



## RESEARCH EXTENSION NOTE

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# RECENT CHANGES IN ABOVEGROUND LIVE-TREE CARBON IN SPRUCE AND FIR-DOMINATED SUB-BOREAL FORESTS IN THE BC INTERIOR

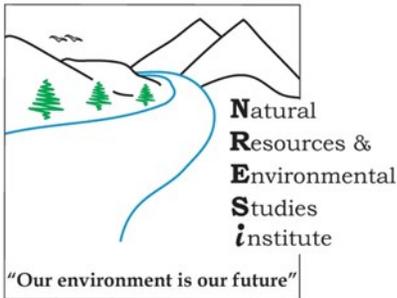
BY

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# Recent Changes in Aboveground Live-Tree Carbon in Spruce and Fir-Dominated Sub-Boreal Forests in the BC Interior

## Abstract

Given the accelerating rates of climate change and reports of increased tree mortality, we decided to re-evaluate recent changes (2003/2004 versus 2018) in live-tree carbon (C) stocks for conifer-dominated forest plots within the sub-boreal spruce (SBS) zone at the Aleza Lake Research Forest (ALRF) in central British Columbia, Canada. The diameters of all small (2.5-9 cm DBH) and large (>9cm DBH) live conifers and deciduous trees were initially measured in the summers of 2003 & 2004 as part of a larger study and remeasured in 2018 for 26 National Forest Inventory (NFI) C plots. Allometric relationships between tree diameter (DBH) and aboveground tree C stocks were applied to contrast recent changes in above ground live forest carbon stocks (i.e. between 2003/04 and 2018) at the tree or stand level. On average, total aboveground live-tree C

stocks increased with increases in DBH. However, aboveground live-tree C stock changes (2018 minus 2003 or 2004) were observed to decline with forest DBH, from C gains in 'younger' (lower maximum DBH) stands to C losses in 'older' (higher maximum DBH) stands. Carbon stocks of the most common tree species in the ALRF; subalpine fir (Bl: *Abies lasiocarpa*), interior spruce (Sx: *Picea glauca* x *engelmannii*) and paper birch (Ep: *Betula papyrifera*), were all increasingly affected by mortality with increasing size. Spruce trees are the dominant trees in older stands at the ALRF, and the single largest spruce in each plot contains around 25% of plot level live-tree C. We examine potential explanations, e.g. climate change, for increased large tree mortality rates and the decreased stand -level C observed in our study.

## Introduction

Forests represent a majority of ecosystem carbon (C), containing as much as 80% of all aboveground terrestrial C (Dixon *et al.* 1994). Many studies suggest that forest C stocks and uptake have been reduced in recent decades as a result of land-use change (i.e. forest degradation), on-going climatic change (Caspersen *et al.* 2000, Houghton 2005, IPCC 2007), including increased drought and heat-induced tree mortality worldwide (Allen *et al.* 2010). Thus, conserving forest C stocks and reducing forest C emissions remain a significant concern for managing greenhouse gases and climate change (Price *et al.* 1996, Canadell *et al.* 2004, Eggleston *et al.* 2006, Smith *et al.* 2014, IPCC 2018).

