBH:	Diameter-at-breast-height						
DOM:	Dead organic matter (including snags, CWD, and forest floor)						
FAO:	Food and Agriculture Organization						
GHG:	Greenhouse gas						
GR:	Group retention						
GS:	Group selection						
ICH:	Interior Cedar Hemlock						
ICHvk2, wk3	ICH for very, wet cool subzone (vk2); wet, cool subzone (wk3)						
IPCC:	Intergovernmental Panel on Climate Change						
ITR(s):	Inland temperate rainforest(s)						
Mg:	Unit: Mega = $10^6$ , Mg = tonne						
NPP:	Net primary productivity						
UN:	Uncut (old-growth stands)						
UNFCCC:	United Nations Framework Convention on Climate Change						

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## **Chapter 1. Introduction**

## 1.1 Background

Old-growth forests in temperate and some boreal regions are important in the maintenance of global C (Luyssaert et al. 2007; Luyssaert et al. 2008; Lichstein et al. 2009) and biodiversity (Mackey et al. 2008; Bradshaw et al. 2009; Stevenson et al. 2010). Of the 1.2 billion ha of global primary forest (forests with no clear human intervention including all age classes) (FAO 2010), old-growth forests (including both managed and unmanaged forests) under natural disturbance regimes constitute about 20% of temperate and boreal coniferous forests (Wirth 2009). Old-growth forests in the Pacific Northwest are distinctive in their ecologically complex and diverse functions, and horizontal and vertical structures represented by gap phase dynamics, as stands develop over time (Franklin et al. 2002; Daniels 2003; Wirth et al. 2009). The complexity of old-growth forest is characterized by the presence of old trees (> 200 years old), considerable amounts of large pieces of dead wood (Harmon et al. 1986; Stevenson et al. 2006), and unique understory environment, including light and vegetation dynamics (Harmon et al. 1986; Messier et al. 2009), all of which have values for storing long-term C (Franklin et al. 2002).

## Paradigm shift in roles of old-growth forest in global carbon

Carbon stocks in old-growth forests have traditionally been considered to be in a steady-state (Luyssaert et al. 2008; Mackey et al. 2008; Schulze et al. 2009), as ecosystem theory provided that C uptake should be balanced by C loss in late successional stages (Odum 1969; Gower et al. 1996). The long-standing view of C steady-state in old-growth forests originated from a comparison of energy flow during stand succession (i.e., the balance between photosynthesis, respiration, and biomass) in a forest and a laboratory environment (Odum 1969), and was tested and supported based on subsequent empirical and modelling studies after they revealed the decline in net primary production with stand ages (Gower et al. 1996; Binkley et al. 2002). Although this assumption was based on even-aged forests only, it has been extensively used to represent uneven-aged old-growth forests. World old-growth forests that are often uneven aged (Schulze et al. 2009) are therefore potentially undervalued in the global C cycle. For example, current international negotiations have not integrated unmanaged forests including old-growth forests into a regulatory framework under the UNFCCC (United Nations Framework Convention on Climate Change) reporting system to mitigate rising  $CO_2$  levels (Mackey et al. 2008). Only since the 2007 UNFCCC in Bali have international communities identified the need to account for the role of old-growth forests in accumulating C (Mackey et al. 2008). This suggests that large knowledge gaps currently exist for their roles in the global C cycle.

The conventional view of C steady-state in old-growth forests has recently been challenged by many researchers. Old-growth forests in temperate and, to a lesser extent, boreal biomes have been found to sequester about  $1.3 \pm 0.5$  Gt (gigatonne is equivalent to  $10^9$ tonne) C per year (Luyssaert et al. 2008). Despite some evidence of large spatial and temporal variability in C exchange in these old forests, the growing body of new empirical evidence clearly shows that broadleaved and coniferous old-growth forests continue to be C sinks and thus play key roles in both carbon storage and carbon sequestration (Carey et al. 2001; Pregitzer and Euskirchen 2004; Zhou et al. 2006; Luyssaert et al. 2008; Bois et al. 2009; Gleixner et al. 2009; Keith et al. 2009; Knohl et al. 2009; Lewis et al. 2009; Schulze et al. 2009; Jonsson and Wardle 2010). Recent studies also found that conventional age-related decline in forest productivity was not so much universal as exceptional after aboveground

live-tree biomass in US old-growth temperate forests was re-examined (Lichstein et al. 2009; Wirth 2009). Instead, age-related decline in net primary productivity (NPP) is typically driven by gap formation, the same process that was responsible for C sequestration through continuous establishment of new cohorts in forest gaps (Keith et al. 2009). Thus, the complex structural properties of old-growth forests provide high spatial variability in forest ecosystem C stocks as well as high spatially averaged total forest C stocks (Mackey et al. 2008; Kranabetter 2009).

# Dead organic matter carbon

Dead organic matter (DOM) is important in regulating total ecosystem C stocks in old-growth forests (Sollins et al. 1987; Harmon et al. 1990; Chen and Klinka 2003; Harmon et al. 2004; Luyssaert et al. 2008). Coarse woody debris (CWD) is a major component of dead-organic and total C pools (Harmon 2009) in old-growth forests. Large CWD in old temperate forests, for example, remained for over 250 years (Maser et al. 1979; Stevenson et al. 2010) due to the cool climate and low decomposition rates, both of which are generally responsible for its accumulations in temperate rainforests in the Pacific Northwestern USA and BC (Spies and Cline 1989). Decaying wood is also a major component of forest-floor C, accounting for up to 60 to 70% of the forest floor in old boreal and temperate forests (Keenan et al. 1993; Trofymow and Blackwell 1998; Laiho and Prescott 2004). Forest floors are involved in key ecological processes such as nutrient retention and release, water retention, and pH buffering that directly or indirectly affect forest C stocks (Tietema et al. 1992; Currie 1999). Conditions that are not conductive to soil microbial activities such as low pH, sandy texture, and low nutrient availability can often lead to a thick forest floor (Vesterdal et al. 1995; Six et al. 2002) and important C reservoirs (Currie et al. 2003).

Mineral soil C comprises up to 40% of C stored in boreal, temperate, and tropical forests (Pregitzer and Euskirchen 2004; FAO 2010). Past studies have reported large reservoirs of mineral soil C in old-growth forests of the Pacific Northwestern USA (Currie et al. 2003; Harmon et al. 2004; Sun et al. 2004; Bisbing et al. 2010) and coastal and interior western redcedar and hemlock forests (Trofymow and Blackwell 1998; Trofymow et al. 2008; Stevenson et al. 2010)(Jull et al. 2002; Feller 2003; Stevenson et al. 2010). A global meta-analysis found that mineral soil C increased with stand age in boreal and temperate forests with the oldest stands showing the greatest soil C (Pregitzer and Euskirchen 2004). Despite large spatial variability in soil C reported in old-growth forests due partly to spatial heterogeneity in soil conditions (Bauhus 2009), mineral soil C contributes to C sinks in these old forests.

# Past findings of management impact on carbon stocks in old-growth forest

Forest ecosystem C can decline by up to nearly 60% in response to clear-cut harvesting relative to old-growth stands because of the significant reduction in aboveground live-tree C (Harmon et al. 2004; Fredeen et al. 2005; Gough et al. 2008; Bisbing et al. 2010). Prolonged recovery time for restoring C has been reported after intensive harvesting (Harmon et al. 2004; Trofymow et al. 2008). It has been increasingly realized that the initial loss of C stocks in old-growth forests may never be recovered by C uptake in subsequent second-growth forests (Harmon and Marks 2002; Harmon et al. 2009). Researchers are therefore exploring the potential of partial harvesting as an alternative to clear-cutting in sequestering and storing C (Liski et al. 2001; Harmon and Marks 2002; Taylor et al. 2008; Harmon et al. 2009). Although the effect of harvesting on soil C is not fully understood due to its high spatial and temporal variability (Yanai et al. 2003; Sun et al. 2004; Nave et al.

2010), recent studies illustrated that forest-floor C was more consistently reduced after harvesting than mineral soil C (Bois et al. 2009; Nave et al. 2010). Past modelling studies have consistently reported the potential of partial harvesting for balancing long-term C stocks and increasing C sink strength and timber production (Harmon 2009; Taylor 2008, Swanson 2009). Therefore, if forests are to be managed to sequester and store C, their responses to different silvicultural systems need to be explored.

# Current knowledge gaps in the assessment of forest carbon in old inland temperate rainforests of British Columbia

Despite detailed assessments of forest C stocks in temperate old-growth forests of the Pacific Northwestern USA (Harmon et al. 1990; Smithwick et al. 2002; Harmon et al. 2004; Bisbing et al. 2010), Australia (Turner and Lambert 1986; Turner et al. 1992; Mackey et al. 2008; Keith et al. 2009), and New Zealand (Silvester and Orchard 1999; Davis et al. 2003), there is little understanding of present forest C stocks in old inland temperate rainforests (ITRs) of BC, Canada. Inland temperate rainforests dominated by old western redcedar (*Thuja plicata* Donn) and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) occur in wet and very wet cool parts of interior BC (Arsenault and Goward 2000). The inland rainforests are globally unique, characterized by both maritime and continental climate that provide distinctive ecosystem features relative to coastal temperate rainforests (Stevenson et al. 2010). These forests are not only similar in complexity of forest structure to coastal counterparts (Goward and Spribille 2005), but also share the prevalence of large cedar and hemlock trees. ITRs may therefore store large reservoirs of C in both vegetation and soils. Several studies have quantified C stocks in old-growth stands of western redcedar and western hemlock in coastal BC (Trofymow et al. 2008; Trofymow and Blackwell 1998; Keenan et al. 1993), but no attempts have been made to quantify forest C stocks in ITRs.

Accurate assessment of forest C stocks in old-growth such as ITRs remains challenging due to exceptionally large trees with high incidence of heart-rot (Kloeppel et al. 2007) that has been documented over a wide range of diameter classes, especially in western redcedar (Buckland 1946; Daniels 2003; Gavin et al. 2009). Previous tree-biomass C estimates, particularly in old-growth forests, may have the potential of systematic, large-scale overestimation by not accounting for internal decay. For example, there was little or no information provided concerning how internal decay in trees was accounted for in previous biomass estimates (Kloeppel et al. 2007; Keith et al. 2009). Moreover, estimates of tree C often rely on allometric equations that relate DBH and/or height to tree biomass, but the equations that cover DBH of exceptionally large trees are scarce (Kloeppel et al. 2007). Nevertheless, such trees with DBH greater than the upper diameter limit for existing allometric equations often contribute a large proportion to total forest biomass (Brown and Lugo 1992). Therefore, biomass estimates of very large DBH trees may be highly inaccurate.

In addition, researchers often report model errors in allometric equations when they are developed. Nonetheless, the model errors are often not integrated into tree biomass estimation when the equations are used (Yanai et al. 2010), which does not allow other researchers or forest practitioners to properly evaluate the reliability of tree biomass. By providing the information on the uncertainties in the biomass model error and heart-rot, researchers are better able to judge the reliability of C stock estimates, make a good comparison among studies, and prioritize efforts to improve the accuracy of future inventory estimations (IPCC 2000; Woolley et al. 2007; Yanai et al. 2010).

Concerning forest management, there have been no studies that examined the effect of harvesting on forest ecosystem C stock in old ITRs. Intensive management practices in the late 20<sup>th</sup> and to a lesser extent early 21<sup>st</sup> century have converted an ICH biogeoclimatic zone dominated by primary old-growth cedar-hemlock leading forests to one increasingly populated by young hybrid white spruce (Picea glauca [Moench] Voss) and lodgepole pine (Pinus contorta Dougl. ex Loud. var. latifolia Engelm.) plantations (Stevenson et al. 2010). During the mid-1970s to early 1990s, harvest volumes of western redcedar and western hemlock peaked at 2.5 to 3 million m<sup>3</sup> yr<sup>-1</sup> in central and southern BC. Current annual harvest volumes have declined by a third (to approximately 1.1 million  $m^3 yr^{-1}$ ) due to increasing public environmental awareness, establishment of legal objectives for the retention of old forests (Ministry of Forests and Range 2008) and increased salvage-logging of mountain-pine-beetle attacked trees (Pedersen 2004). It is possible that harvesting of old cedar-hemlock stands may return to former levels, though perhaps not to former peak levels, once salvage logging of mountain-pine-beetle affected stands abates (Stevenson et al. 2010). In an era of concern over greenhouse gases, it is clear that we should understand the way in which forest C stocks respond to harvesting our carbon rich old-growth ITRs.

#### 1.2 Rationale and objectives

This research addresses the lack of baseline information regarding forest C stocks in Canada's old-growth ITRs and impacts of harvesting on these stocks. Reliable estimates of the forest C stocks using field-based and existing data-based measurements in old ITRs contribute to filling a knowledge gap in ongoing local and regional C inventories, and are also essential to properly value C stored in these old-growth forests. Evaluation of forest C stocks in response to types of variable-retention harvesting is also required to develop forest

management strategies aimed at mitigating C emissions and increasing C sequestration. Furthermore, C stock information can be used for validating and developing more robust forest C models necessary for both managed and unmanaged forests in the longer-term and at broader spatial scales. Given the ongoing debate over the definition of old-growth forests, old-growth in this study is defined, according to Arsenault (2003) and Wirth (2009): to be relatively old (> 200 years old, dominated by large old late-successional species) and to have structural and compositional features observed through gap-phase dynamics (uneven-aged, large snags and logs in varying stages of decay, presence of canopy gaps).

The objectives of this study were to (*i*) obtain forest C stock information for old western-redcedar and western-hemlock stands of Canada's ITRs in central BC (i.e., trees, snags, CWD, and forest floor excluding mineral soil); (*ii*) evaluate impacts of various retention-harvesting practices on forest C stocks of old-ITRs; (*iii*) incorporate hollow and decay densities of vertical heart-rot of western redcedar and western hemlock for accurate assessment of tree C stocks; and (*iv*) make forest management recommendations to optimize or maximize ITR C stocks. Chapter 2. Quantifying C stocks in western redcedar and western hemlock stands and assessment of harvesting on the C stocks.

#### 2.1 Abstract

Post-harvest C stocks of live trees and dead organic matter (snags, CWD, and forest floor) were estimated and compared to assess various retention-harvesting practices in oldgrowth stands of the Interior Cedar Hemlock (ICH) biogeoclimatic zone in central British Columbia (BC). Mineral soil C was not included. Mean total forest and dead-organic-matter C stocks in old-growth stands were  $455 \pm 156$  and  $107 \pm 19$  Mg C ha<sup>-1</sup> ( $\pm 95\%$  confidence intervals), respectively, and the total forest C stocks were not significantly influenced by the properties of hollow stems occupied by decayed wood based on a sensitivity analysis. Live trees, snags, coarse woody debris (CWD), and forest floor accounted for 76, 5, 9, and 10% of the total forest C stocks, respectively. Decaying wood constituted 31 to 44% of the forestfloor C, demonstrating the important roles of CWD in dead-organic-matter C. Old ITRs are important C reservoirs despite a high incidence of heart-rot, but analysis of the contribution of different components to uncertainty identified the need to develop ITR-specific tree allometric equations for more accurate assessment of live-tree C. Significant reduction in total forest C after intensive harvesting (CC and GR) exhibited the potential loss of total C and sink strength of live trees in the long-term. In contrast, low-intensity harvesting (GS) has the potential of maintaining long-term total forest C through sustaining the productivity of the forest.

