NRC Research Press

ARTICLE

Recovery of carbon pools a decade after wildfire in black spruce forests of interior Alaska: effects of soil texture and landscape position

Gregory P. Houle, Evan S. Kane, Eric S. Kasischke, Carolyn M. Gibson, and Merritt R. Turetsky

Abstract: We measured organic-layer (OL) recovery and carbon stocks in dead woody debris a decade after wildfire in black spruce (*Picea mariana* (Mill.) B.S.P.) forests of interior Alaska. Previous study at these research plots has shown the strong role that landscape position plays in governing the proportion of OL consumed during fire and revegetation after fire. Here, we show that landscape position likely influences fire dynamics in these stands through changes in mineral soil texture. The content of fine-textured materials in underlying mineral soils was positively related to OL depths measured 1 and 10 years after fire, and there was an interaction between soil texture and elevation in governing OL consumption and OL recovery a decade following fire. OL depths 10 years after fire were 2 cm greater than 1 year after fire, with a range of 19 cm of accumulation to 9 cm of subsidence. Subsidence was inversely related to the percentage of fine textures within the parent material. The most influential factor determining the accumulation of OL carbon stocks a decade following wildfire was the interaction between landscape position and the presence of fine-textured soil. As such, parent material texture interacted with biological processes to govern the recovery of soil organic layers.

Key words: Picea mariana, parent material, clay, loess, wild fire.

Résumé : Nous avons mesuré le rétablissement de la couche organique (CO) et les réserves de carbone dans les débris ligneux 10 ans après un feu d'origine naturelle survenu dans des forêts d'épinette noire (*Picea mariana* (Mill.) B.S.P.) de l'intérieur de l'Alaska. Une étude antérieure réalisée dans ces placettes de recherche a montré que la position dans le paysage a un effet déterminant sur la proportion de la CO consumée pendant le feu et la revégétalisation après le feu. Dans la présente étude, nous montrons que la position dans le paysage influence probablement la dynamique du feu dans ces peuplements via des changements de texture du sol minéral. Le contenu en matériaux de texture fine dans le sol minéral sous-jacent a été positivement relié à la profondeur de la CO mesurée 1 et 10 ans après le feu, et il y avait une interaction entre la texture du sol et l'altitude concernant la consumation et le rétablissement de la CO 10 ans après le feu. Dix ans après le feu, la CO était 2 cm plus profonde qu'un an après le feu, avec une étendue des observations de 19 cm d'accumulation à 9 cm de subsidence. La subsidence était inversement reliée au pourcentage de matériaux de texture fine dans le matériau parental. Le facteur qui influençait le plus l'accumulation des réserves de carbone dans la couche organique 10 ans après le feu était l'interaction entre la position dans le paysage et la présence de sol à texture fine. Ainsi, la texture du matériau parental interagit avec les processus biologiques pour déterminer le rétablissement des couches organiques du sol. [Traduit par la Rédaction]

Mots-clés : Picea mariana, matériau parental, argile, lœss, feu d'origine naturelle.

Introduction

Boreal forests currently harbor about a third of the Earth's rapidly cycling soil organic carbon (C) (McGuire et al. 1995; Pan et al. 2013), which is a pool size similar in magnitude to that of all of the C in the atmosphere (~800 Pg). Carbon has accumulated in boreal forest soils as the result of C inputs from net primary production slightly exceeding C losses from decomposition and disturbances (erosion, combustion during fire). This imbalance has been maintained through millennia owing to cold and wet conditions, which slow decomposition processes and also contribute to these stores being relatively resilient to disturbances such as wildfire. In the North American boreal forest, black spruce (*Picea mariana* Mill. B.S.P.) cover types are associated with the largest soil organic carbon (SOC) stocks and are also prone to fire (Johnson and Kern 2003). As part of the historic fire regime in this cover type, C losses through combustion recover with succession and regeneration; this pattern of disturbance and regeneration is assumed to follow a sawtooth pattern wherein rapid C losses in wildfire are followed by gradual accrual of SOC with stand replacement (Harden et al. 2000). However, this paradigm may be affected by a changing climate and fire regime in the North American boreal forest.

In recent decades, greater burned area and increasing fire sizes have occurred, particularly in continental regions of Alaska and western Canada (Kasischke and Turetsky 2006). The increased extent and severity of wildfire has implications for C storage in these ecosystems, with many black spruce forests losing more SOC in these large fire events than they have historically been able to accumulate between fire events (Turetsky et al. 2011). As such, it is

Received 16 June 2017. Accepted 12 October 2017.

G.P. Houle and E.S. Kane. School of Forest Resources and Environmental Science, Michigan Technological University, Houghton, MI 49931, USA; USDA Forest Service, Northern Research Station, Houghton, MI 49931-1199, USA.
E.S. Kasischke. Department of Geography, University of Maryland, College Park, MD 20742, USA.
C.M. Gibson and M.R. Turetsky. Department of Integrative Biology, University of Guelph, Guelph, ON N1G 2W1, Canada.
Corresponding author: Evan S. Kane (email: eskane@mtu.edu).

Published at www.nrcresearchpress.com/cjfr on 16 October 2017.

important to investigate the controls on SOC accumulation following severe wildfires in boreal black spruce forests to understand how wildfire may affect C storage capabilities of these ecosystems.

There is an emerging understanding of controls on organic matter consumption in wildfires in boreal black spruce forests (Johnstone et al. 2010; Kasischke et al. 2010; Kasischke and Hoy 2012), but few studies have directly investigated the variable controls on organic-layer (OL) recovery and soil development over time following boreal wildfires (cf. Viereck and Dyrness 1979; Seedre et al. 2014; Kishchuk et al. 2015). Conceptually, our understanding of the development of soils following disturbance, including SOC dynamics, comes from the "state factor concept" proposed by Dokuchaev (Dokuchaev 1879; Hambidge 1938) and Jenny (Amundson and Jenny 1997). State factors include the interacting effects of physiography, climate, time, biology, and parent material that influence soil development. In boreal forests, relief and aspect strongly control microclimate (Slaughter and Viereck 1986), with north-facing and toe-slope forests generally being cooler, wetter, and more likely to be dominated by black spruce and contain permafrost than south-facing or flat upland stands on hill crests. In turn, north-facing and flat lowland forests generally have less OL consumption in any fire event (Turetsky et al. 2011), as well as slower productivity and slower rates of decomposition during fire-free periods. As a result of all of these processes, OL depths and SOC stocks tend to be larger in cooler north-facing and toe-slope forests on time scales appropriate for factoring disturbance effects on the C cycle (Van Cleve and Yarie 1986; Kane and Vogel 2009). Fires generally occur on 100-year intervals in interior Alaska (Yarie 1981), but when the fire return interval is shortened, the "time" component of the state factor concept gets reset, potentially decreasing OL depth recovery and SOC accumulation after fire (Bond-Lamberty et al. 2007; Brown and Johnstone 2011; Harden et al. 2012; Hoy et al. 2016). It can therefore be seen how time interacts with other state factors mainly related to site drainage in controlling the recovery of OL and SOC following wildfire. While there is general consensus of state factor control on soil development in the broad sense, a key state factor control on post-fire OL characteristics in boreal forests — parent material texture — has received far less consideration (Van Cleve et al. 1991; Johnson et al. 2011).

Loess deposits (clay and silt-sized particles) are widespread parent materials throughout Alaska's interior region that develop syngenetically from wind-derived inputs from the major glacial river valleys, namely the Tanana, Yukon, and Koyukuk (Péwé 1955; Beget et al. 2006). As such, the thickness and development of loess deposits generally decrease gradually with distance from alluvial sources (White et al. 2002; Mulligan 2005) and are thinner at higher elevations, particularly near the treeline where there is less tall-statured vegetation to trap eolian deposits (Péwé and Reger 1983; Muhs et al. 2003; Beget et al. 2006). This discontinuous mantle of loess overlies residual, colluvial, and glacially transported materials, which generally have lower water holding capacities due to being coarser textured and containing rock fragments (Ping et al. (2006) and references therein). Changes in loess depth and parent material characteristics also influence soil ice content and permafrost maintenance, which also greatly affect drainage (Swanson 1996; Brown et al. 2015). As such, much of the soil water holding capacity in interior Alaska is derived from variation in the thickness of the loess cap and what it is underlain with, both of which vary with physiography and origin (see, for example, O'Donnell et al. 2013).

While it is well established that soil texture exerts strong control over soil moisture, stand production, and organic matter accumulation in boreal forests (e.g., Bhatti and Apps 2000), its interactions with landscape position in governing the severity of burning and the recovery of SOC stocks following fires have not been directly examined. Here, we investigate changes in OL consumption and recovery on different landscape positions in the additional context of the underlying parent material. First, we examined the interaction between soil texture and landscape position in determining the depth of burn (1 year after fire). Second, we looked at the role of soil texture in influencing recovery of the SOC pool during a 10-year window of post-fire succession. We measured OL depths, bulk densities (BD), SOC stocks, and standing and down woody debris C pools prior to burning (live trees and undisturbed SOC stocks), 1 year after wildfires occurred in 2004, and 10 years after wildfires in interior Alaska. We hypothesized that there would be OL recovery over the 10-year window following fire, with new materials accumulated since fire having a higher C concentration and lower BD than the underlying OL remaining after a fire. On the other hand, there may be changes in the residual OL remaining after fire due to increased decomposition, site drainage, and physical compaction, which in turn contribute to increases in BD, perhaps lower C concentration, and subsidence of the OL. Our objective was to determine the extent to which parent material texture, as a state factor control over site drainage, could explain variance additional to physiographic variables in OL consumption in wildfire, as well as OL recovery, SOC accumulation, and the relative distributions of post-fire C pools associated with woody debris.