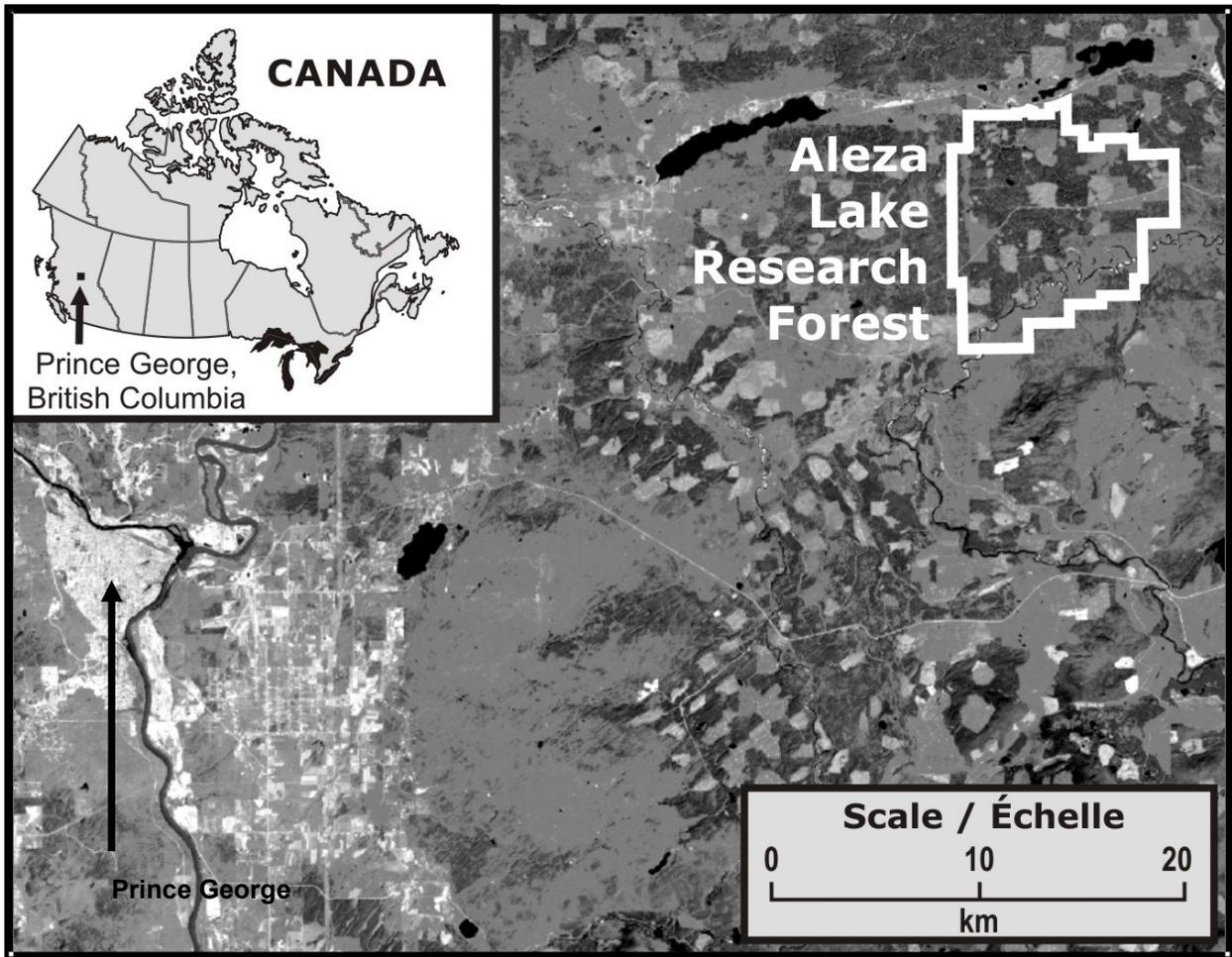
In Canada, 60% of the land surface is covered by boreal forests (Roi 2019), giving them a particularly important role in forest C storage. Current aboveground boreal forest C stocks have been estimated to be as high as ~ 154 Pg C (Bradshaw and Warkentin 2015). These boreal C stocks may be particularly vulnerable to climate change because of the greater temperature increases that have been observed at northern high latitudes (IPCC 2007). As a result, concern over boreal C stocks and the potential for a decline in net C uptake in boreal C in the near future has been expressed (Bradshaw and Warkentin 2015, Allen *et al.* 2010, Hessburg *et al.* 2019). Forest managers (Canadell and Raupach 2008; Laflouer *et al.* 2016), will therefore need to make informed decisions to safeguard Canadian forests and their associated C stocks. To facilitate these decisions, an improved understanding of boreal forest C stocks and fluxes, and their vulnerability to future climate scenarios, is needed (Nie *et al.* 2017, Jochem *et al.* 2010).

Forest aboveground biomass is often the most important and dynamic forest C stock (Brovkina *et al.* 2017; He *et al.* 2012). However, the future of C storage in boreal forests remains uncertain given the impacts of

disturbances and varied management and harvesting practices that alter forest structure and composition (Millar *et al.* 2007, Kerhoulas *et al.* 2013, Laflouer *et al.* 2016).

Common natural disturbances in boreal forests (i.e. fires and insect outbreaks), as well as climate change related drought and temperature events, are thought to have increased in severity, size and frequency in recent decades (Fleming 2000, Chen and Popadiouk 2002, Bond-Lamberty *et al.* 2007, Huang *et al.* 2017), all of which could have large consequences for forest C at the landscape level. In addition to natural disturbances, forest management that alters the composition, structure and functional processes of forests (Franklin *et al.* 2002, Liu *et al.* 2011, Goward *et al.* 2008), can alter forests ability to capture and store C. For example, a recent analysis of BC government forest data show that only 2.7% of old forest stands remain, largely as a result of forest harvesting (Price *et al.* 2020). These forest types contain the largest C stocks in the BC interior (Fredeen *et al.* 2005, Bois *et al.* 2009, Matsuzaki *et al.* 2013).

In this study, we examined recent changes in aboveground C in hybrid interior spruce and subalpine fir dominated stands at the Aleza Lake Research Forest (ALRF) in the central interior of BC. To do this we revisited forest C plots established and measured in 2003 and 2004 (see Fredeen *et al.* 2005, Bois *et al.* 2009) in 2018 that spanned young to old stands. Specifically, we wanted to determine how aboveground C storage in live trees was affected by species and tree size-class (DBH) in the 14 to 15 years since first measurement.