# 2.2 Methods

# 2.2.1 Study area

This study took place in three pre-existing study sites: East Twin Creek, Lunate Creek, and Minnow Creek. All the sites were established between 2000 and 2001, as a long-term experiment to compare alternative silvicultural systems in BC wetbelt forests (Jull et al. 2002). The study sites were located between the communities of Prince George and McBride of east-central BC in the Interior Cedar-Hemlock Biogeoclimatic Zone (ICH) (Ketcheson et al. 1991), also known as inland temperate rainforest (Stevenson et al. 2010), for wet cool (ICHwk3) and very wet cool (ICHvk2) subzones (Figure 1, Table 1) (Meidinger and Pojar 1991). Inland temperate rainforests including the study sites are characterized by both maritime and continental climate, which exhibits more pronounced seasonal temperature differences than coastal counterparts (Stevenson et al. 2010). At the same time, maritime air masses from the west deposit moisture in the ITRs as they are pushed over the mountains.



Figure 1. Location of three study areas (Lunate Creek, East Twin Creek, and Minnow Creek) based on Jull et al. (2002).

Estimated mean annual temperature at Lunate Creek is 1.9 to 2.8°C, while at the East Twin Creek and Minnow Creek sites it ranges from 2.3 to 3.2°C (Wang et al. 2006). Estimated mean annual precipitation at the East Twin Creek, Lunate Creek and Minnow Creek sites is 1169, 939, and 1120 mm, respectively (Wang et al. 2006). The physical characteristics of the study sites are summarized in Table 1.

Old-growth stands at the three study sites are dominated by western redcedar and western hemlock, with forests being approximately 300 to 350 years old at East Twin and

Minnow Creek and 450 to 500 years old at Lunate Creek. Radiocarbon dating of surficial charcoal layers in the forest floor indicated that East Twin and Minnow Creek burned approximately 400 to 450 years ago (Sanborn et al. 2006). Although no surficial charcoal layers in the forest floor were found for radiocarbon dating at Lunate Creek (Jull et al. 2002), stand structure and relative tree size at this site suggest that the stand is considerably older. A higher percentage of species composition is western redcedar and western hemlock at the East Twin and Lunate Creek sites (about 88 to 98%) compared with the Minnow Creek site (48 to 75%). Also, these species account for a higher proportion of the basal area (m<sup>2</sup> ha<sup>-1</sup>) (western redcedar, 83 – 97%; western hemlock, 1 – 11%) at the East Twin Creek and Lunate Creek sites than at the Minnow Creek site (western redcedar, 62 – 82%; western hemlock, 1 – 7%). The remaining species include hybrid white spruce (*Picea glauca* (Moench) Voss x *engelmanii* Parry ex Engelm.) and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.). Podzolic soils predominate in coarse-textured parent materials, while Luvisols have formed in fine-textured parent materials at all sites. Humus forms were predominantly mormoder and humimor and/or hemimor (Jull et al. 2002).

Study area	Elevation (m)	Aspect	Area (ha)	Latitude (N) / longitude (W)	Approximate stand age	Subzone and variant
East Twin Creek	900-1050	NW	26.0	53°28'46"/120°21'07"	300 - 350	ICHwk3
Lunate Creek	950-1200	N	72.4	53°49'53"/121°28'44"	450 - 500	ICHvk2
Minnow Creek	1050-1200	SW	39.2	53°27'56″/120°21'02″	300 - 350	ICHwk3

Table 1. Site characteristics of three Interior Cedar-Hemlock (ICH) study areas.

## 2.2.2 Experimental design

The experimental design was a randomized complete or incomplete block with three or four silvicultural treatments randomly assigned to each block (Figure 2; Jull et al. 2002). The treatments were 0% post-harvest retention or "clearcut" (CC), 30% post-harvest retention or "group retention" (GR), 70% post-harvest retention or "group selection" (GS), and 100% post-harvest retention or "uncut" (UN). East Twin Creek is treated as a randomized incomplete block because of the absence of the GR treatment, while both Lunate Creek and Minnow Creek study sites are randomized complete blocks (Figure 2).

The following summarizes the field operations for the original experimental layout that were conducted between 2000 and 2001 and are described in the Northern Wetbelt Silvicultural Systems Project report (Jull et al. 2002). All harvesting was conducted in the winter on snowpack at all three study sites. East Twin Creek was harvested in March 2000, and the other two sites were harvested in January to March 2001. Ground-based harvesting methods such as hand- or machine-felling with feller bunchers were utilized at all sites with the exception of East Twin Creek where high-lead cable yarding was used due to steep slopes. Post-harvest large-tree permanent sample plots (about 0.10 ha rectangular plots) were established after logging for the purpose of long-term stand monitoring in each treatment unit and within unharvested areas. Living trees  $\geq 4.0$  cm and dead trees  $\geq 17.5$  cm diameter-atbreast height (DBH), species, and height (m) were recorded in each plot. Twenty-four-metre CWD transects were re-established in the same locations as pre-harvest sampling, and each piece > 7.5 cm in diameter at the point of intersection of the sampling line was measured. Diameter, species, length, tilt angle, and decay class (0 to V) were recorded for each CWD intersection. The detailed descriptions of post-harvest sampling methods are provided by Jull et al. (2002).

In this study, the term live tree refers to all living tree components including stem wood, stem bark, branches, foliage, and fine and coarse roots. The term large tree refers to both living and dead trees. The term forest floor refers to all dead organic matter accumulated above the mineral soil, including litter, fine woody debris, humus, and decaying wood. Coarse woody debris refers to all downed logs, while snag refers to all standing dead trees but only includes stem wood. The term dead organic matter (DOM) refers to snags, CWD, and forest floor, and total forest C refers to live tree and dead organic matter (excluding mineral soil). In this study, we did not measure mineral soil C because of resource constraints, and forest-floor C is most likely to be responsive to management and climatic impacts as contrasted to mineral soil C (Fredeen et al. 2005; Nave et al. 2010).



Figure 2. Layout of experimental design in three study areas (Lunate Creek, East Twin Creek, and Minnow Creek) established in 2000 and 2001 from Jull et al. (2002).

# 2.2.3 Field methods and sampling

#### 2.2.3.1 Forest floor sampling

Existing large-tree permanent sample plots, regeneration plots, and vegetation plots were used as reference points for drawing forest-floor line transects (20 to 50-m long), such that 10 cores were successfully taken on a single transect. In total, 641 forest-floor samples (i.e., L, F, and H layers) were cored along line transects randomly chosen for each treatment unit using a fabricated coring bit (5 cm inside diameter) mounted on a battery-powered drill, according to the design and specifications of Nalder and Wein (1998). Each cored sample was separated based on color and texture: woody (prominence of thick, reddish woody materials) or non-woody (black humified materials) types. When cores could not be taken at the designated spot because of logs, large roots, stumps, or rocks, the information was recorded as such and cores were taken at the next closest point free of obstruction.

#### 2.2.3.2 Decayed-wood sampling

To correct for heart-rot and hollow commonly present in stem wood of western redcedar and western hemlock, we collected decayed wood from randomly-selected stumps and downed logs using an electric drill with a 5.0-cm diameter (D) cylindrical pipe and/or manually with a 9.5-cm D  $\times$  7.0-cm H (height) steel pipe. Due to time and resource constraints, we sampled western redcedar only, and also restricted sampling to the Lunate Creek study area. To collect decayed-wood samples from downed logs, we used a chainsaw to remove 7 to 10 cm thick 'cookies' from downed trees at lower, middle, and upper parts of the log for heartwood decay analysis. As needed, decayed-wood samples were cored at the cross section using an electric drill with a 5.0 cm diameter cylindrical pipe and/or manually with a 9.5-cm D  $\times$  7.0-cm H (height) steel pipe. Diameter was recorded at each cross section where the samples were collected. We attempted to collect multiple cores from a single cross-section of downed logs and stumps to increase precision of the measurement. In total, 25 samples (19 replicates) were collected from downed logs, with another 48 samples (33 replicates) cored from independent cedar stumps.

# 2.2.4 Laboratory analysis

Forest-floor samples (both woody and non-woody samples) were oven-dried (g) and weighed, and oven-dry mass per unit area (g cm<sup>-2</sup>) was calculated for each sample based on the dimensions of the fabricated coring bit. The samples were then ground through a 2-mm sieve using a coffee grinder for soil chemical analyses. Total C was measured using a Leco Truspec C/N analyzer (Analytical Chemistry Service, B.C. Ministry of Environment, Victoria). The C concentration was taken as equivalent to organic C.

All the cored samples collected from stumps and downed logs were first air-dried for two weeks prior to oven drying at 72°C for three days before measuring the weight of the oven-dried samples. The fresh-weight volumes (cm<sup>-3</sup>) of decayed wood were immediately measured by calculating the dimensions of corresponding steel pipes or sampled disks. Decayed-wood density (g cm<sup>-3</sup>) of each replicate of western redcedar was calculated by dividing the oven-dried mass by its fresh-weight volume.

# 2.2.5 Calculations

#### 2.2.5.1 Large tree C

Biomass and C estimates of large trees (living and dead trees) were calculated on a per-unit-area basis (Mg C ha<sup>-1</sup>), and determined by summing the biomass values of individual trees in permanent sample plots, estimating C stocks and scaling the estimate to the land area

of the plots. Biomass of large trees was estimated from existing large-tree permanentsample-plot data (i.e., diameter-at-breast height and height) from the Northern Wetbelt Silvicultural Systems Project (Jull et al. 2002) in combination with tree allometric equations (Jenkins et al. 2004; Ung et al. 2008). For partial-cut treatment units such as group retention and group selection, the calculated biomass was corrected by multiplying the percentage of tree removal (i.e., 0.30 and 0.70 for group retention and group selection, respectively). All the individual tree-biomass calculations were performed using STATA (StataCorp LP 2009).

Biomass estimates of tree components (stem wood, stem bark, branch, and foliage) were computed using Equation 1 (Ung et al. 2008):

$$[1] \quad \mathbf{B}_{\mathbf{U}} = \mathbf{a} \times \mathbf{D} \mathbf{B} \mathbf{H}^{\mathbf{b}} \times \mathbf{H}^{\mathbf{c}}$$

where  $\mathbf{B}_{U}$  is individual-tree component biomass (dry weight, kg), **DBH** is diameter-at-breast height (cm), **H** is height (m), and **a**, **b**, and **c** are the equation parameters. For trees exceeding the upper DBH limit for Equation 1, biomass estimates were calculated using Equation 2 (Jenkins et al. 2004):

[2] 
$$B_{I} = Exp(a + b \times ln(DBH)) + \varepsilon_{B_{I}}$$

where  $\mathbf{B}_{\mathbf{J}}$  is total aboveground living tree biomass (kg) including stem wood, stem bark, branch, and foliage (dry weight, kg), **DBH** (cm),  $\boldsymbol{\varepsilon}_{\mathbf{B}_{\mathbf{J}}}$  is error term, and **a** and **b** are the equation parameters. Each of the tree component biomass estimates using Equation 2 was calculated based on component ratios using Equation 3:

$$[3] \qquad \mathbf{R}_{\mathbf{J}} = \mathbf{Exp}\left(\mathbf{a} + \frac{\mathbf{b}}{\mathbf{DBH}}\right)$$

where  $\mathbf{R}_{\mathbf{J}}$  is the ratio of tree component to total aboveground living tree biomass, **DBH** (cm), and **a** and **b** are the biomass-ratio equation parameters. Biomass for each tree component was computed as the product of the ratio and the total aboveground living tree biomass (Equation 2).

Estimates of tree C were computed using the following equation:

$$[4] \qquad \mathbf{C}_{\mathrm{T}} = \sum (\mathbf{B}(\mathbf{c} + \mathbf{\varepsilon}_{\mathbf{c}}))$$

where **B** is the biomass of individual tree;  $C_T$  is the total C content in the tree; **c** is C concentration; and  $\varepsilon_c$  is error term. Stem-wood C percentage of all the species is derived from Lamlom and Savidge (2003), and stem-bark C concentrations of Douglas-fir (*Pseudotsuga menziesii* var. glauca), western hemlock, hybrid white spruce, Douglas maple (*Acer glabrum* Torr. var. *douglasii* (Hook.) Dippel., subalpine fir and western redcedar were 53.0, 51.2, 55.2, 50.4, 52.8, and 53.0%, respectively (Corder 1976). Carbon concentrations of all the small woody (branches) and non-woody components (foliage) were assumed to be 50% (Laiho and Prescott 1999).

# 2.2.5.2 Belowground (roots)

Live-root (fine and coarse roots) biomass was calculated as a function of total aboveground biomass using the stand-level equations (Li et al. 2003) for softwood (Equation 5) and hardwood (Equation 6) species of cold temperate and boreal forests.

$$[5] \qquad \mathbf{RB}_{s} = \mathbf{0.222AB}_{s} (0.222 \pm 0.004)$$

[6] 
$$\mathbf{RB_h} = 1.576 \mathbf{AB_h}^{0.615} (1.576 \pm 0.267, 0.615 \pm 0.035)$$

where  $\mathbf{RB}_{s}$  and  $\mathbf{RB}_{h}$  are total root biomass of softwood and hardwood,  $\mathbf{AB}_{s}$  and  $\mathbf{AB}_{h}$  are total aboveground living tree biomass of softwood and hardwood (Mg ha<sup>-1</sup>), and error terms were reported for each parameter in the parentheses. Total root C (Mg C ha<sup>-1</sup>) was calculated using the same C concentration as stem wood (Hoover 2008).

#### 2.2.5.3 Coarse woody debris

Coarse woody debris C stocks were calculated using established species-specific decay-classes (0 to V) densities (Table 2) and C concentrations. Existing records of transect length, log diameter, species, and tilt angle (Jull et al. 2002) were used to compute volume per-unit area for each treatment unit, according to Marshall et al. (2000). The C stock on a per-unit-area basis (Mg C ha<sup>-1</sup>) was then calculated as the product of the volume per-unit area (m<sup>3</sup> ha<sup>-1</sup>), species-specific decay-class densities (Mg m<sup>-3</sup>) and C concentrations. Given that stands are/were dominated by western redcedar and western hemlock, estimation of C stocks for unknown species was made by averaging decay-class densities of the two species due to the absence of well-documented densities in cedar-hemlock forests. Carbon concentrations of subalpine fir and hybrid white spruce were 0.4844 and 0.5084, respectively (Laiho and Prescott 1999) for all decay classes. Given that C concentrations of CWD in old-growth stands of Douglas-fir, coastal western redcedar and hemlock changed little with decay state (Sollins et al. 1987; Keenan et al. 1993), those of western redcedar and western hemlock in this study were assumed to be the same C concentration as the initial state (i.e., 0.515 for cedar and 0.506 for hemlock) for all decay classes.