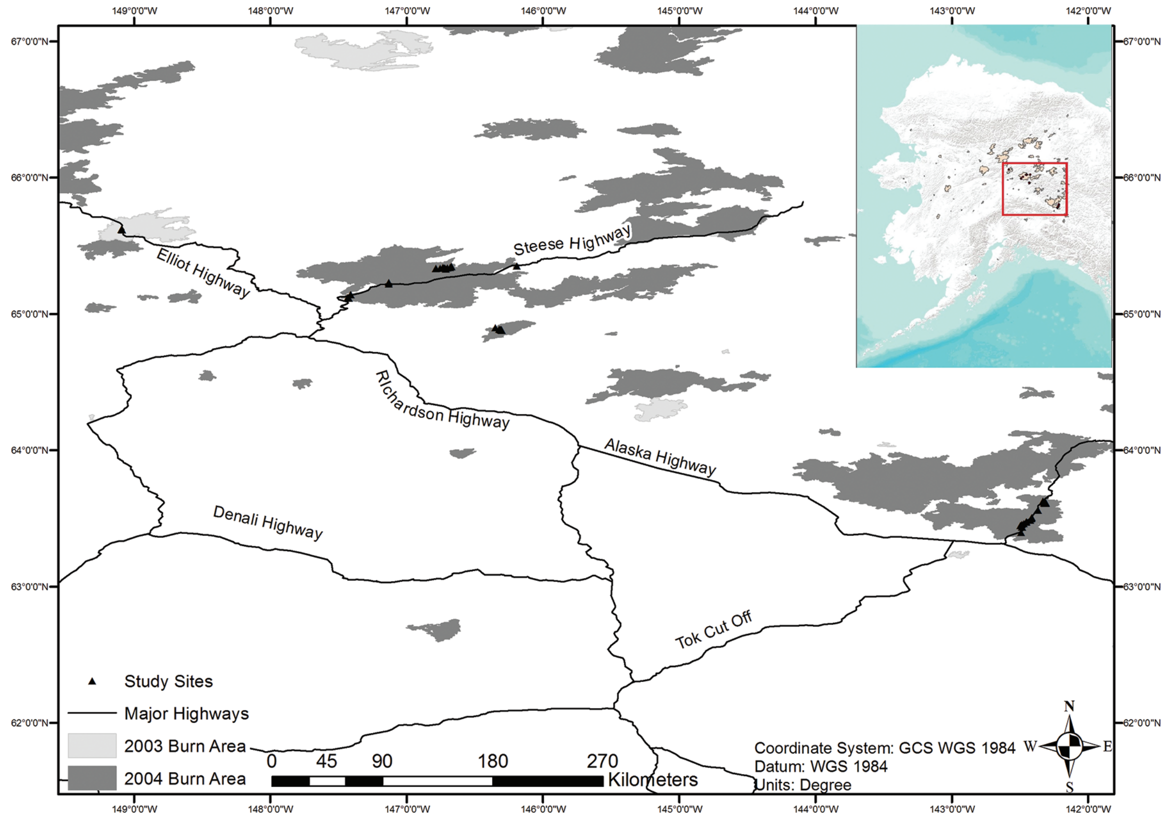
Methods

Site selection and sampling

During the summer of 2004, interior Alaska experienced the hottest and driest weather since 1940. These conditions resulted in widespread fires consuming 2.7×10^6 ha of forests, representing the largest annual burned area in Alaska's 58-year fire record (Todd and Jewkes 2006). Most study sites were established in the following summer (June to August 2005), with some sites established in the summer of 2006, as described in detail by Turetsky et al. (2011). While the sites selected for this study burned during early summer (June and early July) fires, they encompassed a range of fire severities (defined by consumption of aboveground biomass and organic layer) that were controlled by site moisture levels, topographic positions, parent material texture, and geographic dispersion across the landscape (Kasischke et al. 2008). Briefly, site establishment included placing a center wooden post monument (where GPS location was determined) in the middle of a 50 m baseline transect that bisected one 30 m transect at the center and two additional 30 m transects at random distances from the center. Soil OL depths, soil BDs, and samples for elemental analysis were taken as previously described in detail (Kane et al. 2007; Turetsky et al. 2011). Depth of burn was quantified by measuring the distance from the uppermost adventitious root down to the top of the organic soil layer as previously described (Kasischke et al. 2008; Boby et al. 2010).

In this study, 38 study sites encompassing four different landscape positions (flat (<5% slope) lowland, $n = 8$; flat upland, $n = 10$; northern aspect, $n = 12$; and southern aspect, $n = 8$) were revisited for sampling 10 years after fire (Fig. 1). The original wooden monument was located and transects were run as previously described. Organic-layer depths were measured every 5 m along the 50 m baseline and three 30 m transects. Organic-layer soil cores were sampled from each site at 0, 15, and 30 m along the three 30 m transects ($n = 9$). Core sampling occurred after biomass collection using a 5 cm diameter sharpened stainless-steel soil corer attached to a power drill (cf. Nalder and Wein 1998). Cores were sampled in 20 cm increments, with the surface char layer parsed from the remaining core samples in the field. Incremental depths were noted and samples were refrigerated until lab analysis. In this study, only OLs were re-measured for changes in physical and chemical properties. We dug an additional soil pit to a depth of 0.5 m into the underlying mineral soil for texture analysis of site parent material at each site.

Fig. 1. Map of 38 sites burned in 2003 and 2004 revisited for measurements in 2014. Inset depicts study region within the interior region of Alaska, USA. [Colour online.]



Dead woody debris

Down dead woody debris (DDWD) was sampled using the Brown (1974) transect intercept method. Diameters of down dead black spruce were recorded at the intersection of the three 30 m transects, measured using a hand caliper, and classified by five different size classes: class I, 0–0.5 cm; class II, 0.5–1 cm; class III, 1–3 cm; class IV, 3–5 cm; and class V, 5+ cm (Nalder et al. 1997). Five decay classes were also assigned with diameter measurements following Manies et al. (2005). Down dead black spruce biomass was quantified using the fuel load estimate equation by sample size classes I–V (Nalder et al. 1997). Each of the five designated decay classes was used in determining the biomass of woody tissues (Manies et al. 2005), which was assumed to be 50% carbon (Bond-Lamberty et al. 2004; Thomas and Martin 2012).

All standing dead black spruce within 1 m of either side of the 30 m transects (180 m^2) were tallied and their basal diameters were measured. Standing dead black spruce tree biomass was determined using allometric equations for bole, bark, and coarse branches based on basal diameters (Yarie et al. 2007). All standing dead woody debris (SDWD) was assigned a decay class of 1, and biomass was assumed to be 50% carbon (Thomas and Martin 2012).

Soil core processing

Field moist OL soil cores were weighed and cut in half vertically by 20 cm sections. One-half was designated for soil BD, which was determined as the dry core mass (dried to constant mass at 65 °C) divided by the incremental core volume. Dry BD core samples were ground in a Wiley mill for elemental analysis.

Mineral soil samples were prepared by drying in an oven at 65 °C; they were then pulverized with a hammer and sieved through a no. 10 (2 mm) mesh sieve to determine rock content. Particle size analysis for percentages of sand, silt, and clay on the rock-free fraction followed the standard hydrometer method (Gee and Bauder 1986). Briefly, 100 g of mineral soil sample was dispersed with 50 mL of sodium hexametaphosphate. Hydrometer and solution temperature were recorded at 40 s and 8 h after the initial mixing of the solution. A blank solution was made, and hydrometer and temperature measurements were recorded during the same measurement intervals as the parent material solutions.

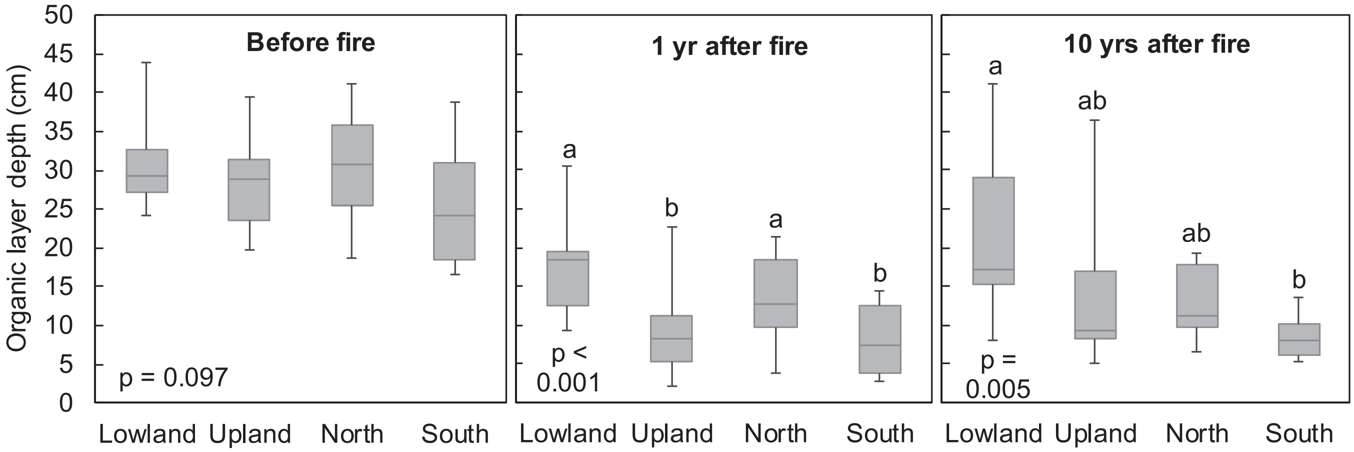
Table 1. Mixed-effects models describing percent changes in organic layer depths from before fire to (*a*) 1 year and (*b*) 10 years after fire.