**Figure 1.** Location map of the Aleza Lake Research Forest (ALRF) with respect to Prince George, British Columbia, Canada.

## Methods and Materials

### Study area

This study was conducted in sub-boreal spruce and fir forests within the University of Northern BC (UNBC) Aleza Lake Research Forest (ALRF: Fig. 1), located 60 km east of the UNBC, Prince George, BC (N.54° 07', W.122° 04.3'). The climate is sub-boreal, with a mean annual precipitation (MAP) of 895 mm and a mean annual temperature (MAT) of 3°C (Jull and Karjala 2005). Our study sites were dominated by mature mixed stands of hybrid white spruce (*Picea engelmannii* Parry ex Engelm x *Picea glauca* (Moench) Voss) and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt), with paper birch (*Betula papyrifera* Marsh) and Douglas-fir (*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco) present in some stands. Recent canopy openings are frequently occupied by trembling aspen (*Populus tremuloides* Michx.) and willow (*Salix* spp.). In addition to sampling intact forest, we remeasured a single plot from a clearcut that had been planted to lodgepole pine (*Pinus contorta* Dougl. ex Loud. var. *latifolia* Engelm.), a practice which is not uncommon in the region.

### ALRF and live-tree measurements

A total of 154 comprehensive NFI forest C-plots were identified using a stratified random framework, with sub-boreal spruce [biogeoclimatic zone] forests stratified according to age, soil type and ecosystem

classification. Plots were established across the ALRF in the summer 2003 to 2005 (see Bois *et al.* 2009; Fredeen *et al.* 2005; Janzen *et al.* 2006). Of these, 51 plots established in 2003 or 2004 fell within an area of the ALRF that was imaged using an airborne LiDAR system in 2006 (Janzen, D. *et al.* 2008 unpublished, UNBC). We sought to remeasure these 51 plots to facilitate modeling and/or validation of landscape-level forest C changes at the ALRF. However, of these 51 LiDAR imaged plots, only 26 unharvested plots were located in 2018. These 26 plots were remeasured for aboveground C stocks using the same approaches used in the original study. Data collection and methodology for measuring the aboveground live-tree C stocks were based on the National Forest Inventory (NFI) Ground Sampling Guidelines of the Canadian Forest Service (Canadian Forest Inventory Committee 2002, version 1.1), as previously outlined in Fredeen *et al.* (2005). Briefly, all 'large' live trees with a stem diameter at breast height (DBH) >9 cm at 1.3 m above ground level were identified at the species level and their DBH measured with a diameter tape within a 400 m<sup>2</sup> (11.28 m radius) circular plot (established 2003 to 2005). Small trees with a DBH ranging from 2.5-9 cm were also identified at the species level and similarly measured as above within a 50 m<sup>2</sup> nested circular subplot (3.99 m radius; shared plot center with the 400 m<sup>2</sup> plot).

## Tree aboveground biomass and C computations

All trees with DBH >2.5 cm were measured or remeasured in 2018 in order to calculate the changes in aboveground biomass and C between initial (2003-2004) and final (2018) measurements. Changes between initial and final C stocks were standardized to decadal time scales for comparison purposes. In order to maintain consistency with the initial (2003-2004) calculations, aboveground biomass was estimated by summing the bark, stem, branch, and foliage biomasses calculated from DBH by species-specific allometric equations developed by Jenkins *et al.* (2003):

$$\text{Aboveground Biomass} = \exp(p_1 + p_2 \ln D), \quad (1);$$

where exp is the exponential function, D is the tree DBH (cm), ln is log base e, and  $p_1$  and  $p_2$  are species-specific allometric parameters.

Aboveground live-tree C for each tree was estimated by multiplying the aboveground biomass by the species-specific C contents in Lamtom and Savidge (2003):

Aboveground live-tree C =

$$\text{Aboveground Biomass} * cc \quad (2);$$

where cc is the C content (% of dry weight).

The equations parameters used to calculate Aboveground Biomass and C for trembling aspen, subalpine fir, paper birch, Douglas-fir, lodgepole pine, interior spruce, and willow, are given in Table 1.

Aboveground C for individual trees was calculated using equation (1) and (2). Changes in live-tree C at the plot-scale were calculated by subtracting the initial (2003 or 2004) plot-level C from that measured in 2018. Because initial plots were measured in either 2003 or 2004, changes in plot-level C were expressed on a decadal basis.

We analyzed the plot level change in aboveground C using linear regression models implemented in R (R Development Core Team 2018). We evaluated six predictor variables: initial C level, stem count of all trees with a DBH > 2.5 cm, median DBH, maximum DBH, Basal Area (BA) of conifers, and BA of deciduous. We applied Multimodel selection using the MuMIn R package (Bartoń, 2019). All potential models that could be generated using the six predictor variables, not including any interactions, were evaluated and ranked according to their Akaike Information Criterion (AIC) value.