Decay classes							
Species	0	I	II	III	IV	V	
Subalpine fir*		0.381	0.222	0.239	0.163	0.116	
Western redcedar <sup>†</sup>	0.333	0.318	0.259	0.248	0.154	0.143	
Western hemlock <sup>†</sup>	0.430	0.376	0.295	0.242	0.146	0.142	
Engelmann spruce <sup>‡</sup>		0.300	0.280	0.230	0.220	0.180	
Hybrid white spruce <sup>‡</sup>		0.300	0.280	0.230	0.220	0.180	
Unknown <sup>§</sup>	0.382	0.347	0.277	0.245	0.150	0.143	

Table 2. Species-specific decay-class densities (Mg m<sup>-3</sup>) used in this study.

D. Sachs, Forest Research Ecologist, Kamloops, British Columbia, unpublished data, 1997.

<sup>†</sup> Decay class 0 (Taylor 1997) and I to IV (Harmon and Sexton 1996).

<sup>‡</sup> (Sandstrom et al. 2007).

<sup>§</sup>Unknown species decay-class densities were averages of densities of western redcedar and western hemlock.

#### 2.2.5.4 Heart-rot and hollow

Stem-wood C of western redcedar and western hemlock was corrected for heart-rot using the measured or assumed decayed-wood density and existing volume-based decaycorrection factors derived from TREEVOL (BC Integrated Land and Management Bureau 2003). TREEVOL enables users to obtain the proportion of volume decayed in a given volume of an individual tree based on input variables such as biogeoclimatic zone and mensurational data. In this study, the correction factors were generated using information on DBH and biogeoclimatic zones (i.e., ICH). The trajectory curves of decay-loss factors for cedar and hemlock are illustrated as a function of DBH for large tree (Figure 3). Decayedwood density of western hemlock (0.22 g cm<sup>-3</sup>) was assumed to be half the sound-wood density (0.43 g cm<sup>-3</sup>) (Harmon et al. 2004). Observations of decayed heartwood of western redcedar at Lunate Creek in this study indicate that the vertical and horizontal distribution, abundance, and size of hollow and heart-rot were found to be extremely variable, dynamic, and inseparable (e.g., the presence of partially decayed wood combined with hollow). As a result, due to the technical difficulties in estimating the proportion of volume loss to hollow, we performed a sensitivity analysis by multiplying the estimated volume of decayed-wood by four proportions of hollow: 100 (baseline), 50, 20 and 0%. The sensitivity of live-tree and total forest C to the hollow proportions was determined for each treatment unit.



Figure 3. Volume-based decay-loss factor (%) as a function of DBH (cm) for large tree (live and dead) derived from TREEVOL. The end of each curve represents the maximum DBH observed in this study.

#### 2.2.6 Uncertainty calculation

In this study, a Monte Carlo approach (Yanai et al. 2010) was used to obtain probability distributions of live-tree, dead organic matter (DOM), and total forest C stocks through multiple iterations using either reported or estimated errors that were randomly selected. The probability distributions were then used to compute means and uncertainties for each treatment unit (see Appendix 1). Uncertainty is expressed as the 95% confidence interval (CI) calculated as the mean  $\pm 1.96 \times$  standard error.

# 2.2.6.1 Live tree C

We estimated mean and uncertainty of live-tree C by propagating reported regression (i.e., root mean square) and parameter error, C concentration error, and sampling error within calculations. The tree and root allometric equations ([1], [2], [5], and [6]) were repeatedly (up to 10,000 times) perturbed by randomly sampling the residuals or parameter errors at each iteration, assuming that all the error terms were independent of each other and normally distributed with a zero mean and  $\pm$  1 standard deviation. Therefore, we used the same equation parameters at each iteration and based the Monte Carlo analysis on the residual error terms for the equation [2] and parameter errors for the equations [1], [5], and [6]. Similarly, error term for the equation [4] was randomly sampled using Monte Carlo simulation with a zero mean and  $\pm$  1 standard deviation. For each treatment unit consisting of 8 permanent sample plots, mean live-tree C and its standard error were computed for Monte Carlo simulation. These numbers were then used to randomly generate a single estimate of a means and standard error for an ecosystem component for which biomass or C stocks were estimated (this calculation gives sampling error). Multiple estimates of the ecosystem values were copied and accumulated in a separate datasheet. Error statistics were

then computed by taking a grand average and standard deviation of the accumulated means for each treatment unit after the calculation was repeated up to 10,000 times, which was sufficient to estimate the mean C stock of live tree. The number of iterations was determined by looking at the distributions of the error statistics as a function of the number of Monte Carlo iterations, depending on the number required for the mean and the standard deviation to stabilize. The standard deviation from the Monte Carlo runs provides an estimate of the variance of the sample means and therefore is equivalent to the standard error of the mean (Altman and Bland 2005).

In addition to representing all sources of error combined for each treatment unit, we partitioned sources of errors from tree and root allometric equations, C concentration, and sampling error to identify which error components contributed most profoundly to total uncertainty in live-tree C. Because C contents of live trees were estimated in response to four proportions of hollow, total uncertainty and individual error components of live-tree C stocks were computed for each proportion of the hollow. The estimated uncertainties were reported and compared among study areas and treatment units to examine the sensitivity of live-tree C to hollow proportions. A detailed description of uncertainty calculation associated with regressions is provided by Yanai et al. (2010).

# 2.2.6.2 Heart-rot

Uncertainty associated with estimated decayed-wood density of western redcedar for heart-rot was calculated using a Monte Carlo approach. To account for uncertainty in the estimated decayed-wood density, we used estimated mean density and its standard deviation. A probability distribution of the mean density was obtained by randomly selecting the standard deviation assuming it was normally distributed. At each iteration, the randomly

chosen standard deviation was added to the mean decayed-wood density which was then used for the computation of decayed-wood mass and C of western redcedar.

# 2.2.6.3 Dead organic matter (DOM) and total forest C

A Monte Carlo approach was also used to estimate combined probability distributions of DOM (snag, CWD, and forest floor) and total forest C stocks (i.e., live tree, snag, CWD, and forest floor) for each treatment unit. We employed a normally distributed randomnumber-generator in STATA defined by mean and standard error of each of the forest components. We assumed that error terms of live tree and DOM were independent of each other. The simulation was repeated 10,000 times, and DOM and total forest C stocks were computed by summing the randomly generated means of forest components at each iteration. We reported 95% CI as a final representation of uncertainty by computing the grand average and standard deviation of a set of accumulated means of DOM and total forest C stocks.

## 2.2.7 Statistical analysis

Linear regression analyses were performed to test effects of diameters of stumps and logs on estimated decayed-wood densities of western redcedar. A *t*-test was also run to test significant difference in the densities between stumps and logs. The results of both analyses were used to estimate a single mean and its standard deviation of the decayed-wood density which were then incorporated into the calculation of stem-wood biomass of cedar.

Across all the study areas, effects of the four treatments were examined for their effects on live-tree, snag, DOM, and total forest C stocks based on estimated 95% confidence intervals after Monte Carlo simulation. A Monte Carlo permutation test (Gotelli and Ellison 2004) was used to detect significant differences in mean forest-floor mass and C stocks. The permutation refers to reordering of the observations of forest-floor mass and C (Baum 2009) where linear regression was executed at each reordering among treatment units. A linear mixed-effect model was used to test significant differences in mean C stocks of CWD where we treated line transect as a random effect and treatment units as a fixed effect. All statistical analyses were conducted using STATA and considered significant when  $\alpha < 0.05$ .

### 2.3 Results

## 2.3.1 Decayed-wood density of western redcedar

There were no significant effects of diameters on decayed-wood density of western redcedar for both stumps ( $F_{1,31} = 0.0033$ , P = 0.97, df = 32; Figure 4a) and logs ( $F_{1,15} = 2.98$ , P = 0.105, df = 16; Figure 4b) and no significant differences in mean densities were found between logs and stumps (z = 1.01, P = 0.27, n = 50). Therefore, the mean and s.d. representing decayed-wood density of western redcedar were calculated using all the density observations (mean = 0.110 g cm<sup>-3</sup>, s.d. = 0.026, n = 50). The uncertainty was propagated using the standard deviation and both of these values were used to correct for heart-rot in stem wood of western redcedar.



Figure 4. Scatter plot of decayed-wood densities of western redcedar estimated from (a) stump (n = 33, mean = 0.105, s.d. = 0.004, p = 0.97) and (b) downed logs (n = 17, mean = 0.120, s.d. = 0.011, p = 0.105).
#### 2.3.2 Importance of components to the uncertainty of live-tree carbon estimates

Coefficients of variation (CVs) were used to discern the relative contributions of the error components examined in this study to the uncertainty of live-tree C stock estimates. Results suggest that there are important ecological differences among stands for relative importance of the components, with implications for assessing the C stocks (Table 3). CVs for the tree allometric equations were high across all the study areas; 20% for group-retention plots to 27% for group-selection plots, indicating the equations are the most important contributor to uncertainty for live-tree C stock estimates. This was especially true for the Lunate Creek study area with the highest CVs (24%, group retention; 29%, group selection; 28%, uncut) among the three study areas. Among the three study areas, CVs for the allometric equations were smaller at Minnow Creek (the range of 15 to 20%) than at the other two study sites (the range of 19 to 29%). The next most important component was sampling errors where CVs were 9 (group retention), 14 (group selection) and 6% (uncut). Low CVs were associated with decayed-wood density of western redcedar (0.2 to 2%) and the CVs declined gradually as the proportion of hollow increased. CVs associated with the root model and C% were even smaller, ranging from 0.2 to 0.4%. CVs for total biomass C stocks varied among treatment units; group-selection plots had the largest CVs (30 to 37%) across and within the study areas.

Study area	Treatment	Error components*:					
	units	Model	Carbon	Sampling	Density	Root	Total
East Twin Creek	GS	28,27,27,27	0.4,0.4,0.4,0.4	7,7,7,7	0,1,2,2.2	0.3,0.3,0.3,0.3	28,28,29,28
	UN	19,19,19,19	0.3,0.3,0.4,0.4	5,5,5,5	0,1,1,1.5	0.2,0.2,0.2,0.2	20,20,20,20
Lunate Creek	GR	23,23,23,24	0.3,0.3,0.3,0.3	10,10,10,10	0,1,2,1.9	0.3,0.3,0.2,0.3	25,25,25,26
	GS	29,29,29,29	0.4,0.4,0.4,0.4	22,22,22,21	0,1,2,2.5	0.3,0.3,0.3,0.3	37,37,37,37
	UN	28,27,28,28	0.4,0.4,0.4,0.4	10,10,10,10	0,1,2,2.3	0.3,0.3,0.3,0.3	30,30,30,30
Minnow Creek	GR	15,15,15,15	0.3,0.3,0.3,0.3	13,13,13,14	0,1,1,1.2	0.2,0.2,0.2,0.2	21,21,21,21
	GS	19,20,19,19	0.3,0.3,0.3,0.3	8,9,8,8	0,1,1,1.6	0.2,0.2,0.2,0.2	22,21,21,21
	UN	18,18,18,18	0.3,0.3,0.3,0.3	9,9,9,9	0,1,1,1.4	0.2,0.2,0.2,0.2	21,20,21,20
All study areas	GR	20,20,20,20	0.3,0.3,0.3,0.3	9,9,9,9	0,1,1,1.7	0.2,0.2,0.2,0.2	22,23,23,23
	GS	27,27,26,26	0.4,0.4,0.4,0.4	14,14,14,14	0,1,2,2.2	0.3,0.3,0.3,0.3	30,30,31,30
	UN	22,22,22,22	0.3,0.3,0.3,0.3	6,6,6,6	0,1,1,1.8	0.2,0.2,0.2,0.2	23,23,23,23

Table 3. Uncertainty (coefficient of variation) of individual error components in live-tree C stocks after Monte Carlo simulation for four proportions of hollow: 100 (baseline), 50, 20, and 0%.

Note: Coefficient of variation (CV) represents percentage (%) calculated as standard deviation divided by mean. CVs are listed in the order of 100 (baseline), 50, 20, and 0% of hollow. Each CV represents an estimate after 10,000 iterations of Monte Carlo simulation. Hollow was corrected for western redcedar and western hemlock only.

\* Model, carbon, sampling, density, and root in the table represent tree allometric equations, carbon concentrations, permanent sample plots, estimated decayed-wood density of western redcedar, and root allometric equations, respectively.

# 2.3.3 Total forest and live-tree carbon

In this study, distributions of total forest and live-tree C stocks displayed large differences among old-growth (uncut), partial-cut, and clear-cut stands. Mean total forest C stocks in group retention, group selection, and old-growth stands respectively were 1.6-, 4.0-, and 4.5-fold greater relative to clear-cut stands (Table 4, Figure 5). The total forest C stocks for old-growth stands ( $455 \pm 156 \text{ Mg C ha}^{-1}$ ; mean  $\pm 95\%$  confidence interval) were not significantly different from those of group selection  $(395 \pm 179 \text{ Mg C ha}^{-1})$  across the three study areas <sup>1</sup>. In contrast, total forest C stocks were significantly lower in clear-cut (99  $\pm$  14 Mg C ha<sup>-1</sup>) and group-retention cut (160  $\pm$  40 Mg C ha<sup>-1</sup>) treatments relative to those of oldgrowth ( $455 \pm 156 \text{ Mg C ha}^{-1}$ ) and group-selection cut ( $395 \pm 179 \text{ Mg C ha}^{-1}$ ) stands. Mean total forest C stocks in group-selection cut stands were significantly greater (2.5-fold) than those in group-retention cut stands across the three study areas. These patterns of differences in total forest C followed those of live-tree C that accounted for 76% of total forest carbon (63% in aboveground living tree and 14% in roots: Table 4, Figures 5 and 6). The hollow proportions did not have significant effects on the total forest and live-tree C stocks within each treatment unit (Figure 5 and 6). Roots accounted for 14% of total forest C in oldgrowth stands across the study areas, thus root C stocks were significantly greater in groupselection cut (56 Mg C ha<sup>-1</sup>) and old-growth (63 Mg C ha<sup>-1</sup>) compared to group-retention cut (16 Mg C ha<sup>-1</sup>) stands. Root C, however, was calculated using allometric equations and thus was simply proportional to the aboveground living tree biomass estimations. The Lunate Creek study area displayed considerably higher mean total forest and live-tree C stocks in

<sup>&</sup>lt;sup>1</sup> Detailed estimates of mean live-tree and total forest C stocks for 50, 20, and 0% hollow proportions are provided in Tables A2-A4 of Appendix 2.

group-selection cut and old-growth stands as compared with those of the East Twin and Minnow Creek study areas.