	(*a*) Percent change 1 year after fire		(*b*) Percent change 10 years after fire	
Effect	*F* value	*p*	*F* value	*p*
Slope	1.16	0.2910	30.88	<.0001**
Elevation	1.93	0.1756	10.11	0.0036**
Percent clay	10.98	0.0025**	50.28	<.0001**
Aspect	0.77	0.3869	1.92	0.1764
Aspect × clay	0.04	0.8403	2.66	0.1139
Slope × clay	2.50	0.1251	7.49	0.0107**
Elevation × clay	6.29	0.0182**	20.63	<.0001**
Slope × elevation	1.37	0.2522	21.68	<.0001**
Slope × elevation × clay	2.15	0.1534	6.59	0.0159*

Note: Model n = 38. Asterisks denote level of significance: *, $p \leq 0.05$; **, $p < 0.01$.

All soil samples from 16 sites (four from each landscape classification unit) were analyzed for elemental C and N content (Costech Elemental Combustion System 4010; Costech Analytical Technologies Inc., Valencia, California, USA). Organic matter content based on loss on ignition (LOI) at 500 °C for 12 h in a muffle furnace was determined on soil samples from all 38 sites, and carbon content was determined from the empirical relationship between LOI and elemental analysis ($R^2 = 0.77$, $p < 0.001$, $F = 33.54$; $\beta_0 = 0.33 \pm 5.07$; $\beta_1 = 0.49 \pm 0.08$). A separate empirical relationship between LOI and elemental analysis was used to determine the C

Fig. 2. Mean organic layer depths before wildfires (adventitious root method) and 1 and 10 years after wildfires (direct measurements) on different landscape positions. Different letters denote significant differences across landscape positions (post hoc comparisons of means by landscape position).



content of char-layer soil samples ($R^2 = 0.73$, $p = 0.002$, $F = 21.14$; $\beta_0 = 2.60 \pm 8.25$; $\beta_1 = 0.48 \pm 0.11$).

Statistical analyses

Effects of landscape position measurements (slope, aspect, and elevation), soil texture, and their interactions on the percent change in OL depth, absolute change in OL depth, change in BD, and change in SOC stocks 10 years after wildfires were investigated. We used a general linear mixed model approach, which enables statistical models to be fit to data where the response is not necessarily normally distributed (PROC GLIMMIX; SAS version 9.4, SAS Institute, Cary, North Carolina, USA). The distributions of the response variables were evaluated in Kolmogorov–Smirnov tests using the UNIVARIATE procedure. The appropriate data distributions satisfying assumptions of normality were assigned in the mixed-effects models (with distribution (dist) assigned as normal, lognormal, exponential, or gamma as appropriate; link = log function). Type-3 tests of fixed effects and post-hoc comparisons of least-squared means tests across landscape positions were considered significant at $\alpha = 0.05$. Least-squared mean value comparisons employed the Tukey–Kramer adjustment. SAS code and all organic-layer, LOI, carbon, and BD data are available in the public domain (https://doi.pangaea.de/10.1594/PANGAEA.880718).

Results

Controls on depth of burn

Pre-fire organic-layer depths ranged from 16.6 to 43.8 cm and averaged (± standard error) 28.7 ± 0.2 cm. Organic-layer depths 1 year after fire ranged from 1.7 to 30.6 cm and averaged 12.0 ± 0.2 cm. For 1-year post-fire measurements, lowland OL depths were thicker than uplands by a factor of 2.1 and north-facing slopes were thicker than south-facing slopes by a factor of 1.7. Factors contributing to the relative change in OL depth (percent change in OL depth after burning) in the years following wildfire included mineral soil texture and its interaction with elevation (Table 1). While 1-year post-fire measurements of OL depths did vary significantly with landscape position (Fig. 2), model results suggest that the change in OL depths is driven mainly by the presence of fine-textured materials within the parent material (Table 1*a*).

Controls on residual OL depths

Organic-layer depths varied with landscape position at both 1 year and 10 years following wildfires (Fig. 2). Across all plots, mean OL depths did not recover to pre-fire depths in 10 years. Ten-year OL depth measurements ranged from 5.2 to 41.0 cm and averaged 13.6 ± 0.2 cm. Ten-year post-fire OL depths differed by a factor of 1.6 between lowlands and uplands and by a factor of 1.5 between north- and south-facing slopes, though variability on north-facing slopes was high and differences were not significant between these two landscape positions (Fig. 2). While the majority of the sites re-sampled 10 years following wildfire exhibited thicker OL depths than those measured 1 year after fire, 16 of 38 sites actually had shallower mean OL depths (Fig. 3*a*). These declines in OL thickness are likely attributed to subsidence, which coincided with an increase in total OL BD across almost every site (Fig. 3*b*). Across all sites, south-facing and upland sites generally had greater increases in OL depths (0.30 ± 0.13 cm·year^{-1}) than did north-facing and lowland sites (0.10 ± 0.11 cm·year^{-1}) over the last 10 years. Relationships between 10-year post-fire OL depths and elevation indicated that as site elevation increased, OL thickness following fire decreased ($R^2 = 0.31$, $p < 0.001$; Table 2).

Factors contributing to the measured change in OL depth over 10 years included slope, mineral soil texture, and their interactions with elevation (Table 2*a*). There was increased accumulation of OL depth over the last 10 years as the clay content of underlying mineral soils increased, and there was an interaction between slope and clay content in explaining variation in OL depths (Table 1*b*).

Controls on post-fire SOC stocks

Factors contributing to changes in OL depth and soil C stocks were related to changes in physiographic variables and parent material texture, with slope and elevation playing a significant role in OL depths and aspect and texture playing a larger role with changes in C stocks (Table 2). In particular, the clay content of underlying mineral soil significantly affected the amount of change in OL depth between measurements taken 1 and 10 years following fire, with an increase in finer textured parent material resulting in a greater accumulation of organic material in the first decade following fire (Fig. 4). In fact, OL subsidence (negative change in OL depth over the last 10 years) occurred when the percentage of clay in the parent material was <20% (Fig. 4*c*). On average, BD measurements were higher in year 10 than in year 1 across all plots by 0.04 ± 0.01 g·cm^{-3}, which is consistent with increasing subsidence in the organic layers (Fig. 3). BD measured 10 years after fire and the change in BD measurements between 1 and 10 years after fire were both lower in sites with a higher percentage of clay in the mineral soil (Figs. 4*a* and 4*b*). However, BD values varied widely across all plots (mean coefficient of variation = 1.1), and there were no significant changes in BD with landscape position (Table 2*b*).

Fig. 3. Comparisons of (*a*) post-fire organic layer (OL) depths, (*b*) organic layer bulk densities (BD), and (*c*) soil organic carbon (SOC) stocks 1 year following wildfire (*x* axes) vs. 10 years following wildfire (*y* axes). The dashed line in each plot is *y* = *x*.

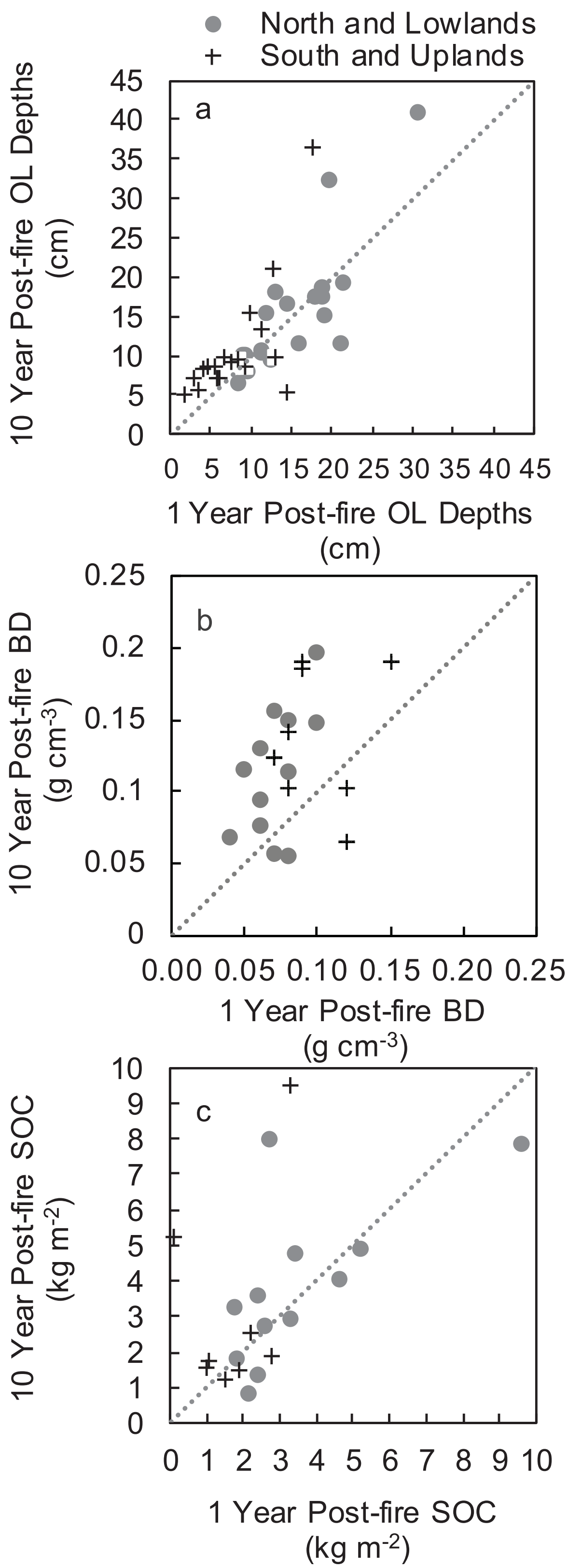


Table 2. Mixed-effects models describing changes in organic layer (OL) (*a*) depths, (*b*) bulk densities, and (*c*) carbon (C) stocks from 1 year after fire to 10 years after fire.