Species code	Species	Aboveground biomass (kg)		Carbon content (%)
		$p_1$	$p_2$	
Bl	Subalpine fir	-2.5384	2.4814	50.08
Ep	Paper birch	-1.9123	2.3651	48.37
Sx	Interior spruce	-2.0773	2.3323	50.39
Pl	Lodgepole pine	-2.5356	2.4349	50.32
At	Trembling aspen	-2.22094	2.3867	47.09
Fd	Douglas-fir	-2.2304	2.4435	50.50
Ww	Willow spp.	-2.2094	2.3867	49.05

Table.1: Parameters used to calculate the aboveground biomass and carbon content for each species encountered at the ALRF (Jenkins *et al.* 2003; Lamtom and Savidge 2003).

## Results

We observed a positive linear relationship between absolute levels of aboveground live-tree C in 2018 versus initial (2003 or 2004) aboveground live-tree biomass (Fig. 2a;  $R^2=0.723$ ) and C (Fig. 2b;  $R^2=0.727$ ) at the plot-level. While higher initial live-tree C levels generally resulted in higher final C levels, the regression line suggests a potential loss of live-tree C with age, particularly in plots with higher initial live-tree C plots. However, the variance in live-tree C values was very large, with almost half of the plot values falling outside of the 95% confidence intervals.

Our model selection analysis indicated that the best model of the absolute change (normalized to decadal rates of change) in plot-level aboveground live-tree C was the one that only included initial C level as a predictor variable (Fig. 3). We found a negative linear relationship between the change in aboveground live-tree C and the initial C level ( $R^2=0.188$ ). Total decadal C change varied

greatly across the plots, with both the greatest decadal C gain ( $50 \text{ tonnes ha}^{-1} \text{ dec}^{-1}$ ) and loss ( $75 \text{ tonnes ha}^{-1} \text{ dec}^{-1}$ ) occurring in older stands. On average, plots with initial C stocks  $< 100 \text{ tonnes ha}^{-1}$  contributed positively to live-tree C storage, while plots with C stocks  $> 100 \text{ tonnes ha}^{-1}$  were more likely to lose C, with the gains and losses under the regression curve more-or-less offsetting. Similar trends were observed when plot-level data for single species were plotted against initial values (data not shown).

We wanted to understand the impact of species and tree size (DBH) on the change in aboveground tree-level C by DBH (Fig. 4a). By only examining trees living in both censuses, we avoided the impact of tree death on C change occurring between the two census dates. All of the three biomass dominant tree species in our ALRF plots had positive linear or exponential growth in aboveground C stocks with increasing DBH, albeit with high and increasing variation with increasing DBH. Paper birch (Ep) and subalpine fir (Bl) both

