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Fuel accumulation in a high-frequency boreal wildfire regime: from wetland to upland

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Fuel accumulation in a high-frequency boreal wildfire regime: from wetland to upland

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Abstract

14 Although it is increasingly accepted that young (e.g., ≤ 30 years) stands originating from wildfire are
considerably less flammable than older stands in the boreal forest of North America, the role of fuel
16 availability and structure in this phenomenon has not been thoroughly investigated. As a regional study
in a high-frequency fire regime, detailed wildfire fuel loading and structure was measured in 66 sites
18 including both wetlands and uplands in the Boreal Plains landscape of Wood Buffalo National Park, in
northwestern Canada. Overall, a significant increase in total flammable biomass occurred in upland
20 sites after 97 years but this increase was not consistently observed in wetlands, except where there was
dense tree cover. Fuel accumulation was highly moderated by canopy fuels, as surface fuels were
22 relatively constant across differing site types and time since fire, averaging 0.4 kg m^{-2} . Significant but
gradual canopy fuel accumulation was observed in moist conifer upland forests dominated by mature
24 black spruce (*Picea mariana*) or white spruce (*Picea glauca*) over 100 years since fire. Outside of these
mature moist conifer uplands, there was no difference in total fuel loading between other upland forest
26 and across the gradient of treed to open wetlands.

1. Introduction

Paleoecological reconstructions from tree rings (Wallenius et al. 2011) and charcoal and pollen (Larsen
30 and MacDonald 1998) in boreal western Canada indicate a historic fire return interval that can be as
short as 50–100 years, on average. Given that young stands (e.g., ≤ 20 years postfire) may act as a
32 barrier to fire spread in the boreal forest (Niklasson and Granström 2000; Bernier et al. 2016), it is in
the most fire-prone parts of the boreal forest—that is, those with the greatest proportion of young
34 stands—that fire activity is likely to be the most limited by the lack of flammable biomass (Héon et al.
2014). Age-dependent thresholds such as the minimum canopy bulk density required to carry crown

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fire can play an important part in changing the flammability of a forest (Van Wagner 1977; Cruz et al. 2005), but how these important canopy structure characteristics change with time since fire in the boreal is poorly understood. Although empirical evidence from the Eurasian (Schimel and Granström 1997) and eastern North American boreal forest (Terrier et al. 2016) point to a general increase in biomass following fire, the rates and patterns by which stands become more flammable is only partially understood in the boreal plains of western Canada, a vast area of intermixed upland forests and wetland complexes.

Another open question is that of the flammability of boreal wetlands, which may cover a large (and even dominant) proportion of boreal landscapes and do indeed burn during periods of severe drought (Turetsky et al. 2004). The importance of flammable biomass accumulation has been mainly documented in upland forest (e.g. Bessie and Johnson 1995; Schimmel and Granström 1997). Whereas fuel accumulation on uplands is largely dependent of tree growth (Lavoie et al. 2010), this is not necessarily the case in boreal lowlands. The boreal forest of western Canada incorporates a wetness gradient from sedge-dominated (hydric) and treed wetlands to dry (xeric) upland forest (Raup 1935). With wetlands covering approximately 25% of the boreal forest of western Canada (Tarnocai et al. 2011), knowledge of fire spread across the entire range of flammable ecosystems is essential to understanding landscape-scale wildfire patterns. The contribution of wetlands to boreal burned area have been reported to be both positive (Turetsky et al. 2004) and negative (Hellberg et al. 2001; Erni et al. 2016). Recent studies have reported surface fuel and canopy fire consumption levels comparable to those of upland forests (Benscoter and Wieder 2003; Lukenbach et al. 2015). Recent examinations have documented the accumulation of wildfire fuels over time in forested bogs (Johnston et al. 2015), in paludified forests of the Abitibi clay belt in Ontario and Quebec (Terrier et al. 2016), alongside

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60 detailed fuels inventories in more minerotrophic wetlands by Schiks et al. (2016). However, a
systematic examination of fuel loading over time and along this moisture gradient from treeless
62 wetlands to upland forest in boreal Canada is lacking.

64 Here we present a dataset of 66 ground plots documenting wildfire fuel structure and composition
across a gradient from wetlands to xeric uplands, and spanning a range of time since fire, in Wood
66 Buffalo National Park, located in the province of Alberta and the Northwest Territories in Canada.
Specifically, this study has three objectives: (1) evaluate how fuel loading changes with time since fire;
68 (2) examine differences between wetlands and uplands up to 250 years since fire; and (3) compare fuel
loading across wetness gradients and time since fire with those from the Canadian Fire Behaviour
70 Prediction (FBP) System. In contrast to prior studies (e.g. Schimmel and Granström, 1997; Johnston et
al. 2015) that focused on a specific ecosystem (e.g. pine uplands or spruce wetlands), here we assessed
72 fuel loading and structure across the range of ecosystems present in a region. These results will help us
understand the landscape-level disturbance dynamics of the area and thereby allow for refinements of
74 our predictions of future fire activity in the North American boreal forest.