Figure 5. Mean total forest C stocks in four treatment units across three ICH study areas with four proportions of hollow: 100 (baseline), 50, 20, and 0%.

Hollow proportions indicate the volume percentage of the internal stem decay which is actually hollow<sup>2</sup>. CC, GR, GS, and UN represent clear-cut, group retention, group selection, and uncut forest harvesting treatments, respectively. Error bars indicate 95% confidence interval. Means sharing the same letter are not significantly different based on 95% confidence intervals derived from Monte Carlo simulation analysis.

<sup>&</sup>lt;sup>2</sup> Detailed calculation of tree biomass estimates for hollow proportions are provided in Figure A2 of Appendix 1.



Figure 6. Mean live-tree C stocks in four treatment units across three ICH study areas with four proportions of hollow: 100 (baseline), 50, 20, and 0%.

Hollow proportions indicate the volume percentage of the internal stem decay which is actually hollow. CC, GR, GS, and UN represent clear-cut, group retention, group selection, and uncut forest harvesting treatments, respectively. Error bars indicate 95% confidence interval. Means sharing the same letter are not significantly different based on 95% confidence intervals derived from Monte Carlo simulation analysis.

Study area	Treatment unit		Live tree			DOM§	Total forest
		$\mathbf{n}^{\dagger}$	Aboveground <sup>‡</sup>	Root	Total		
				Mg C ha <sup>-1</sup>			
East Twin Creek	CC	0	0±0	0±0	0±0	77±19	77±19
	GS	8	201±111	45±25	246±136	65±18	311±137
	UN	8	259±102	58±23	317±125	71±14	388±126
Lunate Creek	CC	0	0±0	0±0	0±0	135±42	135±42
	GR	8	84±42	1 <del>9±9</del>	103±51	67±12	170±53
	GS	8	396±285	88±63	485±346	10 <del>6±</del> 27	592±347
	UN	8	350±202	78±45	427±245	147±46	574±250
Minnow Creek	CC	0	0±0	0±0	0±0	78±15	78±15
	GR	8	56±23	13±5	69±28	83±13	152±30
	GS	8	158±64	35±14	193±79	89±14	282±80
	UN	8	246±96	55±22	301±118	101±18	402±119
All study areas	CC	0	0±0	0±0	0±0	99±14 a	99±14 a
	GR	16	70±30 a	16±7 a	86±37 a	74±15 a	160±40 b
	GS	24	251±148 b	56±33 b	307±178 b	88±15 a	395±179 с
	UN	24	284±127 b	63±28 b	348±155 b	107±19 a	455±156 c

Table 4. Mean live-tree, dead organic matter (DOM), and total forest C stocks ( $\pm$  95% confidence interval) in four forest harvesting treatment units at three ICH study areas. For this baseline case, 100% of the internal stem decay volume was assumed to be hollow.

Note: Means across all study areas within a column sharing the same letter are not statistically significant using 95% confidence intervals generated from Monte Carlo simulation.

CC, GR, GS, and UN represent clear-cut, group-retention, group-selection, and uncut forest harvesting treatments, respectively.

<sup>†</sup>n represents the number of large-tree permanent sample plot of live-tree C stocks.

<sup>‡</sup>Aboveground includes stem wood, bark, branch, and foliage.

<sup>§</sup>DOM (dead organic matter) includes snag, CWD, and forest floor.

<sup>th</sup>Total forest C includes live tree and DOM but excludes mineral soil C which was not measured in this study.

#### 2.3.4 Dead-organic-matter (DOM) carbon

Total DOM C stocks  $(107 \pm 19 \text{ Mg C ha}^{-1})$  in old-growth stands were not significantly different from those of partial-cut and clear-cut stands across all the study areas but were highly variable within each study area (Table 5). Average DOM C stock for uncut old-growth stands accounted for 24% of total forest C stocks (5% in snag, 9% in CWD and 10% in forest floor). Mean snag C stock in old-growth stands (22 Mg C ha<sup>-1</sup>) was significantly greater than that of group-retention cut stands (4 Mg C ha<sup>-1</sup>) but not significantly different from that of group-selection cut stands (10 Mg C ha<sup>-1</sup>) across the three study areas. CWD C stocks were significantly greater in clear-cut stands (62 Mg C ha<sup>-1</sup>) than in group retention (40 Mg C ha<sup>-1</sup>; P = 0.026), group selection (43 Mg C ha<sup>-1</sup>; P = 0.034), or old-growth stands (39 Mg C ha<sup>-1</sup>; P = 0.015) across all the study areas. The Lunate Creek study area showed consistently greater CWD C stocks than those of the other two study areas across the four treatment units. In particular, CWD C stocks in clear-cuts at Lunate Creek (98 Mg C ha<sup>-1</sup>) were 2.5-fold greater than those of the East Twin Creek (41 Mg C ha<sup>-1</sup>) and Minnow Creek (38 Mg C ha<sup>-1</sup>) study areas. Forest-floor C in old-growth stands (46 Mg C ha<sup>-1</sup>) <sup>1</sup>) was significantly greater than in group-retention (31 Mg C ha<sup>-1</sup>; P = 0.026) but not in group-selection (35 Mg C ha<sup>-1</sup>; P = 0.092) and clear-cut (38 Mg C ha<sup>-1</sup>; P = 0.210) stands across all the study areas (Table 5). The difference in forest-floor mass between group retention (71 Mg ha<sup>-1</sup>) and old-growth stands (99 Mg ha<sup>-1</sup>) was nearly significant (P =0.0515), but no differences were statistically detected between any treatment units across all the study areas (all P > 0.094; Table 6). Decaying wood (i.e., woody forest-floor C) accounted for 31% (clear-cut), 30% (group retention), 25% (group selection), and 44% (uncut) of forest-floor C stocks across the study areas (Table 6). The decaying-wood forestfloor C stocks were significantly greater in old-growth stands (21 Mg C ha<sup>-1</sup>) compared to

those in group retention (9 Mg C ha<sup>-1</sup>; P = 0.0435) and group selection (9 Mg C ha<sup>-1</sup>; P = 0.0245) across the three study areas, while no differences in non-woody forest-floor C stocks were statistically detected between any treatment units (all P > 0.292). Although statistically not significant, the reduction in forest-floor C for group-retention cuts relative to uncut old-growth stands was consistently observed at Lunate Creek and Minnow Creek. A similar response was not observed between group-selection and uncut old-growth stands where forest-floor C stocks were higher in the group-selection treatment at Minnow Creek (46 Mg C ha<sup>-1</sup>) than in uncut old-growth stands (44 Mg C ha<sup>-1</sup>).

Study area	Treatment	De			
		Snag <sup>†</sup>	CWD <sup>‡</sup>	Forest floor <sup>§</sup>	Total
			Mg C ha <sup>-1</sup>		
East Twin Creek	CC	0±0 (0)	41±17 (21)	36±8 (4)	77±19
	GS	4±4 (8)	35±16 (18)	26±8 (4)	65±18
	UN	7±7 (8)	23±10 (18)	41±6 (8)	71±14
Lunate	CC	0±0 (0)	98±40 (22)	36±13 (4)	135±42
	GR	3±2 (8)	32±11 (24)	32±5 (4)	67±12
	GS	17±15 (8)	54±22 (24)	34±4 (4)	106±27
	UN	37±30 (8)	57±32 (21)	54±14 (8)	147±46
Minnow	CC	0±0 (0)	38±12 (15)	40±9 (4)	78±15
	GR	5±2 (8)	49±12 (20)	29±3 (4)	83±13
	GS	7±4 (8)	37±11 (22)	46±7 (4)	89±14
	UN	22±11 (8)	35±10 (18)	44±11 (8)	101±18
All study areas	CC	0±0 (0)	62±13 (58) a	38±6 (12) ab	99±14 a
	GR	4±2 (16) a	40±15 (44) b	31±3 (8) a	74±15 a
	GS	10±6 (24) ab	43±12 (64) b	35±5 (12) ab	88±15 a
	UN	22±13 (24) b	39±13 (57) b	46±6 (24) b	107±19 a

Table 5. Estimates of mean dead organic matter ( $\pm$  95% confidence interval) C stocks (sample size in the parentheses) in old-growth stands at three ICH study areas.

Note: Means within a column sharing the same letter are not significantly different. The sample size in the parentheses for snag and CWD and forest floor indicates the number of permanent sample plots and line transects, respectively.

\*CC, GR, GS, and UN represent clear-cut, group-retention, group-selection, and uncut, respectively. \*Snag includes stem wood only. Significant effects were tested based on 95% confidence interval after Monte Carlo simulation.

<sup>‡</sup>A linear mixed-effect model was used to detect significant differences (P < 0.05).

<sup>§</sup>A permutation test was used to detect significant differences (P < 0.05).

Study area	Treatment	n†	mass (Mg ha <sup>-1</sup> )	Type (Mg C ha <sup>-1</sup> )		Carbon (Mg C ha <sup>-1</sup> )	C concentration (% of dry mass)
			× • • •	Woody	non-woody		· · · ·
East Twin Creek	CC	41	<b>94</b> ±11	16±7	22±4	36±4	41±2
	GS	80	76±10	2±2	27±10	26±4	38±2
	UN	47	86±6	23±5	22±3	41±3	48±2
Lunate Creek	CC	41	81±17	9±1	21±4	36±7	46±1
	GR	82	74±7	5±3	26±6	32±2	<b>46</b> ±1
	GS	82	74±5	13±10	22±3	34±2	4 <del>9±</del> 1
	UN	49	106±16	1 <del>9±</del> 8	38±8	54±7	52±1
Minnow Creek	CC	42	112±4	6±4	29±9	40±5	40±4
	GR	88	67±3	12±5	17±3	2 <del>9±</del> 2	44±2
	GS	87	109±10	12±8	31±7	46±4	44±1
	UN	45	106±10	21±7	22±3	44±6	43±3
All study areas	CC	124	96±7 a	11±3 ab	24±3 a	38±3 ab	42±2
	GR	170	71±4 a	9±2 a	21±3 a	31±1 a	45±1
	GS	249	87±7 a	9±3 a	27±4 a	35±3 ab	44±1
	UN	141	99±7 a	21±4 b	27±3 a	46±3 b	48±1

Table 6. Estimates of mean forest-floor mass and measured C content (± standard error) for three cedar-hemlock forest study areas.

**Note:** Means within a column sharing the same letter are not significantly different (permutation test, P < 0.05). \*CC, GR, GS, and UN represent clear-cut, group retention, group selection, and uncut treatment units, respectively. \*n indicates the number of forest-floor samples used for biomass and C stocks estimations.

### 2.4 Discussion

## 2.4.1 Total forest C

Total forest C stocks in forest ecosystems can vary as a result of a number of factors, including stand age, climate, natural disturbance, and management type and intensity. This study examined the impact of harvesting system and intensity on total forest carbon (excluding mineral soil) by measuring or estimating forest C stocks in clear-cut, partial-cut (30% retention and 70% retention) and in uncut old-growth stands in cedar-hemlock or inland temperate rainforests (ITRs) of east-central British Columbia. The gradient of the decline in total forest C stocks was proportional to the percent removal of live-tree C, as it accounts for a large proportion (e.g., 76% in uncut old-growth stands) of total forest C. This proportional contribution of live-tree to total forest C is similar to previous estimates for old western redcedar-hemlock stands of Vancouver Island, BC which ranged from 69 to 89% (Trofymow and Blackwell 1998; Trofymow et al. 2008), but lower than the 82 to 92% in old boreal forests of hybrid white spruce and subalpine fir (Kranabetter 2009) and the 78 to 82% contribution observed for temperate old-growth Douglas-fir and western hemlock forests (Hoadley 1990; Smithwick et al. 2002; Harmon et al. 2004) and 81% contribution observed for inland old larch (Larix occidentalis Nutt.) - Douglas-fir forests of the Pacific Northwestern USA (Bisbing et al. 2010). Note that the comparisons of these proportional contributions between this study and other studies here and hereafter do not account for mineral soil unless otherwise stated.

Mean live-tree and total forest C stocks in this study were consistent with preliminary estimates of C stocks (including mineral soil) for cedar-hemlock stands containing the study areas (478 Mg C ha<sup>-1</sup>: Stevenson et al. 2011) and also with the range of mean live-tree and

total forest C stocks (excluding mineral soil) of old-growth western redcedar and western hemlock on Vancouver Island which ranged from 258 to 532 Mg C ha<sup>-1</sup> and from 357 to 710 Mg C ha<sup>-1</sup>, respectively (Keenan et al. 1993; Trofymow and Blackwell 1998; Trofymow et al. 2008). Our cedar-hemlock forests carbon stocks are 22 to 29% higher than the maximum living biomass C of 138 Mg C ha<sup>-1</sup> (270 Mg ha<sup>-1</sup> biomass, assuming 51% C content) reported for all montane cordilleran hemlock forests in BC (Penner et al. 1997) and higher than mean total forest C in immediately adjacent sub-boreal spruce and true fir forests in central BC (268 Mg C ha<sup>-1</sup>: Fredeen et al. 2005; Bois et al. 2009). Our estimates of live tree ( $348 \pm 155$ Mg C ha<sup>-1</sup>) and total forest (455  $\pm$  156 Mg C ha<sup>-1</sup>) C stocks fall in the middle of the regional averages for total forest C stocks (excluding mineral soil) in old temperate forests of the Pacific Northwestern USA dominated by Douglas-fir, western hemlock, sitka spruce (Picea sitchensis (Bong.) Carr.), and amabilis fir (Abies amabilis Dougl. ex J. Forbes), i.e., 444 Mg C ha<sup>-1</sup> (range of 112 - 596 Mg C ha<sup>-1</sup>) and 540 Mg C ha<sup>-1</sup> (range of 142 - 713 Mg C ha<sup>-1</sup>), respectively (Smithwick et al. 2002). Overall, our estimates of mean total forest C (excluding mineral soil) fall within those of the Pacific Northwestern USA and coastal BC and above those of the adjacent sub-boreal forests.