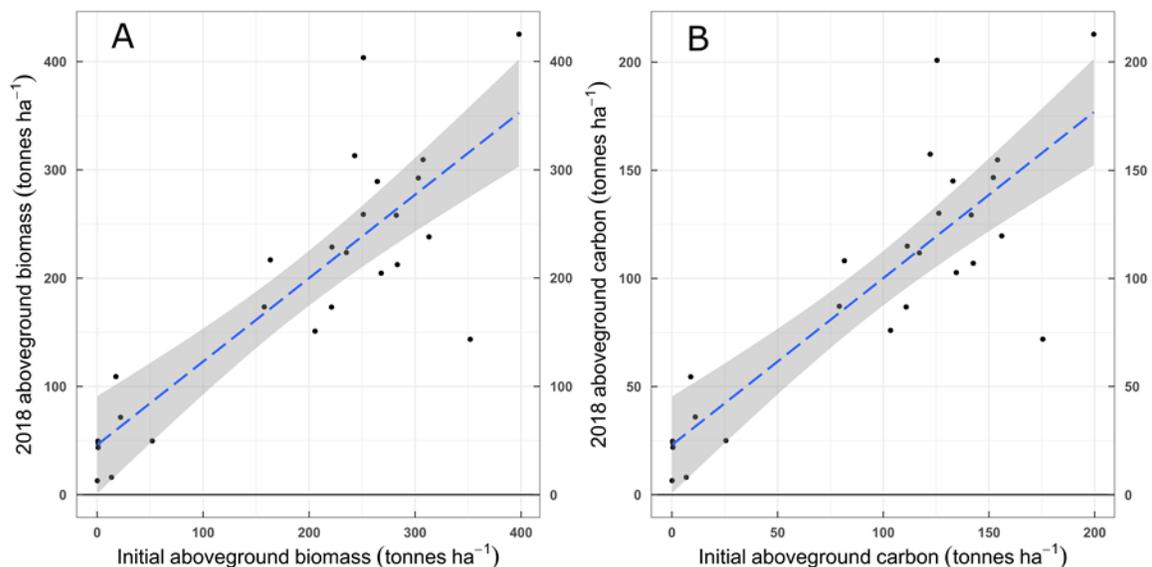


Fig. 2. Linear regression relationship between total initial (2003 or 2004) versus final (2018) aboveground live-tree (a) biomass and (b) carbon at the Aleza Lake Research Forest, BC. The grey area signifies 95% confidence intervals around the linear regression models:  $Aboveground\ biomass_{2018} = 0.77 * Initial\ aboveground\ biomass_{2003-2004} + 46$ ,  $R^2=0.723$ , and  $Aboveground\ carbon_{2018} = 0.771 * Initial\ aboveground\ carbon_{2003-2004} + 23$ ,  $R^2=0.727$ .

reached a maximum DBH of slightly more than 40cm, but grew faster than interior hybrid spruce (Sx) where DBH exceeded 60cm in some trees. Plots with trees that had lower maximum DBH (<40cm) primarily

experienced aboveground live-tree C accumulation, while plots with greater maximum DBH (>40cm) were more likely to experience declines in aboveground live-tree C accumulation (Fig. 4b).

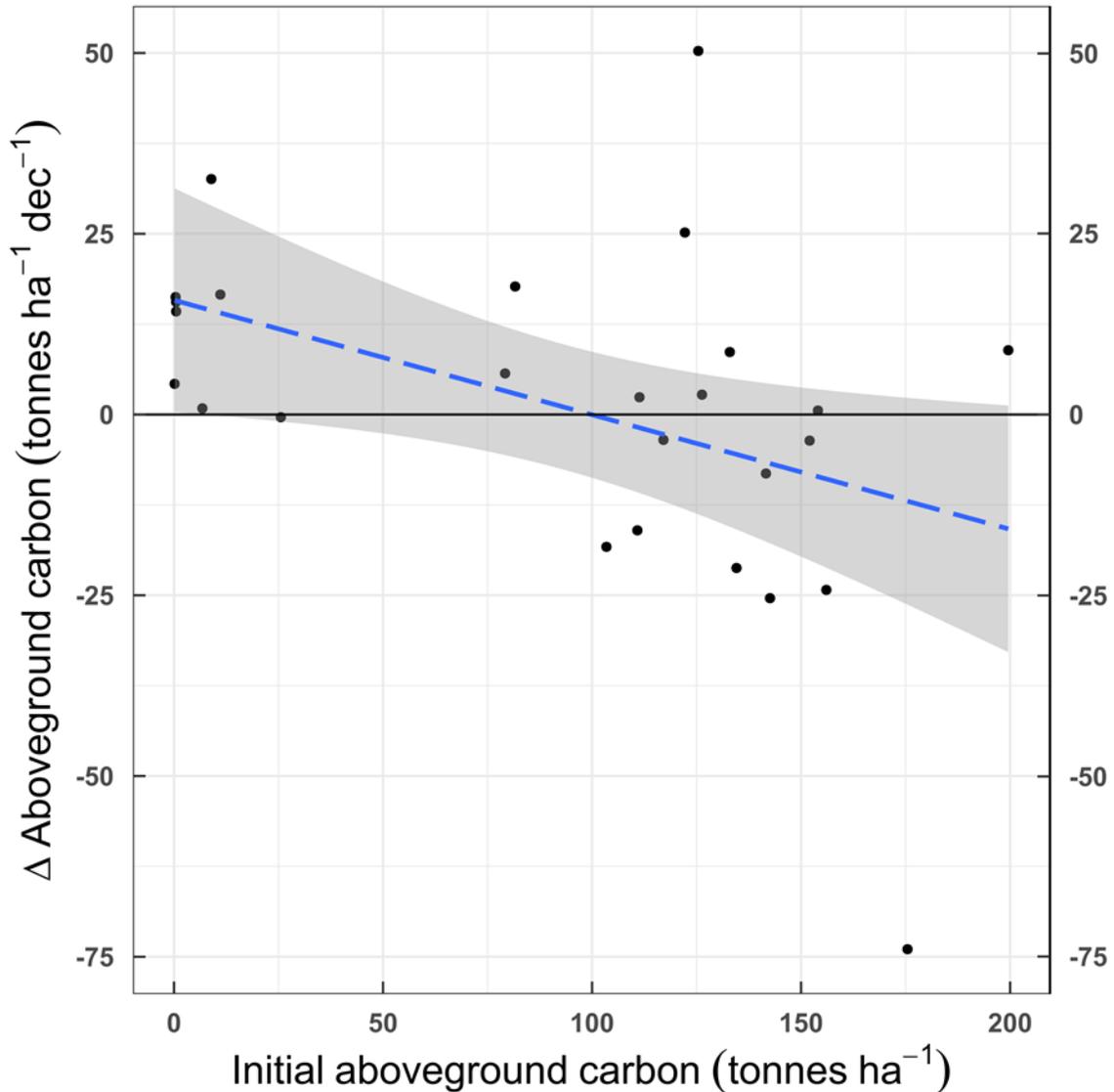


Fig. 3. Linear regression relationship between the decadal changes in total aboveground live-tree carbon versus total initial aboveground live-tree carbon. The grey area signifies the 95% confidence intervals around the linear regression line:  $\Delta \text{ Aboveground carbon} = -0.158 * \text{ Initial aboveground carbon} +$