76 2. Materials and methods

This study took place entirely within Wood Buffalo National Park, which covers 4.4 Mha along the
78 Alberta-Northwest Territories border and is the largest national park with a frequent wildfire regime in
Canada. Burned area within the park varies widely, with 37 of the past 56 years recording less than 10
80 000 ha, while burned area exceeded 100 000 ha in eight years. The climate in the region is dry and
continental, with a mean annual temperature of -1.8°C, annual precipitation of 353 mm (Environment
82 and Climate Change Canada 2015), and annual potential evaporation of approximately 300 mm (Hare

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1980). The area lies in the northernmost extent of the Boreal Plains ecozone (Ecological Stratification Working Group 1996) and boreal mixedwood forest region (Downing and Pettapiece 2005). The park is primarily covered by expansive lacustrine and aeolian glacial sediments, with shallower soils in the Caribou and Birch Mountain plateaus in the southern and western edges of the park (Fenton et al. 2013). Coarse-textured upland mineral soils are dominated by pine (*Pinus banksiana*) or aspen (*Populus tremuloides*), with 40–45% of the park area composed of wetlands (Tarnocai et al. 2011) that can be a mixture of tree, shrub, or graminoid-dominated with at least 40 cm of underlying peat. Permafrost is classified as being only sporadic in the region (Heginbottom 1995) and isolated to small pockets in peatlands. Extensive floodplain grasslands exist in the Peace-Athabasca Delta nearby Lake Claire, the largest body of water in the parks (Figure 1). These wetlands are generally non-peat forming, and have been considered a disjunct prairie grassland community comparable to those 1000 km further south (Schwarz 1994). Reconstructions of the fire regime in the area prior to modern suppression indicate a mean fire return interval of 69 years (Larsen and MacDonald 1998). Though wildfire suppression has been active to some extent since the 1950s, managed wildfire has prevailed over full suppression, with large burned areas in 1981 (650 000 ha), and in 2015 (372 000 ha; Natural Resources Canada 2016). The fire-carrying capacity in Wood Buffalo National Park is evidenced by recent landscape-scale fires from 2012–2015 that burned an average of 5% of the area of the park per year (Natural Resources Canada 2016). The area of Wood Buffalo National Park was listed as being in a long-term abnormally dry period during the summer of 2012 and 2013, with more normal conditions in 2014 followed by severe long-term drought in 2015 (North American Drought Monitor 2016).

2.1 Site selection

A total of 66 ground plots covering a gradient of time since fire and the main vegetation types of the study area were selected and sampled in this study between 2012 and 2015 (Figure 1). These four years, though variable in terms of growing-season weather, were not atypical. The sites were sampled, from mid-June to mid-July of each year in order to minimize the effect of phenological changes. Given the remote nature of the area, site selection was limited to areas within 2 km of the two roads in the park ($n = 42$) (*cf.* Wallenius et al. 2011), except when there was a possibility for helicopter access ($n = 24$), in which case we could sample more remote sites (*cf.* Larsen 1997). Preliminary statistical explorations showed that that our sample size was suitable to meet our main objectives; there was thus no further sampling beyond 2015.

The nature of site selection was inherently opportunistic in light of the accessibility constraints; however, given the vastness of the park (4.5 Mha) it was possible to select and adequately sample a range of stand ages and predetermined stand types. Sixteen of 66 sites burned the year prior under variable burn severity, with a median site time since fire of 42 years that mirrors the short fire cycle in the area. Although we aimed to have a suitable diversity of stand ages, some gaps are present in the time since fire gradient due to periods of low regional fire activity or lack of access. A total of 32 sites were classified as wetlands (11 treed, 5 shrubby, and 16 open sedge). Of the wetlands, 5 were classified as bogs, 17 as fens, and the remaining 10 as floodplain or marsh wetlands. The remaining upland sites consisted of the following ecosites in order of increasing moisture status: dry pine lichen, pine mixedwood, aspen-dominated, and moisture upland conifer. It was not possible to obtain a perfectly balanced design among vegetation types because of the aforementioned accessibility constraints; however, the sampled vegetation types approximate their proportion in the landscape.

130 2.2 Field sampling

The design of the ground plot was modified from Lavoie et al. (2010). At each plot, a 40-m transect
132 was laid out with quadrats at 5-m spacing, with 8 quadrats per plot. Biomass sampling was carried out
by stratum: overstory trees, understory trees, tall shrubs, low live vegetation (<0.5 m in height), and
134 litter. Four overstory trees were measured for height and diameter at breast height (dbh) at each
quadrat using the point-centre quadrant technique. For the two stands measured in 2015, there was a
136 slight variation in the plot design where the quadrats were laid out in a grid at 7.5 m spacing rather than
along a single transect, changing only the layout of the quadrats but no other methods. Trees less than
138 3-cm dbh were measured in 3-m fixed-radius plots until 30 trees were measured or two plots were
completed, whichever came first. Four square clip plots (30×30 cm) captured live shrubs less than 50
140 cm in height, as well as all other live grasses, sedges, and herbaceous vegetation. Surface litter was
collected as dead plant material above the duff horizon with no signs of fungal or other decomposition.
142 The ratio of the accumulation of live sedges and grasses to the litter accumulation was calculated as a
proxy for the residence time of litter in these simple fuel beds where live vegetation and litter
144 intermingle; this comparison is not possible in forests due to the movement of litter from the canopy to
the forest floor. If taller shrubs were present, a 4-m width belt plot was constructed along the length of
146 the transect, with individual shrub volume estimated. Cube samples of foliage bulk density were taken
that incorporated foliage, as well as live and dead branch wood as a bulk sample (Ganteaume et al.
148 2013). Coarse woody debris was measured along one or two 25-m long plane-intercept transects using
the size classes from McRae et al. (1979). Duff and peat depth and density was measured in the field,
150 but not included in the sum of flammable biomass as duff consumption is largely limited to the
smouldering phase (Frandsen 1998).