Analysis of the contribution of different components to uncertainty identified areas of future improvements for more accurate assessment of live-tree C in old ITRs. High coefficients of variation (CVs) for the biomass model equations (range of 20 to 27%) and sampling error (6 to 14%) across and within the study areas reflect the need for better ITR-specific allometric equations to capture the large variation in live-tree C pools in stands dominated by large old-growth western redcedars and hemlocks (Table 3). While allometric equations represent the best fit through the data points, the error tends to be highest in stands with large trees (Harmon et al. 2007). Trees to which the allometric equations are applied are

often not representative of the population of trees from which the equations were developed, thereby a large systematic error (i.e., bias) may be introduced but is not usually described in the regression model (Harmon et al. 2007; Kloeppel et al. 2007). Regional specific growth variation of trees due to genetic, competitive, climatic, and edaphic differences reflected in allometric equations might either underestimate or overestimate tree biomass in regions where the biomass equations were not originally developed (Harmon et al. 2007). Unfortunately, the absence of available equations that covered large-DBH western redcedars in this study did not permit a more suitable model selection.

There are other sources of uncertainty not included in this study such as measurement uncertainty associated with DBH or height. For example, uncertainty associated with DBH measurement accounted for only 0.02 to 0.03% for estimating nitrogen content of northern hardwood trees at the Hubbard brook Experimental Forest in New Hampshire (Yanai et al. 2010). While the measurement uncertainty may be small for stands occupied by small diameter trees, it may not be trivial for stands dominated by large trees such as old ITRs. Remeasurement of the same trees will potentially provide reasonable bounds on the possible errors associated with DBH and height measurements, and Monte Carlo simulation is able to estimate the relative uncertainty associated with the DBH and height measurements to total uncertainty for live-tree C estimates.

The high CVs in the sampling errors discovered in Lunate Creek could mean that a larger sample size is necessary to capture the spatial variation in forest C stocks of oldgrowth stands occupied by large cedars. It is intriguing, however, that group-selection cut stands of Lunate Creek have the highest CVs in the sampling errors and also the highest livetree C stocks. The association between large sampling error (i.e., high spatial variation) and high biomass C stocks observed in this study has also been identified in other old temperate

forests. For example, there was substantial spatial variability in total forest C across sites in old temperate rainforests of Mountain ash (*Eucalyptus regnans*) in southeastern Australia, ranging from 262 to 2844 Mg C ha<sup>-1</sup> (Keith et al. 2009). It was observed that the wide spatial variation came from stands that experienced past partial stand-replacing natural disturbances. The highly complex structure of old-growth stands without intensive land-use change sometimes led to large among-site variation in C pools but higher ecosystem C accompanied by establishment of new cohorts in the gaps (Mackey et al. 2008; Keith et al. 2009). A similar observation in this study suggests that group-selection cut has the potential of mimicking these natural processes at the stand level and thus maintaining the C stocks observed for uncut old-growth stands if growth of planted and naturally-regenerated trees in the gaps is satisfactory.

The mean decayed-wood density of western redcedar (0.11 g cm<sup>-3</sup>) was lower than decayed log densities measured for western redcedar (0.15 g cm<sup>-3</sup>) in old-growth stands of the Cascade Range of Oregon and Washington at the stand age of 80 to 200 years (Sollins et al. 1987) as well as for decay-class V CWD (0.23 g cm<sup>-3</sup>) in old-growth western redcedarwestern hemlock forests (Keenan et al. 1993). Our slightly lower value of decayed-wood density of western redcedar is partly related to our exclusion of sound wood (i.e., sapwood) in our sampling, as we focused on decayed wood in the heartwood only. We speculate that internal heartwood decay of western redcedar is more extensive in the interior than in coastal BC (Buckland 1946) which may further contribute to the low decayed-wood density observed in this study. Evidence for this is that concentrations of decay resistant compounds (e.g., β-thujaplicin) are lower in interior cedar (Daniels and Russell 2007). Within interior cedar, however, the magnitude of internal decay has been reported to be uniform irrespective of dry, moist, and wet subzones in the ICH (Stevenson et al. 2010), suggesting that our

decayed-wood density for interior western redcedar might be within a range of decayedwood densities in other ICH subzones, if such measurements were conducted. Nonetheless, sampling over the wide range of DBHs and sites would be necessary in the future to capture regional spatial variability in the decayed-wood density of interior western redcedar.

Presence of hollow (due to advanced heart-rot) in redcedar in this study had no statistically significant effect on live-tree or total forest C stocks. The abundance and size of redcedar and hemlock had a large influence on the relative contribution of decay-loss factors to stand-level biomass estimates for these species, as the loss factors were determined solely as a function of DBH (Figure 3). Wood loss factors reached an asymptote beyond a certain DBH, depending on the status (i.e., live or dead) and species of tree (Figure 3). Because the estimated decayed-wood volume is a direct reflection of the decay-loss factors and hollow proportions (100, 50, 20, and 0%), the relative effect of hollow on tree C became constant beyond a certain DBH (e.g., 29% beyond 100 cm DBH for cedar). Thus, at the stand level, absolute biomass C stocks of trees declined more in stands with a higher proportion of large cedar (i.e., Lunate Creek) but the relative effect was small. The combined uncertainty in live-tree C was nearly uniform with hollow, indicating that hollow did not contribute to the uncertainty in live-tree and total forest C stock estimates.

Live-root C stocks accounted for a large proportion (14%) of total forest C and the very small uncertainty in the root allometric equations (0.2 to 0.3%) contributed little to the total uncertainty in live-tree C stock estimation in this study. However, the root C stocks were directly dependent on total aboveground living tree C, which was based on allometric equations which are the largest contributor to uncertainty in this analysis. In general, inventory data do not include data for roots because they are difficult and expensive to measure in part due to their high spatial and temporal heterogeneity (Robinson 2007) and this

has been major data gap for ecosystem models (Cairns et al. 1997; Geider et al. 2001). The high uncertainty often leads to poor estimates for the root C pool and requires development for regional allometric equations (Laclau 2003). Thus, further research is needed in testing the reliability of the shoot: root allometry in old-ITRs.

Harvest intensity and rotation lengths are important factors in developing forest management strategies for sequestering forest ecosystem C (Thornley and Cannell 2000; Jandl et al. 2007a; Taylor et al. 2008). Past simulation-modelling studies have suggested that partial harvesting is better able to maintain higher ecosystem C in the long-term compared to clear-cutting (Thornley and Cannell 2000; Harmon and Marks 2002; Harmon et al. 2009; Swanson 2009) while sustaining timber yield (Thornley and Cannell 2000). The temporal variability in total forest C stock (including mineral soil) between different partial harvests (20 to 80% tree removal) was found to be small when long rotation lengths (up to 250 years) were used for typical old-growth forests of Douglas-fir and western hemlock in the Pacific Northwestern USA (Harmon and Marks 2002; Harmon 2009). It is clear, however, that the variability between harvest patterns was most pronounced immediately after harvesting with 80% tree removal resulting in lower total forest C than 20% removal due primarily to the large contribution of trees to total forest C. A similar response was observed in this study with total forest C being proportionally lower in order of decreasing percent tree retention. While frequent partial harvesting of stands has been found to store as much C as long rotations with complete removal, long-rotation intervals store more C than short rotations in Douglas-fir and western hemlock stands (Liski et al. 2001; Harmon et al. 2009), Scots pine (Pinus sylvestris L.) and Norway spruce (Picea abies (L.) Karst.) stands in Finland (Liski et al. 2001), and western Canadian boreal forests (Seely et al. 2002). A similar response was also observed in temperate southern beech (Nothofagus) forests of Tierra del Fuego, Chile,

where partial harvesting with long rotations led to higher biomass C at a landscape level (Swanson 2009). Higher levels of ecosystem C after partial harvesting with longer-rotation lengths relate to maintaining higher net primary productivity (NPP) through a continuous level of growing biomass in remnant standing trees combined with fast growing young cohorts in the harvested area, lowering the effective proportion of C removed per harvest (Taylor et al. 2008; Harmon et al. 2009), and raising litter inputs to the soil (Thornley and Cannell 2000). Thus, the effect of rotation length on total forest C seems particularly great in old temperate forests where aboveground living biomass C accounts for 50 to 70% of total ecosystem C (Lichstein et al. 2009). These modelling studies suggest that the higher retention level of ecosystem C after partial harvesting can be equally important in old ITRs, and the large proportion of tree C indicates that these forests are able to maintain C through careful management. Unfortunately, current studies on the long-term effect of partial harvesting and rotation lengths on total forest C stocks mostly rely on process-based modeling, not field-based studies (Jandl et al. 2007b; Harmon et al. 2009). Thus, the three ICH study sites are able to provide important empirical data for identifying long-term spatial and temporal dynamics of total forest C after partial harvesting.

There is substantive evidence that second-growth forests of varying stand ages store lower C than old-growth forests; for example, in old temperate forests of western redcedar and western hemlock in coastal BC (Trofymow et al. 2008), and Douglas-fir and western hemlock in the Pacific Northwestern USA (Harmon et al. 1990; Janisch and Harmon 2002; Bisbing et al. 2010). Total forest C in southern beech forests of Tierra del Fuego, Chile was significantly decreased after short-rotation (100 years) clear-cutting, on average 21% below that of old-growth forests (Swanson 2009). Sub-boreal stands of old hybrid white spruce and subalpine fir lost total forest C (including mineral soil) about 54 to 41% when converted to

the second-growth (Fredeen et al. 2005). Rotation length of second-growth forests has important implications for the extent of C stocks recovery after clear-cutting of old-growth forests. Based on data collected along a 500-year chronosequence in Washington, regenerating stands stored only half the live-tree and CWD C stocks compared to nearby oldgrowth forests after rotation length of 80 years (Janisch and Harmon 2002). Similarly, model simulations projected that total forest C of old western redcedar-hemlock stands in BC would never return to the pre-logging level after clear-cutting (Trofymow et al. 2008). Modelling studies have demonstrated that the conversion of old-growth forests of Douglas-fir and western hemlock to second growth with a 50- to 100-year rotation length was projected to release up to 250 to 350 Mg C ha<sup>-1</sup> into the atmosphere, even after accounting for C stored in wood products (Harmon et al. 1990; Harmon and Marks 2002). Carbon stored in plantations through short-rotation clear-cutting may not be able to offset the initial loss of large amounts of C stored in old-growth forests because of a nearly 15% reduction in NPP (Grant et al. 2010). Thus, the notion that short rotations in plantations can sequester C fails to account for the net loss in the long-term and at broader spatial scale (Harmon et al. 2009).

It is imperative to examine the mechanisms by which western redcedar and western hemlock forests in old ITRs accumulate large biomass C stocks including natural disturbance history and physiological and geographical attributes. These two species are shade-tolerant and well adapted to wet, toe-slope positions where they tend to accumulate high biomass; therefore, there is a strong association between their survival and geography in the drier interior of BC. For example, these attributes of western redcedar and western hemlock were found to protect landscapes from large-scale wildfire in the southern Washington Cascade range by creating fire breaks (Keeton and Franklin 2004). In addition, aspect (e.g., north vs. south-facing slopes) influences their survival by controlling local climates including soil

moisture and temperature. There is evidence that total ecosystem C of old-growth western redcedar and western hemlock on Vancouver Island were approximately 55% higher in the moister and cooler west than in the drier and warmer east; longer fire-return intervals in the west provided greater protection of these old-growth forests (Trofymow and Blackwell 1998). The north-facing aspect of Lunate Creek where long fire-return intervals are expected would likely favour the development of old-remnant cedar and hemlock in toe-slope positions. Although mean fire-return interval (not catastrophic stand-replacing fire) in wet and very wet subzones of the ICH ranges from 200 to 300 years (Wong et al. 2003; Sanborn et al. 2006), many forests on lower slopes and toe-slope positions contain trees over 500 years old (Stevenson et al. 2010). The landform-related spatial variation in total ecosystem C of old-boreal forests of hybrid white spruce and subalpine fir was also remarkable across soil moisture and nutrient gradients, ranging from 120 Mg C ha<sup>-1</sup> in poor site to 725 Mg C ha<sup>-1</sup> in very rich site (Kranabetter 2009). The observed spatial variation in the distribution of total ecosystem C underscores the importance of realizing the discernable productivity gradients present in old-growth forests at different spatial scales and geographic locations. Given the approximate stand age of the three ICH study areas (300 to 350 in East Twin and Minnow Creek and 450 to 500 in Lunate Creek), the estimated total forest C (excluding mineral soil C) in old ITRs may be at or close to the upper limit for wet and very wet forests in this region.

No direct or detailed assessments of mineral soil C in the inland rainforest have been made in the past, but model estimates for dead-organic-matter C stocks in Interior Cordilleran forests of Canada suggest that soil C (including forest floor and mineral soil) is about 192 Mg C ha<sup>-1</sup> (Kurz and Apps 1999). A simple subtraction of forest-floor C stock in this study from the estimated dead organic matter C indicates that mineral soil C in uncut old-growth stands would be roughly 146 Mg C ha<sup>-1</sup>. Previous biomass studies indicate that

the ratio of mineral soil C to total dead-organic-matter C in temperate old-growth forests of the Pacific Northwestern USA (Harmon et al. 1990; Smithwick et al. 2002; Harmon et al. 2004) and coastal BC (Trofymow and Blackwell 1998; Trofymow et al. 2008) ranged from 0.3 to 0.6, suggesting that mineral soil C stocks in old ITRs range from 46 to 160 Mg C ha<sup>-1</sup>. These estimates of mineral soil C stocks are close to the range of old-growth forests of Douglas-fir, western hemlock, and western redcedar in the Pacific Northwest, USA (100 to 150 Mg C ha<sup>-1</sup>) (Homann et al. 2004) and western redcedar and hemlock in coastal BC (47 to 154 Mg C ha<sup>-1</sup>) (Trofymow and Blackwell 1998; Trofymow et al. 2008). The additional C stocks (46 to 160 Mg C ha<sup>-1</sup>) for mineral soil suggests that our estimates of total forest C stocks (including mineral soil) in uncut old-growth stands could range from 501 to 615 Mg C ha<sup>-1</sup>.

#### 2.4.2 Dead organic matter (DOM) C

The influence of different harvesting methods on DOM C affects net C balance in forest ecosystems (Lal 2005; Jandl et al. 2007b); however, numerous studies have found large spatial and temporal variability in DOM C stocks, depending on site condition, climate, disturbance history, slope, and litter quality (Yanai et al. 2003; Hagemann et al. 2010a). Insignificant differences in DOM C stocks among treatment units in this study seem to reflect the spatial heterogeneity often observed in estimates of DOM C stocks.