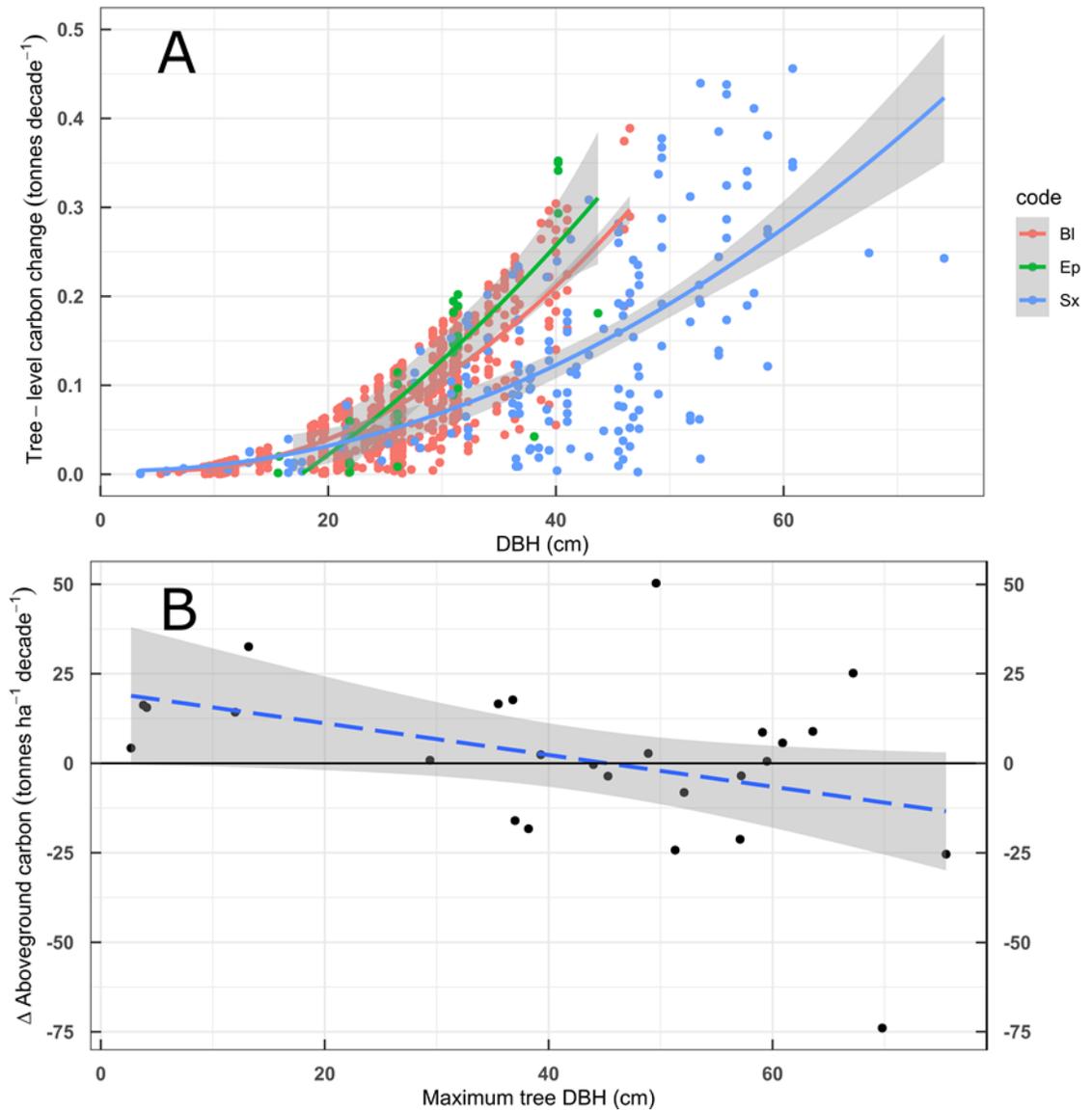


Fig. 4. Decadal changes in above ground live tree carbon (between 2003/04 and 2018) by final DBH: (A) in the three most common species ( $\text{kg C tree}^{-1} \text{dec}^{-1}$ ): subalpine fir (BI; red), paper birch (Ep; green) and hybrid spruce (Sx; blue) by DBH (cm), and (B) at the plot level ( $n=26$ ) ( $\text{tonnes C ha}^{-1} \text{dec}^{-1}$ ) by maximum tree DBH (in 2018) in each plot, at the UNBC Aleza Lake Research Forest. For 4B, the grey regions signify the 95% confidence intervals around the linear regression model line:  $\Delta \text{ Aboveground carbon} = -0.443 * \text{Maximum tree DBH} + 20$ ,  $R^2=0.162$ .