Stand age was determined from several lines of evidence. Where available, stand age was measured by tree-ring analysis from a representative dominant tree. Fire scars were also used to date the last fire; when these were not present, we collected a disk from a dominant tree that established after the fire. For the 36 sites within a mapped historical fire, the age corresponding to the Canadian National Fire Database (Natural Resources Canada 2016) was used. Stand age established from previous studies (Larsen 1997) was also used if located nearby sample plots, and were used to date two sites. For the 18 sites without a known fire origin that also lacked a tree section, we used known age-height relations from Cieszewski et al (1993), which should be considered no more accurate than ± 10 years.

Sites were classified according to the Ecosites of Northern Alberta System (Beckingham and Archibald 1996), down to the ecosite level. This classification scheme adds more detail compared to classification by dominant tree species, as it considers understory vegetation, topographic position, and the moisture status of the site. Drainage class (xeric to hydric) was also assigned for each site. Ecosites were assigned descriptive names similar to the vegetation associations of Raup (1935). Grass and sedge floodplain sites within the Peace-Athabasca Delta did not conform to any ecosites within the framework, and were accordingly assigned an ecosite equivalent to an open fen.

2.3 Fuel loading

Fuel loads were calculated based on fuels available for flaming combustion only, and excludes other biomass not subject to flaming combustion such as large woody debris, duff, and peat (Johnston et al. 2015). Standing deciduous trees were not counted toward fuel load calculations, as boreal deciduous trees do not contribute towards flaming combustion following the FBP system, owing to the high

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moisture content and low surface area of the foliage. For trees larger than 3-cm dbh, foliar biomass was estimated using allometric equations based on height and dbh from Lambert et al. (2005). The fraction of flammable branchwood (<1 cm diameter) was estimated using the relationships from Alexander et al. (2004), which breaks down flammable vs total branchwood. For trees less than 3-cm dbh, allometric equations from Alexander et al. (2004) were used for both foliar and branch wood biomass, as the equations from Lambert et al. (2005) do not apply to such small trees. Since only black spruce (*Picea mariana*) and jack pine (*Pinus banksiana*) species allometric equations were available from Alexander et al. (2004), white spruce was assumed to follow the allometric equations of black spruce; deciduous understory trees (including larch (*Larix laricina*)) were measured but not added to the sum of fuel loading at a site. Total flammable biomass from the 32 measured overstory trees at each site was prorated to per square metre using the stem density computed from the point centred quarter method. Tall shrub canopy characteristics were added into the canopy fuel load calculation, but were excluded from canopy bulk density calculations.

2.4 Statistical Methods

To achieve enough replicates for each ecosite, some of the original ecosite classifications were merged for the purposes of statistical power. For each grouping of wildfire fuels (surface, canopy, and total) in both wetlands and uplands, the presence of a stepwise increase in the fuel load vs time since fire relationship was determined using the cross-entropy approach of Priyadarshana and Sofronov (2015) using a truncated (zero-bound) normal distribution in the *breakpoints* package in the *R* software version 3.2.2 (R Core Team 2016). If two breakpoints were found, a likelihood ratio test was used to determine if the two-breakpoint model significantly ($p < 0.05$) improved the model power, and a one breakpoint model was chosen if equally powerful as a two-breakpoint model. Any resultant breakpoints found

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were then tested for a difference in means using an ordinary t-test between age classes. Additionally, Spearman's correlations were used to determine if a monotonic linear change in fuel load over time was present. Ordinary one-way ANOVAs with fuel loading as the dependent variable were performed on untransformed data using the base *stats* package in *R*. Independent levels for the ANOVAs were the ecosite groups as well as age classes as determined by the breakpoints analysis above. Tukey's HSD post-hoc tests were used to determine which ecosite and drainage levels were significantly different in mean fuel loading from other levels.

3. Results

3.1 Fuel loading over time

Total fuel load available for surface fires (the sum of litter, live, shrubs, woody debris < 1.0 cm and understory tree pools) averaged $0.42 \pm 0.03 \text{ kg m}^{-2}$, and was not related to time since fire (Spearman's $\rho = 0.04$; $p = 0.69$). No breakpoint in the surface fuel load data was found when wetlands and uplands are considered together.

The three upland sites where there was evidence of a surface fire less than 30 years prior (the approximate age limit for the visual detection of prior surface fire) had a mean canopy fuel load of 0.92 kg m^{-2} , while 13 other sites with evidence of stand-replacing fire (as shown by no living trees established before the previous fire) had an average canopy fuel load of 0.16 kg m^{-2} (Figure 2), though small and uneven sample sizes do not allow for a direct comparison of means. Crown fuel load overall was related to time since fire using a Spearman's rank correlation test ($\rho = 0.518$; $p < 0.001$; Figure 2). No stepwise changes in canopy fuel load were observed using the cross-entropy test across the

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220 combination of wetlands and uplands.