	(*a*) Change in OL depth		(*b*) Change in OL bulk density		(*c*) Change in OL C stock	
Effect	*F* value	*p*	*F* value	*p*	*F* value	*p*
Slope	11.27	0.0023**	1.19	0.3016	1.96	0.1916
Elevation	1.40	0.2466	2.39	0.1530	0.28	0.6060
Percent clay	10.07	0.0036**	0.00	0.9505	0.42	0.5337
Aspect	1.48	0.2333	0.03	0.8566	9.10	0.0130*
Aspect × clay	0.09	0.7647	1.96	0.1918	22.95	0.0007**
Slope × clay	1.81	0.1894	0.07	0.7978	0.03	0.8742
Elevation × clay	4.42	0.0447*	0.00	0.9992	3.43	0.0936
Slope × elevation	8.07	0.0083**	2.93	0.1179	2.99	0.1144
Slope × elevation × clay	1.65	0.2091	0.00	0.9662	0.68	0.4292

Note: Model $n = 38$ for organic layers and $n = 20$ for bulk density and C stocks. Asterisks denote level of significance: *, $p \leq 0.05$; **, $p < 0.01$.

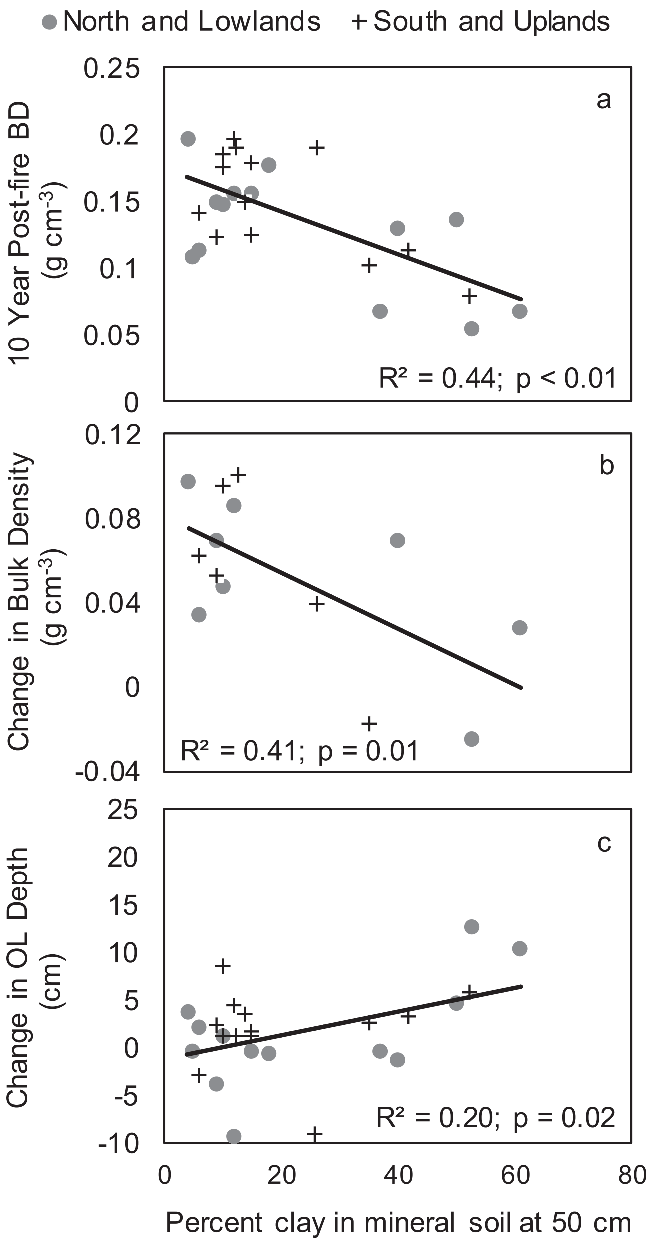
Because SOC stocks are the product of C concentration, BD, and depth considered, interpretation of changes in SOC stocks is best done in the context of changes in BD and OL depth. Most sites in this study accumulated OL SOC stocks in the first decade following fire, suggesting that either C concentration or change in BD were more significant than any effects of OL subsidence (decreasing thickness or depth) in determining changes in SOC stocks (Fig. 3). In considering all physiographic variables together, only aspect had a significant main effect on changes in OL SOC stocks (Table 2*c*). However, because changes in OL depths occurred with texture and elevation, we examined these effects on changes in SOC stocks. Percent clay within the parent material was not considered a significant main effect explaining changes in carbon stocks between 1 and 10 years following fire, but there was a significant interaction between aspect and the clay percentage in explaining changes in SOC stocks (Table 2*c*). While the north-facing and lowland sites generally coincided with having higher mineral soil clay content, these sites accrued SOC more slowly (0.07 ± 0.05 kg C·m^{-2}·year^{-1}) than did flat upland and south-facing sites (0.12 ± 0.06 kg C·m^{-2}·year^{-1}) over the first 10 years since fire.

Woody debris and char-layer carbon following fire

OL consumed in fire (1-year post-fire measurement) was the most significant predictor of the distribution of fire-killed black spruce in either standing or downed woody debris pools. As the amount of OL consumed in fire increased, the relative amount of DDWD carbon measured a decade following fire increased (Fig. 5). DDWD C pools were also affected by parent material clay content (Table 3), with there being generally less DDWD in sites with higher mineral soil clay content. There was also a weak interaction between clay content and aspect (Table 3) owing to the co-occurrence of higher clay but also lower DDWD in flat lowlands. Total DDWD C stocks increased marginally with increased elevation (Table 3). For the four categorical landscape positions, the DDWD pool was larger in flat uplands (0.38 ± 0.08 kg C·m^{-2}) than in flat lowlands (0.11 ± 0.09 kg C·m^{-2}), but there were no other significant differences by aspect (Fig. 6). Compared with pre-fire stand conditions, 39% ± 2% of black spruce carbon was still in standing aboveground pools (SDWD) 10 years after fire, with a range of 0%–49%. While there were no statistical differences in SDWD C pools across landscape positions ($p = 0.47$), flat lowlands generally had more SDWD C (0.60 ± 0.12 kg C·m^{-2}) than flat uplands (0.55 ± 0.11 kg C·m^{-2}), and there was a marginal effect of elevation on these pools across all sites ($F = 3.55$, $p = 0.07$).

In the wildfires included in this study, the amount of char produced per unit carbon consumed (char conversion rate) declined as the depth of the OL increased ($R^2 = 0.72$, $p < 0.002$), as indicated by year 1 post-fire measurements. One year after burning, north-

Fig. 4. Organic layer (OL) physical properties are regressed against mineral soil clay content. (*a*) Mean bulk density (BD) for the whole organic layer measured 10 years after wildfire and (*b*) the change in OL bulk densities from 1 to 10 years following wildfire both declined with increasing mineral soil clay content. (*c*) The change in OL depths from 1 to 10 years following wildfire increased with mineral soil clay content. Coefficients describing the regression lines (± standard error): (*a*) $\beta_0 = 0.174 \pm 0.013$, $\beta_1 = -0.0017 \pm 0.0005$; (*b*) $\beta_0 = 0.080 \pm 0.013$, $\beta_1 = -0.0013 \pm 0.0005$; and (*c*) $\beta_0 = -0.150 \pm 1.443$, $\beta_1 = 0.125 \pm 0.051$.



ern aspects had accumulated fewer products from burning in the presence of a char layer (0.20 ± 0.03 kg C·m^{-2}) than had south-facing aspects (0.30 ± 0.04 kg C·m^{-2}), which is consistent with char conversion rates being higher on south-facing slopes with shallower pre-fire OL depths. Ten years after wildfire, there were no significant differences in the pool sizes of the char-layer C across all aspects ($p = 0.29$), and what was operationally defined as the char layer in field sampling was larger than that measured in year 1 across all sites (0.45 ± 0.04 kg C·m^{-2}). The C pool size of this surface char layer was approximately double that of the DDWD C pool and comprised 5%–30% of the mean OL SOC stock remaining across all landscape positions (Fig. 6).

Discussion

Decadal change in SOC, BD, and OL

In this study, the largest factors influencing the accumulation of OL carbon stocks a decade following wildfire were landscape position and presence of fine-textured materials in the parent material. As hypothesized, sites lacking fine-textured soils (mainly underlain with colluvium) accumulated organic material more slowly in the decade following fire. While there is a developed literature describing changes in OL thickness and SOC stocks on different landscape positions in black spruce forests of interior Alaska (cf. Viereck et al. 1983; Van Cleve et al. 1983; Van Cleve and Yarie 1986; Kane and Vogel 2009; Ping et al. 2010), regional variation is large (Johnson et al. 2011; Turetsky et al. 2011). Prior research has illustrated the strong control of parent material texture on organic soil development along a catena from an alluvial floodplain in interior Alaska (Viereck et al. 1993), and this work suggests that underlying changes in parent material texture can explain additional variation in the recovery of SOC stocks following disturbances in uplands as well as lowlands. Previous work along the Boreal Forest Transect Case Study has clearly demonstrated the strong control of parent material clay content and its interaction with landscape position on aboveground biomass production and SOC within the forest floor (Bhatti and Apps 2000), but to our knowledge, this is the first direct assessment of how OL recovery following wildfire is influenced by parent material texture and its interaction with physiography.