Our estimates suggest that DOM C played an important role in affecting forest C stocks in old ITRs. DOM C stock in all cedar-hemlock old-growth stands in this study accounted for about 23% ( $107 \pm 19 \text{ Mg C ha}^{-1}$ ) of total forest C stocks with CWD and forest floor constituting 36% ( $39 \text{ Mg C ha}^{-1}$ ) and 43% ( $46 \text{ Mg C ha}^{-1}$ ) of the DOM C stocks, respectively. This C pool found in this study is similar to the low end of DOM C stocks (84

- 357 Mg C ha<sup>-1</sup>) found in very wet maritime subzones of the CWH (Coastal Western Hemlock) biogeoclimatic zone of BC, but exceeds that of very dry maritime subzones (27 – 49 Mg C ha<sup>-1</sup>) dominated by old western redcedar and hemlock (Keenan et al. 1993; Trofymow and Blackwell 1998). Coarse woody debris and the forest floor in the very wet maritime subzone accounted for 45% (161 Mg C ha<sup>-1</sup>) and 42% (151 Mg C ha<sup>-1</sup>), respectively (Keenan et al. 1993). Our estimate considerably exceeded DOM C stocks reported in oldgrowth stands of larch/Douglas-fir in the drier inland Northwestern USA (44 Mg C ha<sup>-1</sup>) (Bisbing et al. 2010) but was similar to old Douglas fir/western-hemlock stands of the Pacific Northwestern USA (123 Mg C ha<sup>-1</sup>) (Harmon et al. 1990). DOM C stocks on richer sites of old sub-boreal forests (95 – 110 Mg C ha<sup>-1</sup>) (Fredeen et al. 2005; Kranabetter 2009) were also similar to our estimate. Thus, old ITRs play a similar role in conserving DOM C as other cool, wet temperate forests of the Pacific Northwestern USA and coastal as well as wetter interior of BC.

Considerably greater C stocks of DOM at Luante Creek than the other two study sites partly relate to the age and size of western redcedar better adapted to wetter, cooler climate of Lunate Creek where long-fire return intervals are expected. Larger and older trees of western redcedar at Lunate Creek potentially led to considerably greater DOM C stocks. To a lesser extent, uncontrolled factors during harvest operations (e.g., individual decisions to either leave or take logs) may not be trivial in affecting amounts of CWD left post-harvest, although it is difficult to quantitatively detect these effects on the CWD C stocks. While utilization levels of East Twin Creek and Minnow Creek may be higher than those of Lunate Creek because of their closer proximity to mills, anecdotal evidence shows that utilization levels do not seem to be responsible for such stark differences in DOM C stocks between Lunate Creek and the other two study sites. The physical environments of Lunate Creek

(e.g., north-facing aspect, cool and wet climate) produces larger, older trees at this site, which may better explain the difference in DOM C stocks among three study sites.

Coarse woody debris has important functional and structural attributes in forest ecosystems, regulating nutrient budget and water, providing seedbeds for new cohorts, and storing C (Harmon et al. 1990; Laiho and Prescott 1999; Spears et al. 2003; Spears and Laitha 2004). Accumulations of CWD in old-growth forests of Oregon, Washington, and BC are among the highest in the world (Currie et al. 2003), contributing to important DOM C reservoirs. In this study, mean CWD C stocks (39 Mg C ha<sup>-1</sup>) in old-growth stands were considerably lower than those found in very wet maritime and very wet hyper maritime subzones of the CWH biogeoclimatic zone in BC (100 to 200 Mg C ha<sup>-1</sup>) but similar to those of the very dry maritime subzone (15 to 55 Mg C ha<sup>-1</sup>, assuming wood C% is 50) (Keenan et al. 1993; Feller 2003). Volumes reported across BC's coastal rainforests also support the evidence that drier CWH subzones are similar to wetter ICH subzones (Stevenson et al. 2010). Our estimates are also very similar to the regional average CWD stocks in the Pacific Northwestern USA (41 Mg C ha<sup>-1</sup>, range of 10 - 64 Mg C ha<sup>-1</sup>) (Smithwick et al. 2002). Limited CWD C stocks relative to the more mesic BC's coastal ecosystems (Harmon et al. 1986; Tinker and Knight 2000) in this study may reflect differences in the size and abundance of CWD resulting from low productivity of the dry interior from shorter growing seasons and shorter intervals between fires in ITRs (Stevenson et al. 2010).

In general, harvesting reduces long-term C stored in CWD via removal of its future sources such as aboveground living trees (Harmon 2002; Currie et al. 2003; Swanson 2009). Nevertheless, the influx of woody debris immediately following harvesting often leads to increasing the short-term CWD C pool (Wei et al. 1997; Martin et al. 2005; Taylor et al. 2008). Similarly, significantly greater C stocks of CWD in clear-cut as contrasted with

partial-cut stands across all the study areas in this study may have had the short-term influx of woody debris generated immediately after clear-cutting. Stevenson et al. (2006) indicated that a large quantity of CWD was left as non-merchantable logs during harvesting in these study areas due to the high incidence of heart-rot, which seems to be responsible for the significantly higher CWD C stocks observed in clear-cut stands, and thus may have masked the effect of harvesting in this study.

Post-harvest C stocks of CWD are determined by carryover from pre-harvest and additional amounts of CWD left post-harvest. The potential increase in post-harvest woody debris biomass in the event of a stand-level disturbance can be predicted based on the ratio of current dead-wood (CWD plus snag) to live-tree biomass estimates (Harmon 2009). For example, the ratio in old-growth stands ranges from 0.2 to 0.3 in evergreen conifers of the Pacific Northwestern USA (Harmon et al. 1990; Harmon and Sexton 1996; Harmon et al. 2004). In this study, the ratios found in uncut old-growth, group-selection, and groupretention sites were 0.17, 0.17, and 0.52, respectively, with the old-growth and group selection being very similar to the Pacific Northwestern USA (0.2 to 0.3). This implies that CWD C found in old-growth stands in this study might have the potential to increase three fold if a stand-level disturbance occurred, but at a cost of decreased living C stocks overall.

Amounts of legacy (residual) wood remaining after disturbance or at the start of succession have therefore a major impact on the pattern of net ecosystem C balance, but vary markedly by harvesting intensity and methods. Harmon (2009) indicated that more legacy wood after disturbance becomes a C source to the atmosphere in the short term but increases the sink strength in late succession or old-growth. This modelling study indicated that a 50-to 100-year rotation for harvesting resulted in accumulating only 45 and 59% of the average woody debris C of a 500-year natural cycle and C stocks never exceeding those occurring

during the 500-year natural cycle. Similarly, a chronosequence study in red pine forests across the Great Lakes States of the United States showed that harvested stands had only onefifth the amounts of woody debris of natural stands in early succession and one-third of the woody biomass of natural stands at the end of harvest rotation (Duvall and Grigal 1999). Although decomposition and stand mortality rates profoundly affect the amounts and persistence of CWD (Harmon 2002; Herrmann and Prescott 2008; Hagemann et al. 2010b), the accumulation of new wood from aboveground trees (varying age classes) was reported to increase the sink strength in the late succession (Wirth and Lichstein 2009). Clear-cut generally produces small and short logging debris (Densmore et al. 2004; Stevenson et al. 2006; Hagemann et al. 2010a) as compared with relatively large CWD piece sizes in natural forests, presumably leading to faster decomposition and low C pools in the long-term in clear-cuts. Nevertheless, inputs of CWD, whether from episodic events following harvesting or mortality during stand succession, contribute to long-term C stocks in the form of CWD (Laiho and Prescott 1999; Tinker and Knight 2000; Currie et al. 2003). Significantly greater snag C in old-growth stands compared to partial harvesting (22 vs. 4 and 10 Mg C ha<sup>-1</sup>) in this study undoubtedly assists in preserving long-term CWD C stocks in old-growth stands.

Numerous modelling studies have predicted greater CWD C stocks after partial-cut than after clear-cut harvesting through the retention of aboveground C stocks that generate future CWD (Taylor et al. 2008; Harmon 2009; Swanson 2009). Neilson et al. (2006) demonstrated that partial-cut harvesting enhanced C sequestration compared to clear-cutting in boreal mixed forests. The divergence of the responses between partial-cut harvesting and clear-cutting seems to be more evident among different rotation lengths and percent overstory retention. A 200-year rotation with overstory retention in southern beech forests of Chile exhibited lower reduction rate (-9.8%) of CWD C relative to old-growth stands as

contrasted with -14.0% in a 100-year rotation with the same overstory retention (Swanson 2009). CWD C stocks in partial-cut forests with a 200-year rotation were 87% above that of clear-cuts with the same rotation length in eastern Canadian red spruce forests (Taylor et al. 2008). These studies found that the shorter the harvest rotation, the greater the divergence in CWD C stocks between different overstory retention levels, with shorter-rotation clear-cutting having the lowest C pools (Harmon et al. 2009).

Decomposition of CWD can directly influence how rapidly forests become sinks for C following disturbance (Pregitzer and Euskirchen 2004). Although a typical climate - cold winter and cool summer – in the ICH study areas undoubtedly contributes to the long persistence of CWD (e.g., radiocarbon dating of decay class IV CWD indicated ages of 195-320 years) (Harmon et al. 1986), the more decay-prone heartwood of interior cedar than coastal cedar (Daniels and Russell 2007) may have a large effect on decomposition rates of CWD. Harmon (2009) indicated that the presence of a wood-decomposing pathogen causing heart-rot might affect future decomposition rates by short-circuiting decomposer colonization, while another study indicated that initial slow rates of biomass loss from older pine logs corresponded to exceedingly decomposed, brown-rotted logs that retained no sapwood or bark (Harmon et al. 2000). Similarly, the high incidence of heart-rot observed in old ITRs may initially slow mass loss of old cedars, but the presence of a wood-decomposing organism affecting the extensive heart-rot of interior cedars may or may not raise the speed of mass loss, as those logs proceed to the advanced state of decay. Harmon et al. (2000) indicated that large variation (i.e., five-fold variation) in annual wood decomposition rates was observed in more decay-resistant heartwood of the genus Thuja (k = 0.009; k is a decayrate constant) than decay-prone heartwood of *Abies* (k = 0.05) in 8 to 10°C temperature range. In contrast, the genus Tsuga was not responsive to the temperature change. Similar

annual decay rates were reported for western redcedar (0.009 to 0.01 k) and western hemlock (0.023 to 0.03 k) in Southern and Northern Interior regions of BC (Densmore et al. 2004; Laiho and Prescott 2004). While such physical characteristics as size and position can also affect decomposition rates (Harmon et al. 1986), these study results illustrate the importance of examining the contribution of decay-prone heartwood of interior cedar to the persistence of CWD. This may provide insight on the CWD C dynamics of stands dominated by interior cedar such as old ITRs.

Fragmentation of CWD through breakage or mechanical damage during harvesting can influence its biomass and C stocks through the alteration of decomposition rates. Investigators found that ICHvk2 and wk3 cutblocks had more than 70% of the post-harvest CWD volumes < 6 m long (Densmore et al. 2004), whereas Stevenson et al. (2006) discovered less extreme reduction in the piece length of CWD in clear-cut stands of the same subzones in the three ICH study areas. It was observed that winter harvesting reduced damage to the piece length of CWD (Stevenson et al. 2006). Due to the loss of structural integrity in decayed wood of old western redcedar, it is also likely that the high incidence of heart-rot facilitates the wood fragmentation if mechanical damage is extensive and thus may lead to faster tree- and stand-level decomposition rates (Li et al. 2007) via enhanced wood mineralization. Similarly, harvesting can directly affect the DOM dynamics by deliberate or inadvertent felling of snags, and accelerating the fragmentation, collapse, and settling of CWD, which are likely to increase the rate of C emissions to the atmosphere via increased respiration rates (Bond-Lamberty and Gower 2008). Fragmentation and leaching accounted for about one-third of total debris mass loss in a southern Appalachian forest, USA (Mattson et al. 1987). Fragmentation was also found to account for 10% of CWD mass loss in a North American Acer spp. wetland forest (Chueng and Brown 1995). Disturbance of old-growth

forests with the high incidence of heart-rot may lead to faster decomposition rates than young forests with few incidences of heart-rot via increased chance of fragmentation (Harmon 2009).

Tree mortality is an important ecological process in forest C dynamics but is highly unpredictable due to its complex interactions with multiple biotic and abiotic factors over space and time (Franklin et al. 1987; Harmon 2009). While understanding their interactive effects on dead wood C pools is beyond the scope of this study, increasing our understanding of mechanisms of tree mortality unique in old ITRs is critical to understanding C dynamics of dead woody debris. For example, tree mortality associated with competition and natural disturbances such as stand-replacing fires was found to be a significant biomass flux, particularly in older stands of boreal black spruce in Manitoba, Canada (Bond-Lamberty and Gower 2008). Although fire is one of the major stand- and landscape level disturbances in the Interior Cedar-Hemlock (ICH) biogeoclimatic zone, a western hemlock looper (Lambdina fiscellaria lugubrosa (Hulst)) outbreak was the most recent stand-replacing disturbances that defoliated significant proportions of stands dominated by cedar-hemlock as well as subalpine fir from 1992 to 1995 (Alfaro et al. 1999). Its primary host is cedar-hemlock, but subalpine fir was also found to be highly susceptible to the insect defoliator (Stevenson et al. 2010). Based on the Northern Wetbelt Silvicultural Systems Project report (Jull et al. 2002), over 50% of CWD species composition was western redcedar in East Twin and Lunate Creek, while western redcedar and subalpine fir in Minnow Creek accounted for 25 and 30%, respectively (data not shown). Thus, the large proportion of western redcedar and subalpine fir logs in this study may potentially be the legacy of the hemlock looper outbreak.

Tree mortality via western hemlock looper outbreaks may have turned our study areas into stands with a high proportion of CWD species composition identified as unknown

species (35%). Although it is not possible to identify the proportional allocation of the unknown species to particular species, the relatively small proportion (7 to 11%) of CWD species composition as western hemlock may indicate its contribution to unknown species as a result of continuous western hemlock looper outbreaks. According to the Northern Wetbelt Silvicultural Systems Project report (Jull et al. 2002), pre-harvest (in the year 1999) living and standing dead western hemlock trees accounted for only 0 to 13% (average 3%) and 0 to 16% (average 5%) of total basal areas (m<sup>2</sup> ha<sup>-1</sup>) in the three ICH study areas, respectively. These records suggest that other less severe outbreaks recorded in the ICH in 1952-57, 1963-65, and 1983 as well as severe outbreak in the 1990s (Alfaro et al. 1999) may have continuously killed hemlocks before they became mature. Presumably, after young hemlocks regenerating under the cedar canopy died, they became small downed logs that reached an advanced state of decay we identified as unknown species and/or as decaying wood incorporated into the forest floor.