222 Across both wetlands and uplands there was a strong time since fire effect on fuel accumulation ($\rho =$
498; $p < 0.001$). Though the breakpoint analysis did not reveal any stepwise changes in total fuel
224 loading across both wetlands and uplands, distinct increases in wetland and upland total fuel loads were
observed at approximately 95 and 97 years after fire, respectively (Table 1). Though it was not
226 detected by the breakpoint analysis, we observed that recent fires (those <10 years since fire) showed a
lower mean total fuel loading of 0.36 kg m^{-2} compared to older stands at 1.34 kg m^{-2} ($t_{59} = 4.5$; $p <$
228 0.001).

230 3.2. Wetland and upland fuels

Across all ages, mean litter load did not vary between wetlands ($0.24 \pm 0.04 \text{ kg m}^{-2}$; mean $\pm$ standard
232 error of the estimate of the mean) and uplands ($0.25 \pm 0.03 \text{ kg m}^{-2}$), using a Welch t -test ($t_{60} = -0.25$; p
 $= 0.81$). An ANOVA of ecosite exhibited large but marginally significant declines in litter found in
234 treed poor fens ($-0.25 \pm 0.15 \text{ kg m}^{-2}$; $t = 0.09$). Live surface fuels (shrubs, forbes, grass, and sedges)
were related solely to drainage class ($F_{8,56} = 2.86$, $p = 0.01$) but not ecosite. Live fuels were highest in
236 hydric and subhydric drainage classes that were commonly composed of sedges and graminoids (Table
2) with a mean of $0.16 \pm 0.03 \text{ kg m}^{-2}$. The lowest live surface fuel load was present in xeric and
238 subxeric systems ($0.03 \pm 0.01 \text{ kg m}^{-2}$), where live surface fuels (not including lichens) were limited to
sparse ericaceous shrubs. An ANOVA of flammable (<1 cm diameter) woody debris by ecosite showed
240 the mean load of $0.02 \pm 0.003 \text{ kg m}^{-2}$ increased to $0.07 \pm 0.02 \text{ kg m}^{-2}$ in wetland conifer upland ecosites
($t = 2.7$; $p = 0.007$).

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Fully 46 of 66 sampled sites had an understory tree fuel load (a component of the surface fuel bed given the low crown base height) of less than 0.01 kg m^{-2} . An ANOVA of understory conifer tree loading revealed a significant ecosite effect ($F_{7,58} = 4.78$; $p < 0.001$) that explained 36% of the total variance. A Tukey's HSD post-hoc test revealed that this difference was largely driven by very high understory fuel loading in the moist conifer uplands ecosites ($0.20 \pm 0.07 \text{ kg m}^{-2}$), which was dominated by a dense black spruce understory in five of six measured moist conifer upland ecosites, and white spruce in the other. Total fuel load available for surface fires (the sum of litter, live, woody debris $< 1.0 \text{ cm}$ and understory pools) was not significantly related to drainage class ($F_{8,57} = 0.96$, $p = 0.47$) or ecosite ($F_{7,58} = 1.03$, $p = 0.42$) with an overall mean of $0.42 \pm 0.03 \text{ kg m}^{-2}$ across all ecosites. Of the 66 sites, 33 were treed ($>10\%$ tree cover and height $>5 \text{ m}$), with a mean canopy fuel load of $1.44 \pm 0.26 \text{ kg m}^{-2}$. Within the treed sites, drainage class did not differ amongst drainage classes ($F_{6,26} = 1.24$; $p = 0.319$). The largest canopy fuel loads of 5.4 and 5.8 kg m^{-2} were found nearby in the Peace-Athabasca delta in a riparian white spruce stand, as well as in a dense closed-canopy willow floodplain. Total fuel load (including both surface and canopy fuels) averaged $0.85 \pm 0.20 \text{ kg m}^{-2}$ in wetlands, and $1.45 \pm 0.24 \text{ kg m}^{-2}$ across uplands. However, when the highest fuel loads found in wet upland conifer forests are excluded, there was no difference between the mean total fuel load in all other upland forests versus all wetlands, with the mean total fuel loading in uplands reduced to $1.15 \pm 0.23 \text{ kg m}^{-2}$ ($t_{71} = -1.17$; $p = 0.26$). Total fuel load in these wet upland conifer forests over 95 years of age was 4.5 kg m^{-2} , a nearly four-fold increase over the average fuel load in other upland forests (Table 2).

3.3. Observed fuel structure and the Canadian Fire Behaviour Prediction System

Mean surface fuel loading in open, treeless wetlands was $0.48 \pm 0.06 \text{ kg m}^{-2}$, 37% higher than the standard of 0.35 kg m^{-2} fuel loading for grass fuel (O-1) following the FBP System (Wotton et al.

2009). While the FBP System assumes low grass curing in mid-summer conditions, we observed a significant litter bed of dead grass and sedge fuel load present amongst live grasses, which lowered the effective moisture content (increased curing) of the fuel bed. The mean ratio of live to litter fuel loading in wetland sites did not differ across wetlands ecosite classes (ANOVA; $F_{3,20} = 0.396$; $p = 0.757$; Figure 3), with a mean of 4 years' worth of litter accumulation found in wetlands (counting only sites greater than six years since fire). This significant litter accumulation was often well-aerated and upwards of 30 cm in height, and would effectively shift the cured grass portion of the fuel bed to as high as 80%, with four parts litter accumulation and one part live growth.