While landscape position played a strong role in governing the depth of burn across the sites in this study (Fig. 2; Kane et al. 2007; Turetsky et al. 2011), mineral soil texture and its interactions with landscape position did a better job in explaining variation in the percent change in OL in the year following wildfire and in the decade following wildfire (Table 1). The influence of mineral soil texture was likely present at the time of burning in controlling depth of burn as well as in the years following fire. An increase in mineral soil clay content increases site water holding capacity, which has been shown to decrease fire severity (Bergeron et al. 2007; see also Peng et al. 1998) while also increasing aboveground biomass in upland boreal forests (Bhatti and Apps 2000; Banfield et al. 2002). These findings suggest that accurate appraisals of fire severity effects on OL consumption and recovery in the years following wildfire need to consider parent material texture in addition to landscape position.

Combined, aspect and parent material texture were the strongest predictors of OL carbon recovery in the decade following wildfire (Table 2*c*). While this is due to increased OL recovery with increased fineness of parent material texture as well as increases in BD, we highlight here that changes in OL BD over the last decade were highly variable across sites and likely represent the largest source of error in our SOC recovery estimates (Table 2*b*). Sources contributing to variability in BD measurements likely change with landscape position and antecedent OL characteristics. For example, the colonization and growth of surface moss over the last decade following wildfire would likely result in decreasing BD, whereas the subsidence of organic layers would increase BD measurements in year 10 compared with those measured 1 year following fire. Shallow soils following wildfire are not as susceptible to subsidence as deeper organic soils, especially when underlain with fine-textured mineral soil. South-facing slopes had the greatest amount of OL consumption during fires and therefore exhibited little subsidence or change in BD over 10 years (increased by 0.02 ± 0.02 g·cm^{-3}), while flat uplands had much greater OL depths following wildfire, had greater change in BD over 10 years (increased by 0.07 ± 0.02 g·cm^{-3}), and had the highest BD, on average, a decade after wildfire (0.16 ±

Fig. 5. The proportional amount of down dead woody debris 10 years following wildfire increased as the severity of burning within the organic layer increased across all sites. Coefficients describing the regression line: $\beta_0 = -8.85 \pm 10.55$, $\beta_1 = 0.40 \pm 0.17$.

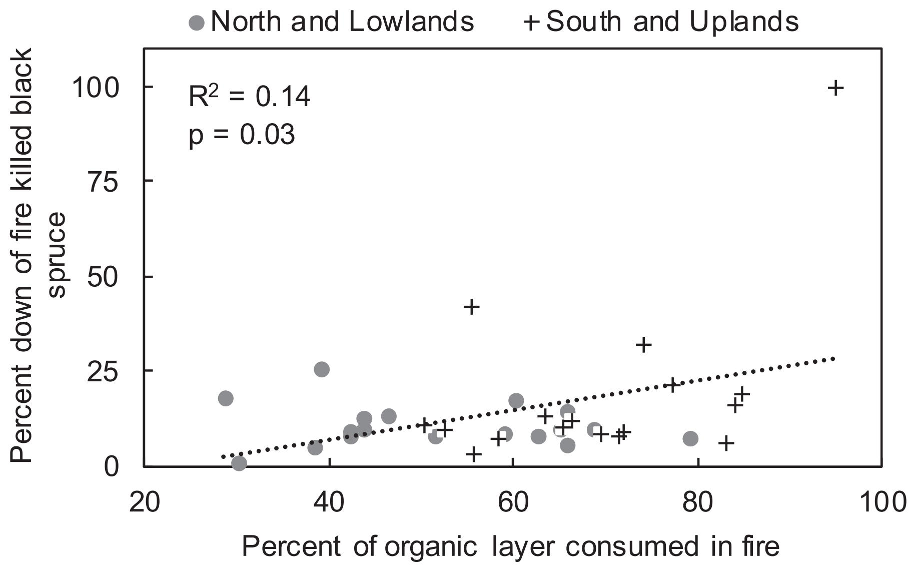


Table 3. Mixed-effects model describing changes in the pool size of organic carbon in dead down woody debris 10 years after fire.

	Dead down woody debris carbon		
Effect	df	*F* value	*p*
Slope	28	0.52	0.4771
Elevation	28	3.74	0.0633
Percent clay	28	9.40	0.0048**
Aspect	28	1.27	0.2698
Aspect × clay	28	3.87	0.0592*
Slope × clay	28	0.28	0.5994
Elevation × clay	28	1.97	0.1711
Slope × elevation	28	0.48	0.4959
Slope × elevation × clay	28	0.02	0.8843

Note: Asterisks denote level of significance: *, $p \leq 0.05$; **, $p < 0.01$.

0.03 g·cm^{-3}). The countervailing effects of increased relative OL recovery and low BD change on sites that did not have much organic matter remaining following fire vs. sites with deep OL remaining following fires, which are susceptible to subsidence in the post-burn environment when underlain with relatively coarse-textured soils, highlight the importance of texture, aspect, and their interaction in determining SOC accumulation following fire (Table 2*c*).

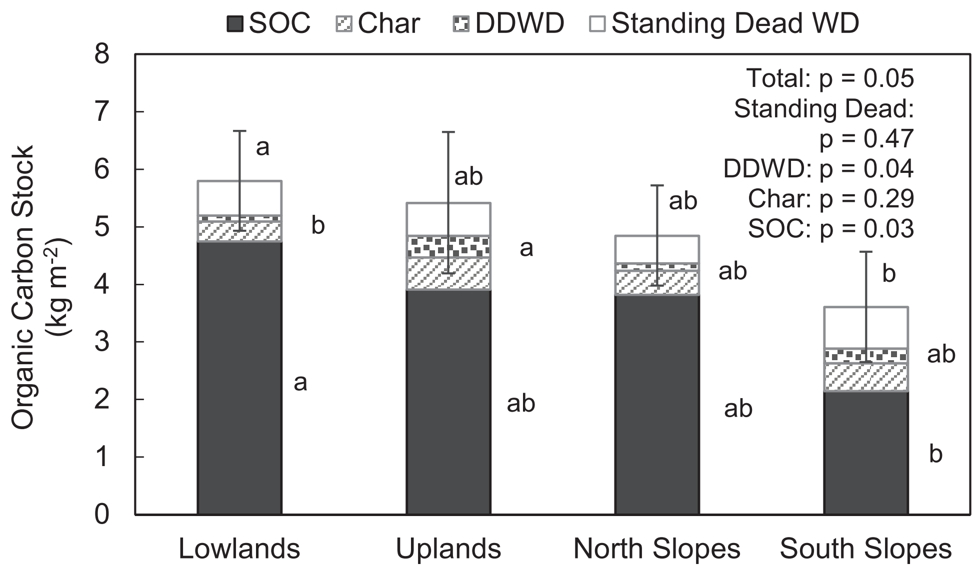
The fact that some upland sites did not demonstrate statistical increases in SOC a decade after year 1 measurements suggests that inputs from primary production have been less than or equal to C outputs from decomposition. Prior work at these sites has shown that severe burning (defined by OL consumption) is likely to alter the successional trajectory of regenerating vegetation communities (Gibson et al. 2016). As depth of burn increased, the regeneration of *Sphagnum* mosses and *Picea mariana* declined, with no *Sphagnum* regeneration occurring at sites with >65% OL consumption (Gibson et al. 2016). Sites with a greater percentage of fine-textured mineral soil exhibited the lowest reduction in OL and are therefore most likely to self-replace with a *Picea mariana* overstory and associated vegetation–moss communities. This is an important consideration with respect to the C balance of early successional forests because not only do mosses often dominate C inputs to soil pools but they also produce biomass that decomposes more slowly than most woody tissues (Turetsky et al. 2010; Bona et al. 2013). The effects of these early successional changes on SOC may not be evident in 10 years but certainly have implications for long-term ecosystem C accumulation.

Lower moss inputs in the decade following fire could have lessened SOC accumulation at more severely burned sites underlain with coarser textured mineral soil, but mean rates of SOC accrual across all sites in this study were at the high end of published values (Manies et al. 2016). Rates of soil C accumulation following disturbance are not linear, with rapid accumulation occurring in early succession and attenuation occurring with stand maturation (Harden et al. 2000). Studies investigating SOC accumulation rates that focus on short temporal dynamics often overestimate rates, which likely explains our relatively high estimates of SOC accumulation (Manies et al. 2016). We suggest that interpreting changes in the rates of SOC accumulation in this study are most appropriate in the relative sense, owing to the short time since stand-replacing fire. As such, landscape positions that promote higher net primary production (south-facing and upland sites) accumulated SOC more rapidly in the first decade following fire, particularly when underlain with finer textured parent material (aspect × clay interaction; Table 2*c*). This is in agreement with post-disturbance studies throughout the boreal region (cf. Thiffault et al. 2011), wherein most forests have experienced wildfire or harvest. In Canada (Bhatti and Apps 2000), Siberia (Siewert et al. 2015), and Fennoscandia (Callesen et al. 2003; Strand et al. 2016), an increase in OL SOC is generally observed in upland boreal forests underlain with finer textured soil.