It was also observed that the typical mortality causes in old-growth forest tend to shift from density-dependent to density-independent mortality such as species longevity, the presence of pathogen and insects, and susceptibility to wind (Franklin et al. 1987; Harmon 2009). Recent studies discovered that over 50% of mortalities across all the plots in oldgrowth forests dominated by white fir (*Abies concolor* Gordon & Glend.), Shasta red fir (*Abies magnifica* var. shastensis), incense cedar (*Calocedrus decurrens* [Torr.] Florin.), and sugar pine (*Pinus lambertiana* Dougl.) on the western slope of the Sierra Nevada of California was associated with biotic agents such as root rots and several bark beetles (Das et al. 2011). Long-lived western redcedar often out-competes western hemlock when cedar grows from the sub-canopy to canopy cover during stand succession (Daniels 2003), which often leads to stands dominated by western redcedar like those observed in the three ICH

study areas. Thus, even in the absence of stand-replacing disturbances, western hemlock may die and fall to the ground more rapidly than cedar during stand succession because of natural competition. Moreover, gap formation via senescence, heart-rot, and root rots (Stevenson et al. 2010) may also be an important mortality-related process creating the type of stand structure present in the old ITRs, while the high incidence of heart-rot should also affect trees that are already weakened by western hemlock looper, or vice versa. Because old-growth forests are a highly diverse mosaic of stand structures and stand developments, the large spatial variation in the tree mortality associated with competition and biotic agents was found to be more evident in old-growth forests such as ITRs (Das et al. 2011).

Overall, CWD C stocks in old ITRs are resilient to the short-term impacts of different harvesting methods, and leaving sufficient amounts of CWD may contribute to large C reservoirs. Nevertheless, the amount of future dead-woody-debris C varies in response to type of various retention-harvesting methods and rotation lengths as well as multiple mortality agents such as fire, wind, western hemlock looper, heart-rot, and competition. Given that harvesting directly or indirectly influences almost all these mortality agents and decomposition rates in space and time, our understanding of CWD C dynamics will require frequent observation of the CWD population over time.

Failure to statistically detect differences in forest-floor C between clear-cutting and old-growth stands in this study is common due to high spatial and temporal variability (Wallace and Freedman 1986; Prescott et al. 2000a; Yanai et al. 2003; Fredeen 2006; Shaw et al. 2008b; Nave et al. 2010). The use of appropriate sampling design has often been a challenge (Yanai et al. 2003; Shaw et al. 2008b), and the causal mechanisms of forest-floor C difference influence the magnitude of difference, and accordingly the ability to detect such differences. For example, forest-floor C stocks in old-growth forests of hybrid white spruce

and subalpine fir substantially decreased from  $78 \pm 54$  to  $27 \pm 6$  Mg C ha<sup>-1</sup> when converted to the second growth, but no difference was statistically detected due to the high spatial variability in old-growth stands (Fredeen 2006). The insignificant difference in forest-floor C of clear-cut relative to old-growth and group selection in this study may be explained by high accumulations of small logging residues including litter in clear-cut stands (Jandl et al. 2007b). Moreover, past studies have indicated that the recalcitrant nature of cedar-hemlock litter reduced the litter decomposition and led to the high accumulation of litter in the forest floor of BC's coastal rainforests (Keenan et al. 1996; Prescott et al. 2000b). Clear-cutting is often considered to stimulate microbial soil processes by exposing the forest floor to solar radiation and more precipitation (Prescott et al. 2000b; Jandl et al. 2007b). However, there is no consensus to support this among studies (Prescott et al. 2000b). In addition, winter harvesting conducted in the three ICH study areas may have avoided excessive mixing of forest floor into the upper mineral soil from mechanical disturbance (Ryan et al. 1992; Yanai et al. 2003). Our results indicate that the effect of clear-cutting on forest-floor C is difficult to statistically detect.

In this study, only group-retention harvesting significantly reduced mean forest-floor mass and C stock (33%) relative to old-growth stands across the study areas. This pattern is partly explained by more rapid and direct physical response of forest floor to mechanical disturbances such as logging equipment (Currie et al. 2003; Yanai et al. 2003; Nave et al. 2010), which mixes forest floor into upper mineral soil (Ryan et al. 1992). This response was also more pronounced in Podzolic soils, typically encountered in the three ICH study areas, where the high accumulation of C in the forest floor is usually observed (Nave et al. 2010). While winter harvesting conducted in the three study areas may have avoided the excessive mechanical disruption, the observed reduction (i.e., 33%) of forest floor C stock in group-

retention cut relative to old-growth stands could have been intensified by summer harvesting. Direct changes in amounts of litter inputs from remnant trees after harvesting clearly reflect changes in forest-floor C stocks (Thornley and Cannell 2000; Currie et al. 2003; Jandl et al. 2007b; Taylor et al. 2008; Swanson 2009). Given that the relative proportion of forest-floor C stocks to old-growth stands was consistently lower in group-retention cuts than in groupselection cuts within and across the three ICH study areas, this response seems to be a direct reflection of the lesser aboveground litter inputs in a treatment with lower overstory retention (i.e., group retention).

Partial-cut harvesting with long-rotation lengths has the potential of sustaining NPP (net primary productivity) and thus maintaining a high litter input to forest floor, generating nearly twice as much C in forests compared with clear-cut harvesting (Thornley and Cannell 2000; Currie et al. 2003; Jandl et al. 2007b; Taylor et al. 2008). Securing continuous litter inputs into humus layer was reported to accumulate C in forest floors (Gleixner et al. 2009; Prescott 2010), implying that partial-cut harvesting, particularly group-selection harvesting, has the potential to generate forest-floor C stocks. Indeed, insignificant difference in forest-floor C between group-selection harvesting and uncut old-growth stands across the study areas in this study indicates that litter production and inputs to the forest floor were similar to those of old-growth stands.

Mean forest-floor C stocks found in old-growth stands was 46 Mg C ha<sup>-1</sup>, which is similar to the mean C stocks (36 Mg C ha<sup>-1</sup>) of Podzolic soils in Canada's forested area (Shaw et al. 2008a) and 26 to 41 Mg C ha<sup>-1</sup> in montane and coastal Pacific Northwestern USA (Smithwick et al. 2002; Harmon et al. 2004). The C stock in this study also falls between 5 and 50 Mg C ha<sup>-1</sup> of forest-floor C (Podzolic soils) observed in global temperate forests (Nave et al. 2010) but lower than the 58 to 151 Mg C ha<sup>-1</sup> in old cedar-hemlock stands

of coastal BC (Keenan et al. 1993; Trofymow et al. 2008). The lower C stock in this study than coastal counterparts may reflect slightly different pedological processes between the interior and coastal regions (Shaw et al. 2008a). Although winter air temperatures can be very cold in the ICH, soil temperature is moderated by deep, insulting snowpack (Stevenson et al. 2010). As a result, Podzolic soils in the ICH continue soil physical, chemical and biological activities even during winter, as those of the CWH do, and this may account for similar litter decomposition rates between ICH and CWH sites (Prescott et al. 2000b). This suggests that the seasonal length of podzolization processes may be similar between interior and coast, but the extent of the process varies because of lower net primary productivity in drier interior as compared with wetter coast. This leads to lower accumulation of litter in the forest floor.

The connection between CWD and forest-floor pools plays an important role in regulating DOM C in forest ecosystems (Harmon 2002; Currie et al. 2003; Moroni et al. 2010a). It has been realized in old black-spruce forests in the high boreal of eastern Canada that once woody debris was incorporated in the forest floor, the buried woody debris persisted far longer than when it was resting on the soil surface (Moroni et al. 2010a). The decay rate of the buried wood was substantially reduced once the logs were covered by bryophytes; as a result, wood was found to remain over 250 years. Similar observations were also made in old-growth Douglas-fir, western hemlock, and western redcedar stands of the Pacific Northwestern USA where decay-class V logs may persist longer than 200 years in the soil (Sollins et al. 1987). Because the buried woody debris is higher in C concentrations than non-woody debris in the forest floor, maintaining the buried woody debris can contribute to retaining DOM C pools in the long-term. In this study, decaying wood accounted for 31% (clear-cut), 30% (group retention), and 25% (group selection) of mean forest-floor C stocks

with old-growth stands accounting for 43%. In other studies, decaying wood accounted for 5 to 70% of the forest floor biomass in the Olympic National Forests, 63% in the old-growth cedar-hemlock stands and second-growth hemlock-fir stands (Keenan et al. 1993), and up to 54% in northern coniferous forests (Laiho and Prescott 2004). In cool, humid coniferous ecosystems downed-woody debris may become an important C stock following burial with bryophytes stabilizing the woody C by slowing down the decomposition rates (Moroni et al. 2010b). This suggests that the retention of large logs in old ITRs may contribute to long-lived DOM C pools.

#### **Chapter 3. Conclusions and management recommendations**

While this study did not examine the effect of climate change on total forest C stocks, it is clear that old-growth forests around the world are currently facing unprecedented challenges resulting directly or indirectly from climate change. Recent studies have reported rapid and pervasive increases in mortality rates of old-growth forests in the Pacific Northwestern USA due to the recent warming and drought (Adams et al. 2009; van Mantgem et al. 2009). Modellers have raised the possibility that increased natural disturbances such as insect and fire will result in a net efflux of C from forests (Breshears and Allen 2002; Kurz et al. 2008a; Kurz et al. 2008b), surpassing expected C uptake via increased NPP induced by rising temperatures or rainfall (Kurz et al. 2008a). Mean annual temperature in old ITRs is projected to rise 2 to 3°C by the year 2070 (Stevenson et al. 2010), and fire frequency by nearly 78 - 110% in all ecozones of Canada, despite its wide local variation (Flannigan et al. 2005). Gavin et al. (2009) indicated that dramatic changes in moisture availability in toe slopes induced by recent climatic fluctuations may jeopardize western redcedar regeneration in ITRs. Increased frequency and severity of natural disturbances imply that management of C stocks in old ITRs must accommodate such climate-induced uncertainty. When we factor in management impacts found in this study, the integrated effects will generate even higher uncertainty in future forest C stocks.

The forecast of such climatic fluctuations poses a question of what the best management options will be to maintain or increase C sequestration in old ITRs. Management of forest ecosystem C in old ITRs needs to encompass uncertainties in both climate and management impacts. Given large uncertainty in the impacts of future climatic changes, conserving existing ecosystem function and diversity present in old-growth forests are better alternatives to accommodate such uncertainty. Past scientific reports proposed that

protecting present temperate and boreal forest communities that still store vast amounts of C can adequately mitigate climate change effects (Bradshaw et al. 2009). Retention of biodiversity in old-growth forests may help avoid the loss of large reservoirs of C through their adaptation to constantly changing environment (Mackey et al. 2008). The maintenance of genetic and taxonomic traits unique in old ITRs has the potential to combat climate-induced uncertainties; therefore, losing the natural complexity therein via intensive logging may eventually accelerate the degradation of C sequestration capacity. Intact mature and old forests (> 140 years old) still account for 55% of the total forested area in wet and very wet cool subzones of the ICH biogeoclimatic zone (Stevenson et al. 2010). Conservation of these intact forests may at the same time conserve important C reservoirs.

Silvicultural treatments that maintain ecological processes, ecosystem functions, and stand development are therefore likely to maintain long-term forest C stocks in old ITRs. Such silvicultural practices seem to be valuable, particularly in drought-sensitive wet and very wet, cool parts of the ICH where large western redcedars can accumulate large biomass C stocks. In this study, group-selection harvest with intermediate/long rotations may confer great benefits for storing C in old ITRs due to fast recovery time because trees are retained within patches. It is often perceived that short-rotation intensive harvesting can store more C in forest products; however, increasing the rate of C stored in forest products via shorter rotations has been shown to do little to increase long-term C stocks (Harmon et al. 1990), and would not be able to offset losses of the large reservoirs of C stored in old-growth forests (Harmon et al. 2009).

According to the Northern Wetbelt Silvicultural Systems Project report, a 50-50 mix of western redcedar and fast-growing hybrid white spruce seedlings were planted in the three ICH study areas in the years 2001 and 2002 (Jull et al. 2002). The performance of the

planted seedlings will markedly affect the ability to store future C stocks in old ITRs. If natural regeneration of western redcedar, western hemlock, and subalpine fir successfully supplements the performance of the planted seedlings, the total regeneration success may partially be able to offset the loss of economic benefits due to rising conservation efforts. This may supply sufficient forest products to meet commercial demands. Benefits of intermediate rotations over longer ones by replacing slow-growing species with fast-growing species were also indicated to be very effective in stands where there is a major decline in live C stocks in older stands (Johnson and Curtis 2001). Partial-cut harvesting with the success of both planted seedlings and natural regeneration may provide the ability to maintain compositional, structural and functional attributes of old-growth forests but simultaneously meet socioeconomic needs (Thornley and Cannell 2000; Liski et al. 2001; Harmon and Marks 2002; Gutrich and Howarth 2007; Taylor et al. 2008; Harmon et al. 2009). Thus, the sequestration and storage of long-term C stocks in old ITRs require continued monitoring of the performance of live tree, snags, and CWD in the four silvicultural systems, including the performance of regenerating planted seedlings in the openings of various sizes.

Given ongoing initiatives of the BC provincial government and forest industry for sequestering forest C stocks to reduce greenhouse gas (GHG) emissions (Greig and Bull 2009), a precautionary approach potentially satisfies such initiatives in old ITRs. Stevenson et al. (2010) suggested managing ITRs by zoning the landscapes into three zones: protected zone, structural diversity zone, and intensive management zone. The backbone of the zoning system stems from a precautionary approach embedded in ecosystem-based management where protecting natural complexity and diversity inherent in old ITRs will be the best approach to meet social, cultural, and ecological values. In BC, there is a potential shift in forest management from managing forests for conventional forest products to managing
forests for C sequestration and carbon credit generation (Greig and Bull 2009). BC's forest industry has already undertaken initiatives to reduce the C footprint in response to the government initiatives. Under the ongoing movement toward low-carbon economy in BC, conserving old ITRs and restricting extensive harvesting appear to go in tandem with such movements. One advantage of the precautionary approach will be that the increased C sequestration as a result of reforestation and conservation efforts may, for example, be saved as carbon credits after these management actions are approved to generate C benefits (i.e., additionality). On the other hand, a forest management plan can only gain additionality above some reference level or "baseline" value, but uncertainty may still remain about how to determine an appropriate baseline value for old-growth such as old ITRs that already store large reservoirs of C. It is argued that the baseline value depends on estimates of initial and forecasted carbon stocks and can significantly affect if a management plan becomes C positive or negative (McKinley et al. 2011); therefore, estimates of additionality may not be straightforward in the case of old-growth forests.