Canopy fuel loads greater than the 1.15 kg m^{-2} from the standard C-3 mature jack pine fuel type from the FBP System were found in 11 of 21 upland stands (Table 1), and were observed in sites as young as 31 years since fire. Only two forested wetlands exceeded the 0.8 kg m^{-2} of canopy fuel loading from the standard boreal spruce C-2 fuel type (Figure 4). Only 26 of 33 sites with a tree canopy exceeded the approximate minimum crown bulk density of 0.1 kg m^{-3} required to sustain a crown fire (Cruz et al. 2005). Sites with a conifer basal area of less than $10 \text{ m}^2 \text{ ha}^{-1}$ in 8 of 9 cases did not have sufficient canopy bulk density to sustain a crown fire, while sites with greater than $10 \text{ m}^2 \text{ ha}^{-1}$ did in every case. For forested wetlands (excluding those dominated by tall shrubs), a rise in crown base height was observed alongside increasing canopy fuel load (Figure 4). Crown base height in uplands showed no pattern, and varied widely from less than 3 m to over 17 m (Figure 4).

4. Discussion

4.1 A changing boreal fuels mosaic

The results of this study provide further support for the claims of an effect of stand age on wildfires in

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the boreal forest. They reaffirm what we know about fuel accumulation in upland boreal forests: rates of accumulation are highly variable, but do, on average take a few decades to reach pre-fire levels. Despite the variability in fuel loads among forest stands, very high levels of biomass are only observed after about 20 years following a fire, which is consistent with the results of Héon et al. (2014) in eastern Canada. Our results suggest that this age dependence, however, only holds true for upland forests in our study area. Fire–vegetation relationships of the lowlands add complexity to landscape dynamics in that, when largely treeless, they have a surface-fire regime that can re-burn only a few years after a fire. This is particularly consequential given the large fraction of the landscape (up to half) covered by wetlands in our study area of Wood Buffalo National Park; as such, recent burns may—on the whole—not provide the degree of resistance to subsequent burning observed elsewhere in the boreal forest (Bernier et al. 2016; Erni et al. 2016).

Wetland fuel load and structure are for the first time documented and contrasted with those of upland forests. While fuel loads in these abundant boreal wetlands are often comparable to uplands, site-level differences such as fuel moisture, phenology, and access to groundwater are factors that likely contribute to their widespread burning during periods of profound drought. We found the highest fuel loads in the wet upland conifer ecosite (Table 2) that often lies in the transition zone between wetlands and drier forests that are either deciduous-dominated or else more sparsely treed. In general, fire may be possible in upland forests after only several days of fuel drying, but this is not the case in wetlands, which have a much more restricted window of burning that often negates their availability of flammable biomass. However, during drought conditions some studies report that wetlands are more likely to contribute to burn area (Turetsky et al. 2004; Turetsky et al. 2015). Indeed, the rapid recovery of surface fuels over organic wetland soils documented here may allow for the re-burn of wetland fuels in nearly consecutive years as deeper organic horizons are exposed by a lowered water table. The

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314 extent to which the rapid accumulation of flammable biomass is counteracted by the narrowness of the
seasonal burn window in explaining fire activity in wetland ecosystem warrants further study.

316
The variability in flammable fuels observed in both upland and lowland ecosystems reported in this
318 study highlight the extreme variability that exists in our study area and, as such, caution against
drawing steadfast conclusions regarding fire-fuels dynamics. Patterns observed may nevertheless be
320 instructive as to the limits under which the recent burns and wetlands are able to resist subsequent
burning. Our results show that the burning of wetlands dominated by graminoids, which is highly
322 dependent of drought recurrence, is not completely independent of fuel succession when these have a
tree cover—that is, they can behave as an intermediate between wetland and upland sites. In particular,
324 the potential of high-frequency fire in graminoid-dominated wetlands contrasts strongly with treed and
forested wetlands with a *Sphagnum* moss groundcover, where any significant fire spread is only
326 possible 80 years after fire (Johnston et al. 2015). It should be noted that these ecosystems are viewed
with increasing interest in the Boreal Plain of western Canada because they are projected to become an
328 important refugium for conifers under climate change in the region (Schneider et al. 2015).

330 An interesting, if somewhat anecdotal, aspect of the study area is the commonness of surface fires
(Figure 2). Although surface fires have been reported as frequent in other parts of the boreal forest
332 (Heinselman 1973), the conceptual fire-regime model for the boreal forest stipulates that the clear
majority of the area burned by wildfires is through stand-renewing crown fires (Johnson 1996). Surface
334 fires in the study area occur almost exclusively in pine-dominated stands. These have low surface fuels
and vegetation cover, and high (several meters) crown fuel base, factors that contribute to limiting
336 crown involvement. While this remains to be fully assessed, we speculate that stands that repeatedly
burn as surface fires do so because of the high frequency of fires—one of the highest of the North

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American boreal forest—in this area. The phenomenon may thus be self-reinforcing, as frequent fires further reduce the potential for subsequent crown fires. Our results are coherent with those of other studies that suggest that frequent surface fires may lead to a fuel-accumulation trajectory that is highly divergent from that of stands originating from stand-renewing fires (Miquelajauregui et al. 2016).