Distribution of organic C pools a decade following fire

The interaction between aspect and fine-textured mineral soil in governing the severity of burning into the OL had a resulting effect on the fate of dead woody debris in either standing or down pools. This is perhaps not surprising because of the shallow lateral root system of black spruce (LeBarron 1945), which renders this species particularly vulnerable to uprooting when the canopy is more exposed (Ruel 1995; Angers et al. 2011); this would be prevalent with severe burning at higher elevation sites underlain with drier soils lacking fine textures. In addition, increased depth of burn consumes and (or) kills shallow roots, making residual SDWD more susceptible to windfall and increasing DDWD pools following wildfire (Smirnova et al. 2008). Prior work has suggested large changes in the residence times of woody debris C in standing or down pools, with downed wood likely representing a signifi-

Fig. 6. Changes in the distributions of organic carbon pools across four landscape positions a decade following wildfire. Different letters denote significant differences in pool sizes across landscape positions (no significant differences across char or standing dead woody debris (WD) pools). Error bars represent standard errors of pool sums for a given landscape class. DDWD, down dead woody debris; SOC, soil organic carbon.



cant component of the total OL C stock (Moroni et al. 2015). For example, Seedre et al. (2014) documented a redistribution of SDWD to the DDWD pool approximately 8 years after wildfire. This input is likely to be more important in lowland forests with deeper antecedent OL depths and continuous proliferation of mosses (Hagemann et al. 2010; Jacobs et al. 2015), which may be more effective in protecting wood-derived C in the OL from burning in subsequent fire events. However, our study suggests that more time may be required for flat lowland SDWD to be incorporated in downed or soil C pools than SDWD in upland and sloped sites. Prior research along a boreal Canadian fire chronosequence has estimated that between 10% and 60% of OL C stocks were derived from woody biomass inputs, with the majority of this input occurring between 20 and 40 years following fire (Manies et al. 2005). Sensitivity analyses conducted by Manies et al. (2005) suggested that variation in woody debris inputs, in addition to variation in post-fire inputs from primary production and the products of burning (char) with changes in landscape position, could account for large differences in deep-soil carbon. We suggest here that in addition to these variables, the underlying mineral soil texture exerts direct and indirect controls on the distribution of woody debris C. Knowledge of mineral soil composition in relation to landscape position should therefore improve estimates of SOC recovery following wildfire.

Field estimates of the heterogeneous residues from burning accumulated at the surface of the soil, or "soil char layer", are somewhat problematic from a C assessment perspective because ocular estimates cannot distinguish different types of char (Soucémarianadin et al. 2015*a*) and its vertical stratification changes in variable ways with time since disturbance as it is leached within and from the soil (Santín et al. 2016). That said, surface char material is distinct from live or unburned moss and can represent a significant portion of the post-fire SOC stock (Harden et al. 2004). In this study, C within the operationally defined char pool comprised a significant amount of variation in SOC stocks across sites, was almost double the size of the DDWD C pool, and was generally larger than measurements obtained 1 year following fire. However, we caution here that char delineated in this way 10 years after fire is likely not representative of the char layer measured in the year following fire, and an investigation of the fate and mobility of char in the decade following fire was not possible in this study. While we measured changes in SDWD relative to DDWD pools, it is hard to know how charred woody debris continued to contribute to SOC pools in the decade following fire. Earlier study across the Alaskan study region had observed higher accumulation of burn residues in mineral soils from southerly aspects with more complete combustion of the OL than north-facing or toe-slope forests (Kane et al. 2007), and the products of burning have been shown to accumulate within the surface mineral soils of other boreal spruce forests (Soucémarianadin et al. 2015*b*). These prior investigations have shown little correlation between fire frequency and the accumulation of burn residues in mineral soil, suggesting alternative mechanisms of fire-derived C stabilization or a susceptibility to combustion of this pool in subsequent wildfires before it reaches the mineral soil (Kane et al. 2010). However, there has been little attention to the complexation of fire-derived C with clay textured mineral soil in boreal forests, which has been shown to exert strong control over SOC stabilization in other forested ecosystems (Santos et al. 2012; Wang et al. 2016). The co-occurrence of increased fine textures in mineral soils and deeper OL depths in flat lowland landscape positions may be conducive to the stabilization of relatively more char C in deeper soils (Knicker 2007). Systematic examination of the products of burning in deep soil profiles with changes in landscape position (cf. Manies et al. 2005) and mineral soil texture would certainly be helpful in explaining the large variation observed in the size of this significant C pool a decade following fire.

Broader implications for SOC assessments

In recent geospatial analyses synthesizing hundreds of SOC profiles in Alaska, climatic variables (temperature and soil moisture) explained the most variance in regional SOC stocks, as might be expected (Johnson et al. 2011; Mishra and Riley 2012). In these studies, soil moisture was a significant predictor of SOC stocks, independent of precipitation regime, highlighting the important controls of topographic attributes and parent material on water holding capacity. Mishra and Riley (2012) suggest that poor representation of these controls on "soil wetness" could contribute to large uncertainty in predicted SOC stocks in response to future climate change in Earth system modeling scenarios. The data presented here agree with this assessment and suggest that incorporating an understanding of parent material texture, as provided in the Family level of soil taxonomy, would constrain the variation in SOC stock estimates in geospatial analyses of SOC changes in response to disturbances.

Conclusions

Parent material texture as a "state factor" control over soil development was shown to be at least as important as landscape position in controlling SOC stock recovery in the decade following wildfire in interior Alaskan black spruce forests. Together, parent material and landscape position explained most of the variance in SOC changes over 10 years since burning, with the largest contributions to variability in stock assessments likely being associated with changes in BD and char C pools across sites. The influence of mineral soil texture was likely present at the time of burning in controlling depth of burn as well as in the years following fire; this was evident in the significant effects of texture on the percentage of OL consumed measured in the year following fire. Aspect and parent material texture also influenced the fate of dead woody debris in either standing or down pools in the decade following fire, largely through their controls on fire severity and OL consumed in fire. Collectively, these findings demonstrate the importance of understanding the source of soils in designing studies to understand fire ecology in interior Alaska.

Acknowledgements

We thank Rosanne Broyd and Melissa Nelson (Bergstrom) for assistance with fieldwork. We acknowledge the following financial support: a Natural Sciences and Engineering Research Council of Canada (NSERC) Discovery grant to M.R.T.; a Michigan Tech University Research Excellence Fund Seed (REF-Seed) Grant to E.S.K.; the Bonanza Creek Long-Term Ecological Research program (funded jointly by NSF Grant DEB-0620579 and USDA Forest Service Pacific Northwest Research Grant PNW01-JV11261952-231); NASA (grant NNG04GD25G); and in kind support from the USDA Forest Service, Northern Research Station.

References

Amundson, R., and Jenny, H. 1997. On a state factor model of ecosystems. BioScience, **47**(8): 536–543. doi:10.2307/1313122.

Angers, V.A., Gauthier, S., Drapeau, P., Jayen, K., and Bergeron, Y. 2011. Tree mortality and snag dynamics in North American boreal tree species after a wildfire: a long-term study. Int. J. Wildland Fire, **20**: 751–763. doi:10.1071/WF10010.

Banfield, G.E., Bhatti, J.S., Jiang, H., and Apps, M.J. 2002. Variability in regional scale estimates of carbon stocks in boreal forest ecosystems: results from West-Central Alberta. For. Ecol. Manage. **169**: 15–27. doi:10.1016/S0378-1127(02)00292-X.

Beget, J.E., Stone, D., and Verbyla, D.L. 2006. Regional overview of interior Alaska. *In* Alaska's changing boreal forest. *Edited by* F.S. Chapin, III, M.W. Oswood, K. Van Cleve, L.A. Viereck, and D.L. Verbyla. Oxford University Press, New York. pp. 12–20.

Bergeron, Y., Drapeau, P., Gauthier, S., and Lecomte, N. 2007. Using knowledge of natural disturbances to support sustainable forest management in the northern Clay Belt. The Forestry Chronicle, **83**(3): 326–337.

Bhatti, J.S., and Apps, M.J. 2000. Carbon and nitrogen storage in upland boreal forests. *In* Global climate change and cold regions ecosystems. *Edited by* R. Lal, J.M. Kimble, and B.A. Stewart. CRC Press, Boca Raton, Fla. pp. 79–90.

Boby, L.A., Schuur, E.A.G., Mack, M.C., Verbyla, D., and Johnstone, J.F. 2010. Quantifying fire severity, carbon, and nitrogen emissions in Alaska's boreal forest. Ecol. Appl. **20**(6): 1633–1647. doi:10.1890/08-2295.1. PMID:20945764.

Bona, K.A., Fyles, J.W., Shaw, C., and Kurz, W.A. 2013. Are mosses required to accurately predict upland black spruce forest soil carbon in national-scale forest C accounting models? Ecosystems, **16**: 1071–1086. doi:10.1007/s10021-013-9668-x.

Bond-Lamberty, B., Wang, C., and Gower, S.T. 2004. Net primary production and net ecosystem production of a boreal black spruce wildfire chronosequence. Global Change Biol. **10**: 473–487. doi:10.1111/j.1529-8817.2003.0742.x.

Bond-Lamberty, B., Peckham, S.D., Ahl, D.E., and Gower, S.T. 2007. Fire as the dominant driver of central Canadian boreal forest carbon balance. Nature, **450**: 89–92. doi:10.1038/nature06272. PMID:17972883.

Brown, C.D., and Johnstone, J.F. 2011. How does increased fire frequency affect carbon loss from fire? A case study in the northern boreal forest. Int. J. Wildland Fire, **20**: 829–837. doi:10.1071/WF10113.