## Discussion

Older forests, often in mixed-aged stands, typically contain the greatest stocks of C (Lutz et al. 2012, Slik et al. 2013). This is also the case for sub-boreal wet forests of the Aleza Lake Research Forest (ALRF) study area where the present study was conducted (Fredeen et al. 2005; Bois et al. 2009). However, missing from one-time snapshots of forest C stocks (e.g. live tree C stocks presented here) is how they might be changing over time. In our remeasurement of total forest C plots 14 to 15 years after first measurement, we focused on aboveground live-tree C for several reasons: 1) they represent the single largest C stock (>50% of all total live and dead stocks in mature and older stands) in sub-boreal forests at the ALRF (Fredeen et al. 2005), and 2) they are the portion of the tree directly involved in sequestration of new C. There is broad consensus that younger to mature forests have the higher rates of C storage (Pugh *et al.* 2019), while old forest stands have reduced rates of C storage (Gao *et al.* 2018). Never-the-less, but not surprisingly, higher biomass and C stocks (in 2018) at the ALRF were generally observed in plots that had the higher initial stocks, i.e. in 2003 or 2004 (Fig. 2), albeit with much higher variance (roughly equal numbers of increases and decreases in C stocks) in older stands. At the same time, a best-fit line for remeasured stocks over initial stocks was suggestive of higher and positive C sequestration rates in live-tree C in younger stands than in older forest stands (Fig. 2), consistent with the findings of others (e.g. Gao *et al.* 2018, Pugh *et al.* 2019),

Greater volatility of live-tree C stocks were observed in older forest plots where large tree mortality events occurred. While forest stands typically undergo ‘self thinning’ (Drew and Flewelling 1977, Lonsdale 1990) during their development, younger stands are made up of more individual trees, each with a relatively small proportion of total forest stand C. As stands age, the asymmetry of biomass and C between large and small trees is accentuated (e.g. Fredeen et al. 2005), with each large tree containing an ever greater proportion of total forest stand C. A reexamination of C stocks

contained in old SBS forest plots (age-class 7/8 or 8) at the ALRF in 2004 (see Bois et al. 2009) supports this assertion, demonstrating that the single largest tree in each of these plots represented  $25.6 \pm 7.3\%$  of total live-tree plot C (Fredeen, data not shown). As a result, large/old tree mortality events in a  $400 \text{ m}^2$  plot can result in the loss of ~ one-quarter of plot-level live-tree C (*also see* Lutz et al. 2012).

While old tree mortality is expected and inevitable, there is concern that rates of mortality of old/large trees may be increasing. A recent study in the Western U.S. and south coast of British Columbia examined old and undisturbed stands (>200y old stands) in seventy-six long-term forest plots (Mantgem et al. 2009). After controlling for normal (noncatastrophic) mortality, they found substantial and consistent increases in recent rates of mortality for large as well as small trees, with a doubling rate of 17 to 29 years. These results are mostly mirrored by those of Peng et al. (2011) who examined mortality rates of ‘Permanent Sampling Plot’ trees (including white spruce) across Canada from 1963-2008. Their results suggested that tree mortality rates in western Canada increased by an average of 4.7% per year between 1963 and 2008, much of this strongly correlated with the rise in climate-driven drought. Spruce and pine have been shown to exhibit the highest mortality rates in mature stands in central Europe, as well as the highest sensitivity to factors such as climate change (Pajtik *et al.* 2018). Together, these results suggest that increased mortality of trees in western Canada are a distinct possibility, and while not yet fully explained, may already be impacting forest C storage, and the balance between live and dead tree carbon in SBS forests.