From the results of the upland fuel loading observations, white spruce is present in almost all sites showing high fuel loads ($>4 \text{ kg m}^{-2}$) in this region. White spruce is generally considered to be a climax species that is indicative of long periods since fire, as this shade-tolerant species may succeed after pine on subxeric sites (Purchase and La Roi 1982) and in aspen forest on mesic sites (Rowe 1961), but is considered intolerant to fire disturbance with poor post-fire recruitment (Purdy et al 2002; Johnstone and Chapi 2004). Arguably, these pockets of high fuel load may exist as fire refugia (Raup 1935; Madoui 2011); in this study the mature white spruce stands were found only in areas such as upland islands in the middle of rich fens with low fuel load, in the lee of lakes, and other areas that may provide shelter from advancing wildfire (Larsen 1997).

4.2 Importance of fuel loading

In this study, we only used surface and canopy fuels capable of flaming consumption, which discounts larger diameter coarse woody debris, as well as duff. Though duff and its moisture content is a vital predictor of lightning fire ignition in boreal Canada (Wotton and Martell 2005), organic soils typically combust in the smouldering phase at rates on the order of 1 cm h^{-1} (Prat et al. 2015) and therefore are unlikely to significantly contribute to the flaming head fire intensity. Experimental burning that measures the depth of burn immediately after the passage of the fire front and before significant smouldering typically records around 1 kg m^{-2} of surface fuel consumption, while data taken from

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paired burned-unburned plots weeks after wildfire shows consumption values closer to 3 kg m⁻² (de
362 Groot et al. 2009). Following Byram's formulation for fireline intensity (Byram 1959), fuel loading is
an independent input to fire intensity alongside a fire's rate of spread and the heat of combustion of the
364 fuels. In this sense, fuel loading as represented here is a proxy for the maximum potential intensity for
a given rate of spread, fuel structure, and moisture condition of a fuel bed. Fuel structure in the canopy,
366 in turn, is a critical component of the initiation of crown fire in boreal forests (Van Wagner 1977).
Since crown fire initiation is based on critical surface fire intensity, an increase in the fuel loading at
368 the surface can increase fire intensity beyond the threshold for heating of canopy fuels. Indeed, in the
analysis of crown fire initiation probability, Cruz et al. (2004) found that surface fuel consumption
370 played a direct role in determining the probability of crown fire initiation. At a larger scale, Cumming
(2001) as well as Krawchuk et al. (2006) found significant differences in lightning fire probability
372 between even black and white spruce stands, suggesting this role of canopy structure and fuel load
extends to patterns of lightning fire initiation.

374 While we have discounted deciduous overstory trees from fuel loading calculations, deciduous shrubs,
376 sedges, and graminoids have been included in fuel loading in this study. Our measurements of shrub
fuel loading are inclusive of both live foliage, as well as live and dead branch wood within the shrub
378 canopy. From our measurements of the bulk moisture content within the shrub canopy, we observed
that up to half of the branch wood in the wetland shrub canopy (primarily *Salix spp.* with some *Betula*
380 *spp.*) is composed of dead branch wood that is at equilibrium moisture content with the atmosphere,
which during wildfire-prone periods is 5–10% gravimetric moisture content (Amiro et al. 2005). In the
382 context of live deciduous leaves measured in this study at 150–200% gravimetric moisture content, this
results in a canopy moisture content more on the order of 30–80% gravimetric moisture content, which
384 is lower than or similar to conifer overstories. Similarly, the high litter load relative to live green

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biomass in the sedge and graminoid-dominated wetlands means that even in mid-summer that the
386 system can sustain at least a low-intensity fire if the litter fuels are sufficiently dry. This contrasts with
sub-tropical to semi-arid grasslands, where litter does not generally persist over a year before
388 decomposing, and therefore generally no more than a year's worth of growth is present in the litter
layer (Cheney and Sullivan 2008).

4.3 Relevance to fire behaviour models

392 A relative invariance of surface fuel loading was observed across a large range of ecosites and time
since fire. While the total surface fuel load did not systematically vary across sites, the origin of the
394 surface fuels varied across sites. Xeric sites such as the dry pine-lichen shown here featured high litter
loads of lichens and needle litter, while mesic upland sites and forested wetlands featured higher
396 densities of small conifer trees as sources of surface fuel loading. In open wetlands, sedge and
graminoid litter was the main source of surface fuels, in addition to short shrubs <0.5 m tall.

398
In addition to the effect on fire intensity, canopy fuel loading has a significant effect on the surface
400 microclimate for litter and duff fuels, an effect observed by Wotton and Beverly (2007). Low light
levels at the forest floor promote the expansion and growth of feathermoss at the expense of understory
402 trees, herbs, and shrubs (Bisbee et al. 2001). Despite the lower moisture content of feathermosses
compared to forest floor litter layers (Wilmore 2001), lower surface wind speeds (Massman 1997)
404 caused by high wind attenuation in the canopy may somewhat ameliorate this effect. Future
improvements to fire behaviour models in Canada would benefit from being adaptive to the large
406 variation in canopy fuel load and structure.