Brown, D.R.N., Jorgenson, M.T., Douglass, T.A., Romanovsky, V.E., Kielland, K., Hiemstra, C., Euskirchen, S.E., and Ruess, R.W. 2015. Interactive effects of wildfire and climate on permafrost degradation in Alaskan lowland forests. J. Geophys. Res.: Biogeosci. **120**: 1619–1637. doi:10.1002/2015JG003033.

Brown, J.K. 1974. Handbook for inventorying downed woody material. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah, Gen. Tech. Rep. INT-16.

Callesen, I., Liski, J., Raulund-Rasmussen, K., Olsson, M.T., Tau-Strand, L., Vesterdal, L., and Westman, C.J. 2003. Soil carbon stores in Nordic well-drained forest soils — relationships with climate and texture class. Global Change Biol. **9**: 358–370. doi:10.1046/j.1365-2486.2003.00587.x.

Dokuchaev, V.V. 1879. Abridged historical account and critical examination of the principal soil classifications existing. Transactions of the Petersburg Society of Naturalists, **1**: 64–67.

Gee, G.W., and Bauder, J.W. 1986. Particle-size analysis. *In* Methods of soil analysis: Part 1 — physical and mineralogical methods. *Edited by* A. Klute. Soil Science Society of America (SSSA), Madison, Wis. SSSA Book Series 5. pp. 404–408.

Gibson, C.M., Turetsky, M.R., Cottenie, K., Kane, E.S., Houle, G., and Kasischke, E.S. 2016. Variation in plant community composition and vegetation carbon pools a decade following a severe fire season in interior Alaska. J. Veg. Sci. **27**: 1187–1197. doi:10.1111/jvs.12443.

Hagemann, U., Moroni, M.T., Gleißner, J., and Makeschin, F. 2010. Accumulation and preservation of dead wood upon burial by bryophytes. Ecosystems, **13**: 600–611. doi:10.1007/s10021-010-9343-4.

Hambidge, G. 1938. Soils and men — a summary. *In* Soils and men: yearbook of agriculture 1938. U.S. Department of Agriculture, Washington, D.C. pp. 38–40.

Harden, J.W., Trumbore, S.E., Stocks, B.J., Hirsch, A., Gower, S.T., O'Neill, K.P., and Kasischke, E.S. 2000. The role of fire in the boreal carbon budget. Global Change Biol. **6**(S1): 174–184. doi:10.1046/j.1365-2486.2000.06019.x.

Harden, J.W., Neff, J.C., Sandberg, D.V., Turetsky, M.R., Ottmar, R., Gleixner, G., Fries, T.L., and Manies, K.L. 2004. Chemistry of burning the forest floor during the FROSTFIRE experimental burn, interior Alaska, 1999. Global Biogeochem. Cycles, **18**: GB3014. doi:10.1029/2003GB002194.

Harden, J.W., Manies, K.L., O'Donnell, J.A., Johnson, K., Frolking, S., and Fan, Z. 2012. Spatiotemporal analysis of black spruce forest soils and implications for the fate of C. J. Geophys. Res. **117**: G01012. doi:10.1029/2011JG001826.

Hoy, E.E., Turetsky, M.R., and Kasischke, E.S. 2016. More frequent burning increases vulnerability of Alaskan boreal black spruce forests. Environ. Res. Lett. **11**(9): 095001. doi:10.1088/1748-9326/11/9/095001.

Jacobs, J., Work, T., Paré, D., and Bergeron, Y. 2015. Paludification of boreal soils reduces wood decomposition rates and increases wood-based carbon storage. Ecosphere, **6**(12): 1. doi:10.1890/ES14-00063.1.

Johnson, K.D., Harden, J., McGuire, A.D., Bliss, N.B., Bockheim, J.G., Clark, M., Nettleton-Hollingsworth, T., Jorgenson, M.T., Kane, E.S., Mack, M., O'Donnell, J., Ping, C.-L., Schuur, E.A.G., Turetsky, M.R., and Valentine, D.W. 2011. Soil carbon distribution in Alaska in relation to soil-forming factors. Geoderma, **167–168**: 71–84. doi:10.1016/j.geoderma.2011.10.006.

Johnson, M.G., and Kern, J.S. 2003. Quantifying the organic carbon held in forested soils of the United States and Puerto Rico. *In* The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. *Edited by* J.M. Kimble, L.S. Heath, R.A. Birdsey, and R. Lal. CRC Press, Boca Raton, Fla. pp. 47–72.

Johnstone, J.F., Chapin, F.S., III, Hollingsworth, T.N., Mack, M.C., Romanovsky, V., and Turetsky, M. 2010. Fire, climate change, and forest resilience in interior Alaska. Can. J. For. Res. **40**: 1302–1312. doi:10.1139/X10-061.

Kane, E.S., and Vogel, J.G. 2009. Patterns of total ecosystem carbon storage with changes in soil temperature in boreal black spruce forests. Ecosystems, **12**(2): 322–335. doi:10.1007/s10021-008-9225-1.

Kane, E.S., Kasischke, E.S., Valentine, D.W., Turetsky, M.R., and McGuire, A.D. 2007. Topographic influences on wildfire consumption of soil organic carbon in interior Alaska: implications for black carbon accumulation. J. Geophys. Res. **112**: G03017. doi:10.1029/2007JG000458.

Kane, E.S., Hockaday, W.C., Turetsky, M.R., Masiello, C.A., Valentine, D.W., Finney, B.P., and Baldock, J.A. 2010. Topographic controls on black carbon accumulation in Alaskan black spruce forest soils: implications for organic matter dynamics. Biogeochemistry, **100**: 39–56. doi:10.1007/s10533-009-9403-z.

Kasischke, E.S., and Turetsky, M.R. 2006. Recent changes in the fire regime across the North American boreal region — spatial and temporal patterns of burning across Canada and Alaska. Geophys. Res. Lett. **33**: L09703. doi:10.1029/2006GL025677.

Kasischke, E.S., and Hoy, E.E. 2012. Controls on carbon consumption during Alaskan wildland fires. Global Change Biol. **18**: 685–699. doi:10.1111/j.1365-2486.2011.02573.x.

Kasischke, E.S., Turetsky, M.R., Ottmar, R.D., French, N.H.F., Hoy, E.E., and Kane, E.S. 2008. Evaluation of the composite burn index for assessing fire severity in Alaskan black spruce forests. Int. J. Wildland Fire, **17**: 515–526. doi:10.1071/WF08002.

Kasischke, E.S., Verbyla, D.L., Rupp, T.S., McGuire, A.D., Murphy, K.A., Jandt, R., Barnes, J.L., Hoy, E.E., Duffy, P.A., Calef, M., and Turetsky, M.R. 2010. Alaska's changing fire regime — implications for the vulnerability of its boreal forests. Can. J. For. Res. **40**: 1313–1324. doi:10.1139/X10-098.

Kishchuk, B.E., Thiffault, E., Lorente, M., Quideau, S., Keddy, T., and Sidders, D. 2015. Decadal soil and stand response to fire, harvest, and salvage-logging disturbances in the western boreal mixedwood forest of Alberta, Canada. Can. J. For. Res. **45**: 141–152. doi:10.1139/cjfr-2014-0148.

Knicker, H. 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry, **85**(1): 91–118. doi:10.1007/s10533-007-9104-4.

LeBarron, R.K. 1945. Adjustment of black spruce root systems to increasing depth of peat. Ecology, **26**: 309–311. doi:10.2307/1932413.

Manies, K.L., Harden, J.W., Bond-Lamberty, B.P., and O'Neill, K.P. 2005. Woody debris along an upland chronosequence in boreal Manitoba and its impact on long-term carbon storage. Can. J. For. Res. **35**: 472–482. doi:10.1139/x04-179.

Manies, K.L., Harden, J.W., Fuller, C.C., and Turetsky, M.R. 2016. Decadal and long-term boreal soil carbon and nitrogen sequestration rates across a variety of ecosystems. Biogeosciences, **13**: 4315–4327. doi:10.5194/bg-13-4315-2016.

McGuire, A.D.M., Melillo, J.M., Kicklighter, D.W., and Joyce, L.A. 1995. Equilibrium responses of soil carbon to climate change: empirical and process-based estimates. J. Biogeogr. **22**: 785–796. doi:10.2307/2845980.

Mishra, U., and Riley, W.J. 2012. Alaskan soil carbon stocks: spatial variability and dependence on environmental factors. Biogeosciences, **9**: 3637–3645. doi:10.5194/bg-9-3637-2012.

Moroni, M.T., Morris, D.M., Shaw, C., Stokland, J.N., Harmon, M.E., Fenton, N.J., Merganičová, K., Merganič, J., Okabe, K., and Hagemann, U. 2015. Buried wood: a common yet poorly documented form of deadwood. Ecosystems, **18**(4): 605–628. doi:10.1007/s10021-015-9850-4.

Muhs, D.R., Ager, T.A., Bettis, E.A., III, McGeehin, J., Been, J.M., Beget, J.E., Pavich, M.J., Stafford, T.W., Jr., and Stevens, D.S.P. 2003. Stratigraphy and palaeoclimatic significance of Late Quaternary loess–palaeosol sequences of the Last Interglacial–Glacial cycle in central Alaska. Quat. Sci. Rev. **22**: 1947–1986. doi:10.1016/S0277-3791(03)00167-7.

Mulligan, D.K. 2005. Soil survey of the Greater Fairbanks area. USDA Natural Resources Conservation Service, Fairbanks, Alaska.

Nalder, I.A., and Wein, R.W. 1998. A new forest floor corer for rapid sampling, minimal disturbance and adequate precision. Silva Fenn. **32**(4): 373–382. doi:10.14214/sf.678.