The substantial losses of live-tree C indicated for older ‘C-rich’ plots in our study need to be put in perspective and do not inherently mean that live-tree or total forest C stocks were reduced or forest C emissions increased in the ALRF. Because our sample size (26 plots) and plot areal coverage (<0.1% of the study area) were small, extrapolation must be treated with

great caution, especially with the large variances observed and low significance values of the linear regression relationship (see Fig. 3). Furthermore, we only considered live-tree C in this study. Standing dead trees often remain for decades and may end up leaning and off the ground for additional significant periods of time before eventually contributing to long-lived dead tree C stocks on the ground. The initial forest C stocks at the ALRF (from 2003 and 2004) confirms this, showing that total forest C stocks, inclusive of live as well as dead trees aboveground and belowground, increase with forest age (Fredeen *et al.* 2005, Bois *et al.* 2009). Finally, a time series of Landsat TM and Landsat ETM+ from 1992 to 2003 and 2004 C stock data indicated that the oldest SBS stands at the ALRF had the highest C stocks of all stand ages, which were essentially stable over the previous decade (Janzen *et al.* 2010). Similar conclusions have been reached by others working in wet temperate conifer systems in North America (e.g. Harmon *et al.* 1990). Even more surprising, Luysaert *et al.* (2008) in a global assessment of old-growth forests, found that old forests remain sinks for carbon for their life-times, albeit at reduced levels, contrary to the conventional wisdom of the last century (e.g. Bormann and Likens 1979, Oliver 1981).

### **Impact of forest species composition on C dynamics**

Conifer trees, primarily interior spruce and subalpine fir, are the dominant tree species in the ALRF. Therefore it was not surprising that they contributed most to live-tree C stocks (often exceeding 100 tonnes in aboveground live-tree conifer C ha<sup>-1</sup>; Fig. 2) as well as to aboveground C change (both positive and negative) in the intervening 14 to 15 years between C stock assessments. Across our 26 plots, initial live-tree C in deciduous trees was on average approximately one-third of the initial C contained in conifer species and lost that C more rapidly relative to conifers (Gan 2019). These results are consistent with the normal successional pathway in sub-boreal mixedwood forests, where faster growing species such as trembling aspen are eventually

out-competed by longer-lived, shade-tolerant coniferous trees, even though canopy gaps are frequently occupied by young shade-intolerant trees such as birch, trembling aspen, pine, and willow (Chen and Popadiouk 2002). Negative relationships were found between initial aboveground live-tree deciduous and conifer C versus total changes in aboveground C relationships, but low R<sup>2</sup> values in both cases speak to the high level of variability between plots (Fig. 4). This variability is understandable given the successional, forest management and disturbance histories across the research area (see Fredeen *et al.* 2005 and Bois *et al.* 2009).

Forest management, natural disturbances, and climate change can all be important drivers impacting forest composition in the decades ahead (Chen and Popadiouk 2002). In sub-boreal mixedwoods, multiple succession pathways can shape forest composition and structure depending on the type and variety of disturbances, site characteristics, biotic components, and their interaction. For example, insect outbreaks can accelerate or set back the stages of forest succession, depending on the type and extent of insect disturbance (Chen *et al.* 2002). Also, stand-replacing fires in boreal forests can greatly increase the abundance of deciduous species such as aspen, greatly increasing their contribution to C storage in mixed-species stands (Gao *et al.* 2018; Laganière *et al.* 2015). The age-class structures of boreal forests are dependent on biotic disturbance rates, and is modified by forest management activities such as fire suppression, harvesting and afforestation (*et al.* 1995). While some climate-induced increases in forest productivity are possible in the future, these increases are not expected to compensate for losses in biomass due to increased rates of natural disturbance in Canadian boreal forests (Kurz *et al.* 2008). Suitable management approaches may be helpful in shifting forest compositions and age structures towards less disturbance susceptible stands, but a shift to younger managed forests will store much less C than the large, old trees and forests they replace. If the interval of natural disturbances

that influenced the average age of old-growth forests are longer than the age of harvest rotation, the C benefit of old-growth forests can be large (Price *et al.* 1996). Alternatively, other studies suggest that harvesting combined with silviculture could increase C storage over unmanaged stands affected by an increased frequency of climate-related natural disturbance rates (Harmon *et al.* 1990; Price *et al.* 1996; Vitousek 1991). Additional field work and modeling at regional scales are needed to assist future forest managers to conserve or enhance future forest C storage.

## Conclusions

A large proportion of live-tree C stocks at the stand level are typically found in older, large diameter Interior spruce trees at the ALRF. While our data were limited by plot number (n=26), our results suggest that decadal changes in live-tree C at the plot or stand level are highly sensitive to mortality events in large DBH tree plots. By contrast, younger stands (i.e. smaller DBH) more consistently exhibited

gains in live-tree C. Never-the-less, large tree mortality events, while important for live-tree C stocks, would not necessarily boost C emissions because of large dead tree C persistence and compensatory C uptake by subordinate trees and ground cover.

Future studies are needed to better identify the factors and interactions affecting sub-boreal forest C dynamics from early to late succession. This study was based on the remeasurement of only 26 plots (400m<sup>2</sup> plot<sup>-1</sup>). The remaining 2003 and 2004 legacy plots could be remeasured in future years to expand the sample size over longer intervals of time. Finally, more repeated LiDAR assessments of C stocks at the ALRF and other sub-boreal forests could broaden our understanding of the amounts and vulnerability of live aboveground forest C stocks across the region.

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