5. Conclusions

Wildfire fuel loading was assessed across 66 plots following a gradient of drainage and time since fire. By far the highest fuel loads were observed in moist white spruce upland forests where fuel loading exceeded 5 kg m^{-2} at times, fully four times greater than the standard boreal spruce fuel model (Forestry Canada Fire Danger Group 1992). Notwithstanding densely stocked spruce stands, total fuel loading across the rest of the range of ecosites and ages varied relatively little between 0.8 and 1.2 kg m^{-2} , and fell within the range of standard fuel models in the Canadian Fire Behaviour Prediction System. Wetlands, inclusive of treed, shrub, and sedge-dominated, showed no substantial difference in total fuel loading compared to most uplands owing to substantial accumulations of litter and small conifers. The fuel load observations made here are evidence of wetlands in this region having largely similar fuel loads as uplands, if all the wetland fuels are considered available for flaming combustion. The quantification of wetland fuels presented here made without regard to the moisture status of the fuels and represent maximum fuel availability during drought. The rate and patterns of landscape fuel availability during periods of increasing is poorly parametrized in current fire behaviour and fuel consumption models.

The measurements are representative of the current climate, and do not attempt to normalize post-fire fuel loading as a response to variable fire severity. With a more severe fire regime being a likely future for the boreal forest (Boulanger et al. 2013), the feedback of increased fire severity on post-fire fuel accumulation and recovery remains to be tested under a new climate. In the dry conifer forests of the southwestern USA where climate change is almost certain to reduce productivity due to water limitation, increased fire frequency in a future warmer climate may create a strong fire-fuels negative feedback, with less fuels on the landscape as a result (Diggins et al. 2010). Future ecosystem

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trajectories in this wetland-abundant region are less certain (Schneider et al. 2015), as wetlands are likely to show a lagged vegetation response to climate change. In our study area, although many vegetation types appear to exhibit a rapid recovery of fuels following wildfire, it is still uncertain to what extent variable fuel recovery rates limit subsequent wildfire activity, as reported by Héon et al. (2014). A recent broad-scale study (Price et al. 2015) in Alberta did not observe such a dependence, but this may be due to the broad-scaled nature of the analysis, as well as a strong anthropogenic effect on fire activity, which is largely absent from our study area. While this study examines the accumulation of wildfire fuels as a function of time since fire, future research is required to better understand fire behaviour and controls on post-fire vegetation succession in the Boreal Plains, particularly under extreme weather conditions.

6. Acknowledgements

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602 List of Tables

604 Table 1. Breakpoint statistics showing the time since fire where a marked increase in mean surface,
canopy, and total fuel load was observed. Mean ($\bar{x}$) and standard deviation (s) before ($\bar{x}_1$) and after
($\bar{x}_2$) the breakpoint is shown, as is a Welch's t -test for differences in means between the two samples.
606 Comparison with Fire Behaviour Prediction (FBP) system fuel types are also shown. Note that FBP
surface fuel load estimates include duff consumption in the flaming phase, which was not considered
608 in the wetland and upland fuel load measurement performed in this study.

610 Table 2. Summary of fuel loading by ecosite and age class. WD represents woody debris with
diameter less than 1 cm.

612

List of Figures

614 Figure 1. Map of Wood Buffalo National Park, showing wetland and upland sampling locations. The
extent of sampled fires with mapped perimeters is also shown.

616 Figure 2. Fuel accumulation in canopy and surface fuels as a function of time since fire. The time axis
is transformed by the natural log for clarity. For reference, the canopy fuel loads in C-2 and C-3 fuels
618 are 0.8 and 1.15 kg m⁻². Sites with evidence of recent surface fire are denoted with an asterisk.

Figure 3. The ratio of measured surface litter to live surface fuel load in open wetlands. This sum is
620 inclusive of sedges, grasses, and short shrubs <1.5 m in height.

Figure 4. The relationship between canopy fuel load and crown base height of conifers (calculated as
622 the Lorey crown base height). Crown base height and bulk density standard values for C-2 (boreal

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spruce) and C-3 (mature pine) fuel types are shown as dashed lines.

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Sheet1

Wetland fuel load (kg m ⁻²)									
	Breakpoint (years)	<i>x</i> ₁	<i>s</i> ₁	<i>x</i> ₂	<i>s</i> ₂	<i>t</i> _{1,2}	<i>p</i> _{1,2}	Grassland O. 1	Boreal spruce C-2
Surface	78	0.39	0.27	0.50	0.22	-1.29	0.21	0.35	4.00
Canopy	85	0.11	0.39	1.21	1.83	-1.71	0.12	0.00	0.80
Total	95	0.56	0.46	1.75	1.82	-1.63	0.15	0.35	4.80

Upland fuel load (kg m ⁻²)										
	Breakpoint (years)	<i>x</i> ₁	<i>s</i> ₁	<i>x</i> ₂	<i>s</i> ₂	<i>t</i> _{1,2}	<i>p</i> _{1,2}	Boreal spruce C-2	Mature pine C-3	Immature pine C-4
Surface	NA	0.44	0.25	0.37	0.22	1.19	-0.24	4.00	4.50	4.50
Canopy	104	0.50	0.63	2.23	1.87	-2.86	0.02	0.80	1.15	1.20
Total	97	0.95	0.66	2.45	1.90	-2.59	0.02	4.80	5.65	5.70