Nalder, I.A., Wein, R.W., Alexander, M.E., and de Groot, W.J. 1997. Physical properties of dead and downed round-wood fuels in the boreal forests of Alberta and Northwest Territories. Can. J. For. Res. **27**: 1513–1517. doi:10.1139/x97-083.

O'Donnell, J.A., Harden, J.W., Manies, K.L., Jorgenson, M.T., Kanevskiy, M., and Xu, X. 2013. Soil data from fire and permafrost-thaw chronosequences in upland black spruce (*Picea mariana*) stands near Hess Creek and Tok, interior Alaska. U.S. Geological Survey Open-File Report 2013–1045.

Pan, Y., Birdsey, R.A., Phillips, O.L., and Jackson, R.B. 2013. The structure, distribution, and biomass of the world's forests. Annu. Rev. Ecol. Evol. Syst. **44**: 593–622. doi:10.1146/annurev-ecolsys-110512-135914.

Peng, C., Apps, M.J., Price, D.T., Nalder, I.A., and Halliwell, D.H. 1998. Simulating carbon dynamics along the boreal forest transect case study (BFTCS) in Central Canada: 1. Model validation. Global Biogeochem. Cycles, **12**: 381–392. doi:10.1029/98GB00351.

Péwé, T.L. 1955. Origin of the upland silt near Fairbanks, Alaska. Geol. Soc. Am. Bull. **66**: 699–724. doi:10.1130/0016-7606(1955)66[699:OOTUSN]2.0.CO;2.

Péwé, T.L., and Reger, R.D. 1983. Guidebook to permafrost and quaternary geology along the Richardson and Glenn Highways between Fairbanks and Anchorage, Alaska. Alaska Division of Geological and Geophysical Surveys, Department of Natural Resources, Fairbanks, Alaska. doi:10.14509/263.

Ping, C.-L., Boone, R.D., Clark, M.H., Packee, E.C., and Swanson, D.K. 2006. State factor control of soil formation in interior Alaska. *In* Alaska's changing boreal forest. *Edited by* F.S. Chapin, III, M.W. Oswood, K. Van Cleve, L.A. Viereck, and D.L. Verbyla. Oxford University Press, New York. pp. 21–38.

Ping, C.-L., Michaelson, G.J., Kane, E.S., Packee, E.C., Stiles, C.A., Swanson, D.K., and Zaman, N.D. 2010. Carbon stores and biogeochemical properties of soils under black spruce forest, Alaska. Soil Sci. Soc. Am. J. **74**: 969–978. doi:10.2136/sssaj2009.0152.

Ruel, J.-C. 1995. Understanding windthrow: silvicultural implications. For. Chron. **71**: 434–445. doi:10.5558/tfc71434-4.

Santín, C., Doerr, S.H., Kane, E.S., Masiello, C.A., Ohlson, M., de la Rosa, J.M., Preston, C.M., and Dittmar, T. 2016. Towards a global assessment of pyrogenic carbon from vegetation fires. Global Change Biol. **22**: 76–91. doi:10.1111/gcb.12985.

Santos, F., Torn, M.S., and Bird, J.A. 2012. Biological degradation of pyrogenic organic matter in temperate forest soils. Soil Biol. Biochem. **51**: 115–124. doi:10.1016/j.soilbio.2012.04.005.

Seedre, M., Taylor, A.R., Brassard, B.W., Chen, H.Y.H., and Jõgiste, K. 2014. Recovery of ecosystems carbon stocks in young boreal forests: a comparison of harvesting and wildfire disturbance. Ecosystems, **17**: 851–863. doi:10.1007/s10021-014-9763-7.

Siewert, M.B., Hanisch, J., Weiss, N., Kuhry, P., Maximov, T.C., and Hugelius, G. 2015. Comparing carbon storage of Siberian tundra and taiga permafrost ecosystems at very high spatial resolution. J. Geophys. Res.: Biogeosci. **120**: 1973–1994. doi:10.1002/2015JG002999.

Slaughter, C.W., and Viereck, L.A. 1986. Climate characteristics of the taiga in interior Alaska. *In* Forest ecosystems in the Alaskan taiga. *Edited by* K. Van Cleve, F.S. Chapin, III, P.W. Flanagan, L.A. Viereck, and C.T. Dyrness. Springer-Verlag, New York. pp. 9–21. doi:10.1007/978-1-4612-4902-3_2.

Smirnova, E., Bergeron, Y., Brais, S., and Granström, A. 2008. Postfire root distribution of Scots pine in relation to fire behaviour. Can. J. For. Res. **38**: 353–362. doi:10.1139/X07-127.

Soucémarianadin, L.N., Quideau, S.A., Wasylishen, R.E., and Munson, A.D. 2015*a*. Early-season fires in boreal black spruce forests produce pyrogenic carbon with low intrinsic recalcitrance. Ecology, **96**(6): 1575–1585. doi:10.1890/14-1196.1.

Soucémarianadin, L.N., Quideau, S.A., MacKenzie, M.D., Munson, A.D., Boiffin, J., Bernard, G.M., and Wasylishen, R.E. 2015*b*. Total and pyrogenic carbon stocks in black spruce forest floors from eastern Canada. Org. Geochem. **82**: 1–11. doi:10.1016/j.orggeochem.2015.02.004.

Strand, L.T., Callesen, I., Dalsgaard, L., and de Wit, H.A. 2016. Carbon and nitrogen stocks in Norwegian forest soils — the importance of soil formation, climate, and vegetation type for organic matter accumulation. Can. J. For. Res. **46**: 1459–1473. doi:10.1139/cjfr-2015-0467.

Swanson, D.K. 1996. Soil geomorphology on bedrock and colluvial terrain with permafrost in central Alaska, USA. Geoderma, **71**: 157–172. doi:10.1016/0016-7061(96)00003-1.

Thiffault, E., Hannam, K.D., Paré, D., Titus, B.D., Hazlett, P.W., Maynard, D.G., and Brais, S. 2011. Effects of forest biomass harvesting on soil productivity in boreal and temperate forests — a review. Environ. Rev. **19**: 278–309. doi:10.1139/a11-009.

Thomas, S.C., and Martin, A.R. 2012. Carbon content of tree tissues: a synthesis. Forests, **3**: 332–352. doi:10.3390/f3020332.

Todd, S.K., and Jewkes, H.A. 2006. Fire in Alaska: a history of organized fire suppression and management in the last frontier. Agriculture and Forestry Experiment Station Bulletin 114. University of Alaska Fairbanks, Fairbanks, Alaska.

Turetsky, M.R., Mack, M.C., Hollingsworth, T.N., and Harden, J.W. 2010. The role of mosses in ecosystem succession and function in Alaska's boreal forest. Can. J. For. Res. **40**: 1237–1264. doi:10.1139/X10-072.

Turetsky, M.R., Kane, E.S., Harden, J.W., Ottmar, R.D., Manies, K.L., Hoy, E., and Kasischke, E.S. 2011. Recent acceleration of biomass burning and carbon losses in Alaskan forests and peatlands. Nat. Geosci. **4**: 27–31. doi:10.1038/ngeo1027.

Van Cleve, K., and Yarie, J. 1986. Interaction of temperature, moisture, and soil chemistry in controlling nutrient cycling and ecosystem development in the taiga of Alaska. *In* Forest ecosystems in the Alaskan taiga. *Edited by* K. Van Cleve, F.S. Chapin, III, P.W. Flanagan, L.A. Viereck, and C.T. Dyrness. Springer-Verlag, New York. pp. 160–189. doi:10.1007/978-1-4612-4902-3_12.

Van Cleve, K., Oliver, L., Schlentner, R., Viereck, L.A., and Dyrness, C.T. 1983. Productivity and nutrient cycling in taiga forest ecosystems. Can. J. For. Res. **13**: 747–766. doi:10.1139/x83-105.

Van Cleve, K., Chapin, F.S., III, Dyrness, C.T., and Viereck, L.A. 1991. Element cycling in taiga forests: state-factor control. BioScience, **41**(2): 78–88. doi:10.2307/1311560.

Viereck, L.A., and Dyrness, C.T. 1979. Ecological effects of the Wickersham Dome fire near Fairbanks, AK. USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, Oregon, Gen. Tech. Rep. 90.

Viereck, L.A., Dyrness, C.T., Van Cleve, K., and Foote, M.J. 1983. Vegetation, soils, and forest productivity in selected forest types in interior Alaska. Can. J. For. Res. **13**: 703–720. doi:10.1139/x83-101.

Viereck, L.A., Dyrness, C.T., and Foote, M.J. 1993. An overview of the vegetation and soils of the floodplain ecosystems of the Tanana River, interior Alaska. Can. J. For. Res. **23**: 889–898. doi:10.1139/x93-117.

Wang, J.Y., Xiong, Z.Q., and Kuzyakov, Y. 2016. Biochar stability in soil: meta-analysis of decomposition and priming effects. GCB Bioenergy, **8**(3): 512–523. doi:10.1111/gcbb.12266.

White, J.D., Koepke, B.E., and Swanson, D.K. 2002. Soil survey of North Star area, Alaska. USDA Natural Resources Conservation Service, North Star Borough, Alaska.

Yarie, J. 1981. Forest fire cycles and life tables: a case study from interior Alaska. Can. J. For. Res. **11**: 554–562. doi:10.1139/x81-076.

Yarie, J., Kane, E.S., and Mack, M.M. 2007. Aboveground biomass equations for the trees of interior Alaska. Agricultural and Forestry Experiment Station, Bulletin 115. University of Alaska Press, Fairbanks, Alaska.