In this study, we could identify areas of future improvements to more accurately evaluate management impacts on forest C stocks in old ITRs. A major limitation of our study is that an unknown, large-scale uncertainty remains over our live-tree C stocks (especially for western redcedar) estimates associated with decay-loss factors used. While each curve of the decay-loss factors reaches an asymptote beyond a certain DBH, anecdotal evidence often shows that interior western redcedar may have a higher incidence of heart-rot and hollow with greater DBHs. Nonetheless, the curve for live western redcedar, for example, assumes that beyond a 100-cm DBH the loss factor levels off and stays the same till the curve hits the maximum diameter of 249-cm. The frequency distribution of live cedars as a function of DBH in this study is right skewed with those beyond 100-cm DBH accounting

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for only 7.4%. Despite the small proportion of live cedars occupies where the decay-loss factor curve is flat, the relative contribution of these large cedars to total live-tree C stocks may be high because of their size. The absence of potential bounds of error around the DBHs of large stems (> 100 cm) can therefore lead to large, unexplained bias. The greater sensitivity of our live-tree C stocks estimates to decay-loss factors demonstrates the need for a better representation of the loss curves for greater DBHs, especially for stands dominated by large western redcedars, as well as potential bounds of error around the loss factors (%). Once such bounds of error are made available, a Monte Carlo approach can then be used to estimate the relative contribution of the decay-loss factors to total uncertainty in live-tree C stocks, which will improve our assessments of live-tree C stocks with accounting for internal decay and hollow. To a lesser extent, our live-tree C stock estimates are also sensitive to estimated decayed-wood density of western redcedar. For instance, we collected only three replicates of decayed-wood samples of cedar downed logs beyond a 100-cm DBH at Lunate Creek (Figure 4). This, in combination with the unexplained uncertainty associated with the decay-loss factors stated above, further increases the total uncertainty in live-tree C stocks that were not accounted for in this study. Thus, a better representation of decayed-wood density of cedar and hemlock requires sampling over a wide range of diameter classes at broader spatial scales in the future to improve our estimates of live-tree C stocks that reflect wetter, cool parts of inland temperate rainforests.

The next important is that ITR-specific tree biomass allometric equations are needed to reduce the high uncertainty associated with existing allometric equations (i.e., to properly capture the large variation in stands dominated by large western redcedars). Because live trees make up a large proportion of total forest C stocks, more robust allometric equations are able to reduce such high uncertainty. While development of new allometric equations often

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entails tremendous amounts of time and cost, past studies may provide useful information for addressing these resource constraints. For example, Ketterings et al. (2001) developed new tree allometric equations for tropical forests by calibrating existing tree allometric equation parameters based on information that can be more readily gathered such as the average wood density and the site-specific relationship between height and diameter. Given that such information as the height-diameter relationship is also available at the three ICH study areas, this method may be useful in building new tree biomass allometric equations that specifically target old ITRs. Thirdly, accurate assessment of CWD C stocks will require information about ITR-specific decay-class wood densities for western redcedar as well as western hemlock, as the information is scarce in the BC interior. Finally, although we did not measure mineral soil C stocks, these C stocks may profoundly increase uncertainty in our total forest C stocks estimates in old-growth ITRs due to the large spatial heterogeneity of soil. Estimation of mineral soil C stocks in the future will not only increase accountability of landscape-level or regional C inventory but also help to make more accurate assessments of uncertainty in our total forest C stocks in old-growth ITRs.

Like other old-growth temperate forests, old ITRs are important C reservoirs, and managing these forests for C sequestration will require efforts in developing strategies that seek to maintain their old-growth attributes. However, quantifying current C stocks and evaluating various retention-harvest impacts on their C at the landscape level is of particular importance to properly value C in old ITRs. Old ITRs were found to be resilient to lowintensity partial harvesting but highly susceptible to intensive logging (clear-cutting and group retention). The large contribution of live trees to total forest C stocks in this study underscores the importance of retaining live trees as major long-term C reservoirs and also demonstrated the need to recognize the unique dynamic system between live and dead pools

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in old ITRs. For example, the approximate maximum threshold of total forest C reaching at an age below 200 years in Douglas-fir stands will exhibit very different spatial and temporal pattern in live and dead C pools from cedar-dominated stands reaching this stage at an age beyond 600 years (Schulze et al. 2009). The high incidence of heart-rot in cedar-dominated stands may lead to lower C stocks compared to those dominated by western hemlock (Harmon et al. 2009), but this loss can be offset by the long life and large size of western redcedar. Thus, understanding the old-ITR specific ecosystem function and species in response to different harvesting methods provides a great deal of information on guiding forest management for C sequestration. This study demonstrated that group-selection cut provides a good compromise between forest harvesting to provide wood products and maintenance of forest C stocks, but intensive harvesting may lead to an extended period of recovery to restore C stocks to pre-harvest levels, making the future C trajectory highly uncertain. Nonetheless, we still need a better scientific underpinning to identify the best partial harvest patterns and rotation lengths for sequestering C stocks at various temporal and spatial scales. Until such scientific information is available, ecosystem management that aims to sustain C stored in old ITRs is a valuable choice to satisfy environmental, economic and social needs into the future.

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# Appendices

#### 1 Steps in the Monte Carlo calculation of live-tree C and its uncertainty

In this study, the following calculation steps were used to estimate live-tree biomass and C stocks for all the species in the three study areas (Figure A1 and A2). All the steps including A and B in the flowcharts below were executed at each iteration using STATA (StataCorp LP 2009).

- A. After Monte Carlo simulation began, sample parameter (or model error) values such as root mean square (i.e., standard deviation) of tree biomass allometric equations, C concentration, root allometric equation were randomly selected assuming that these errors were normally distributed. This means that each iteration provided slightly different parameter or error values depending on the magnitude of uncertainty to be selected.
- **B.** The entire step of **B** was applied to all the trees observed in the three study areas (Figure A2).
  - **B1.** Individual tree stem-wood biomass for each species was calculated using tree biomass allometric equations.
  - B2. The calculated biomass of stem wood was converted to wood volume (m<sup>3</sup>) using species-specific wood density for each species (Table A1) (Taylor 1997). This conversion was necessary to apply volume-based decay-correction factors to the stem wood at the next stage.

Species	Wood density (g cm <sup>-3</sup> )
Alder	0.4
Subalpine fir	0.4
Western redcedar	0.33
Douglas-fir	0.43
Western hemlock	0.43
Hybrid white spruce	0.38

Table A1. Species-specific relative wood density (oven dried mass per fresh volume) used in the Monte Carlo calculation of live-tree C.

Note: The wood densities were derived from Taylor1997.

- B3. To separate the estimated wood volume into sound- and decayed-wood volumes, volume-based decay-loss factors from TREEVOL (BC Integrated Land and Management Bureau 2003) were multiplied by the wood volume. This step was only applied to western redcedar and western hemlock.
- B4. Sound-wood volumes estimated in the last step were converted back to biomass (Mg) because of no need to correct for heart-rot and hollow. This indicates that species with no indication of heart-rot or hollow such as Alder, subalpine fir, Douglas-fir, and hybrid white spruce proceeded straight to the biomass calculation.
- B5. For western redcedar and western hemlock, the calculated decayed-wood volumes were corrected for hollow by multiplying four hollow proportions: 100, 50, 20, and 0%. Therefore, hollow proportions mean the volume percentage of internal stem decay which is actually hollow.
- **B6.** Only decayed-wood volumes (without hollow) of western redcedar and western hemlock remained at this step. Decayed-wood biomass (Mg) of these species was then computed using species specific decayed-wood density: 0.12 (cedar)

and 0.22 g cm<sup>-3</sup> (hemlock). We assumed that the decayed-wood density of western hemlock was half the sound wood.

**B7.** Finally, both sound- and decayed-wood biomass was combined to represent a single estimate of individual tree biomass with accounting for heart-rot and hollow.



Figure A1. Flowchart of the steps in the Monte Carlo calculation of mean C stock and uncertainty in live trees.



Figure A2. Sub-steps in the flowchart of the Monte Carlo calculation of live-tree biomass and C stocks described in Figure A1.

# 2 Estimates of total forest C stocks at 50, 20, and 0% hollow proportions

Table A2. Mean live-tree, dead organic matter (DOM), and total forest C stocks ( $\pm$  95% confidence interval) in four forest harvesting treatment units at three ICH study areas. For this case, 50% of the internal stem decay volume was assumed to be hollow.

Study area	Treatment unit		Live tree			DOM§	Total forest <sup>  </sup>
		$\mathbf{n}^{\dagger}$	Aboveground <sup>‡</sup>	Root	Total		
East Twin Creek	CC	0	0±0	0±0	0±0	77±19	77±19
	GS	8	210±117	47±26	256±143	65±18	321±144
	UN	8	268±108	60±24	328±133	71±14	399±133
Lunate Creek	CC	0	0±0	0±0	0±0	135±42	135±42
	GR	8	87±44	19±10	107±53	67±12	174±54
	GS	8	414±301	92±66	503±364	10 <del>6±</del> 27	610±365
	UN	8	364±210	81±47	445±259	147±46	592±264
Minnow Creek	CC	0	0±0	0±0	0±0	78±15	78±15
	GR	8	58±24	13±5	71±29	83±13	154±31
	GS	8	162±67	36±15	199±82	89±14	288±83
	UN	8	253±101	56±23	309±123	101±18	410±124
All study areas	CC	0	0±0	0±0	0±0	99±14 a	99±14 a
	GR	16	72±31 a	16±7 a	89±38 a	74±15 a	163±41 b
	GS	24	261±153 b	58±34 b	320±186 b	88±15 a	408±187 c
	UN	24	295±132 b	66±29 b	361±160 b	107±19 a	468±161 c

Note: Means across all study areas within a column sharing the same letter are not statistically significant using 95% confidence intervals generated from Monte Carlos simulation.

<sup>\*</sup>CC, GR, GS, and UN represent clear-cut, group-retention, group-selection, and uncut, respectively.

<sup>†</sup>n represents the number of large-tree permanent sample plot of live-tree C stocks.

<sup>‡</sup>Aboveground includes stem wood, bark, branch, and foliage.

<sup>§</sup>DOM (dead organic matter) includes snag, CWD, and forest floor.

<sup>h</sup>Total forest C includes live tree and DOM but excludes mineral soil C which was not measured in this study.

Study area	Treatment unit		Live tree			DOM§	Total forest
		$\mathbf{n}^{\dagger}$	Aboveground <sup>‡</sup>	Root	Total		
East Twin Creek	CC	0	0±0	0±0	0±0	77±19	77±19
	GS	8	214±118	48±26	261±145	65±18	326±146
	UN	8	273±110	61±24	334±135	71±14	405±136
Lunate Creek	CC	0	0±0	0±0	0±0	135±42	135±42
	GR	8	8 <del>9±</del> 45	20±10	109±55	67±12	176±56
	GS	8	423±305	94±68	518±373	106±27	624±374
	UN	8	372±215	83±48	454±260	147±46	601±263
Minnow Creek	CC	0	0±0	0±0	0±0	78±15	78±15
	GR	8	58±24	13±5	71±29	83±13	154±32
	GS	8	165±68	37±15	201±84	8 <del>9±</del> 14	290±85
	UN	8	257±102	57±23	315±126	101±18	415±128
all study areas	CC	0	0±0	0±0	0±0	99±14 a	99±14 a
	GR	16	74±33 a	16±7 a	90±40 a	74±15 a	165±43 b
	GS	24	268±162 b	60±36 b	328±200 b	88±15 a	416±200 c
	UN	24	301±139 b	67±31 b	369±172 b	107±19 a	476±173 c

Table A3. Mean live-tree, dead organic matter (DOM), and total forest C stocks ( $\pm$  95% confidence interval) in four forest harvesting treatment units at three ICH study areas. For this case, 20% of the internal stem decay volume was assumed to be hollow.

Note: Means across all study areas within a column sharing the same letter are not statistically significant using 95% confidence intervals generated from Monte Carlos simulation.

CC, GR, GS, and UN represent clear-cut, group-retention, group-selection, and uncut, respectively.

<sup>†</sup>n represents the number of large-tree permanent sample plot of live-tree C stocks.

<sup>‡</sup>Aboveground includes stem wood, bark, branch, and foliage.

<sup>§</sup>DOM (dead organic matter) includes snag, CWD, and forest floor.

Total forest C includes live tree and DOM but excludes mineral soil C which was not measured in this study.

Study area	Treatment unit		Live tree			DOM§	Total forest
		$\mathbf{n}^{\dagger}$	Aboveground <sup>‡</sup>	Root	Total		
East Twin Creek	CC	0	0±0	0±0	0±0	77±19	77±19
	GS	8	216±119	48±26	264±144	65±18	330±146
	UN	8	276±110	61±24	338±134	71±14	40 <del>9±</del> 134
Lunate Creek	CC	0	0±0	0±0	0±0	135±42	135±42
	GR	8	90±45	20±10	110±54	67±12	177±56
	GS	8	<b>429±303</b>	95±68	525±374	106±27	631±376
	UN	8	377±216	84±48	461±263	147±46	608±268
Minnow Creek	CC	0	0±0	0±0	0±0	78±15	78±15
	GR	8	59±24	13±5	72±30	83±13	155±32
	GS	8	166±68	37±15	203±84	89±14	293±85
	UN	8	259±103	58±23	317±126	101±18	418±127
all study areas	CC	0	0±0	0±0	0±0	99±14 a	99±14 a
	GR	16	74±33 a	17±7 a	91±40 a	74±15 a	165±42 b
	GS	24	270±159 b	60±36 b	330±195 b	88±15 a	418±195 c
	UN	24	304±138 b	68±31 b	372±168 b	107±19 a	479±169 c

Table A4. Mean live-tree, dead organic matter (DOM), and total forest C stocks ( $\pm$  95% confidence interval) in four forest harvesting treatment units at three ICH study areas. For this case, 0% of the internal stem decay volume was assumed to be hollow.

Note: Means across all study areas within a column sharing the same letter are not statistically significant using 95% confidence intervals generated from Monte Carlos simulation.

<sup>\*</sup>CC, GR, GS, and UN represent clear-cut, group-retention, group-selection, and uncut, respectively.

<sup>†</sup>n represents the number of large-tree permanent sample plot of live-tree C stocks.

<sup>‡</sup>Aboveground includes stem wood, bark, branch, and foliage.

<sup>§</sup>DOM (dead organic matter) includes snag, CWD, and forest floor.

<sup>II</sup>Total forest C includes live tree and DOM but excludes mineral soil C which was not measured in this study.

## Forest floor line transect selection

Forest-floor C stocks were sampled using line-transect method. The location of the line transects was pre-determined based on the information on existing coordinates, Geographic Information System (GIS), site maps, and aerial photographs. Except clear-cut treatment unit, all the line transects were initiated at established plots (i.e., permanent sample plots, vegetation plots, and regeneration plots) that were randomly chosen in advance. In the field, a coloured iron stake was identified at each plot as a reference point where a random bearing was taken for establishing line transect. In group-retention and group-selection cut treatment units, four plots were randomly selected in harvested and unharvested areas (i.e., in total, 8 permanent sample plots for each partial-cut treatment). In clear-cut stands, we randomly selected four reference points in advance using GIS where a line transect was drawn at the points via random bearing. All the existing 8 permanent sample plots were selected in uncut stands and a line transect was measured out between two iron stakes established at the permanent sample plot in a walking direction.