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Sheet1

Time Since Fire	Ecosite	Ecosite	<i>n</i>	Litter	Wood Debris	Live	Shrub	Under kg m ⁻²	Σ Surface	Σ Crown	Σ Fuel
Uplands											
1-95	Dry pine-lichen	a	7	0.31	0.03	0.03	0.00	0.10	0.48	0.73	1.21
95-128		a	1	0.10	0.14	0.02	0.00	0.00	0.26	1.52	1.78
1-95	Pine mixedwood	b	5	0.24	0.03	0.11	0.00	0.01	0.39	1.09	1.48
95-146		b	3	0.23	0.02	0.05	0.00	0.01	0.31	0.94	1.25
1-95	Mesic upland	d	5	0.18	0.03	0.12	0.00	0.00	0.34	0.91	1.25
95-250		d	3	0.27	0.03	0.07	0.00	0.00	0.38	1.63	2.01
1-95	Moist upland conifer	g	3	0.29	0.06	0.07	0.00	0.23	0.65	0.35	1.01
95-155		g	4	0.21	0.04	0.06	0.00	0.13	0.43	4.11	4.54
Wetlands											
1-97	Bog	i	7	0.16	0.02	0.14	0.00	0.08	0.39	0.09	0.48
1-97	Treed fen	j	4	0.08	0.00	0.10	0.02	0.03	0.23	0.59	0.81
97-113		j	1	0.06	0.04	0.10	0.00	0.02	0.22	0.55	0.77
1-97	Open fen	k	11	0.35	0.00	0.18	0.03	0.00	0.56	0.00	0.56
97-155		k	5	0.29	0.02	0.15	1.62	0.01	0.50	0.00	0.53
1-97	Saline wet meadow	l	4	0.18	0.00	0.20	0.00	0.00	0.38	0.00	0.38
97-250		l	1	0.21	0.05	0.16	0.00	0.05	0.47	1.61	2.08

Map of Wood Buffalo National Park, showing wetland and upland sampling locations. The extent of sampled fires with mapped perimeters is also shown.

125x175mm (300 x 300 DPI)

Fuel accumulation in canopy and surface fuels as a function of time since fire. The time axis is transformed by the natural log for clarity. For reference, the canopy fuel loads in C-2 and C-3 fuels are 0.8 and 1.15 kg m⁻². Sites with evidence of recent surface fire are denoted with an asterisk.

124x105mm (300 x 300 DPI)

The ratio of measured surface litter to live surface fuel load in open wetlands. This sum is inclusive of sedges, grasses, and short shrubs <1.5 m in height.

87x58mm (300 x 300 DPI)

The relationship between canopy fuel load and crown base height of conifers (calculated as the Lorey crown base height). Crown base height and bulk density standard values for C-2 (boreal spruce) and C-3 (mature pine) fuel types are shown as dashed lines.

159x160mm (300 x 300 DPI)

S1

Site	Lat	Long	Time Since Fire	Drainage	Ecosite	Litter	Wood Debris	Live	Shrub	Under kg m ⁻²	Σ Surface	Σ Crown	Σ Fuel
14-07	59.876	-112.471	1*	subxeric	a	0.34	0.01	0.10	0.00	0.01	0.46	0.94	0.46
14-08	59.904	-112.486	1	subxeric	a	0.27	0.01	0.00	0.00	0.00	0.28	0.00	0.28
13-01	59.711	-112.748	10	xeric	a	0.16	0.02	0.02	0.00	0.01	0.21	0.00	0.21
13-16	59.126	-112.055	14	xeric	a	0.39	0.04	0.08	0.00	0.40	0.91	0.00	1.31
12-03	60.045	-112.820	31	subxeric	a	0.13	0.07	0.00	0.00	0.30	0.50	0.95	1.44
12-05	59.244	-112.401	31	subxeric	a	0.55	0.03	0.00	0.00	0.01	0.59	2.08	2.67
13-02	59.711	-112.745	42	subxeric	a	0.34	0.04	0.03	0.00	0.00	0.41	1.17	1.58
13-10	59.826	-112.863	124 ¹	subxeric	a	0.10	0.14	0.02	0.00	0.00	0.26	1.52	1.78
12-01	59.743	-112.240	128	subxeric	a	0.11	0.01	0.00	0.00	0.01	0.13	1.81	1.94
14-10	60.343	-114.169	1	submesic	b	0.65	0.00	0.07	0.00	0.03	0.75	0.00	0.75
14-19	59.910	-112.484	1*	subxeric	b	0.33	0.00	0.01	0.00	0.00	0.34	1.42	0.34
13-26	60.132	-113.400	2	subxeric	b	0.07	0.05	0.11	0.00	0.00	0.23	0.00	0.23
14-15	59.946	-112.441	2*	submesic	b	0.30	0.07	0.36	0.00	0.00	0.73	0.63	0.73
13-23	59.349	-112.479	32	subxeric	b	0.30	0.01	0.04	0.00	0.00	0.35	0.83	0.35
12-11	59.801	-112.009	59	mesic	b	0.21	0.04	0.02	0.00	0.04	0.31	2.57	2.87
14-12	60.336	-114.158	104	submesic	b	0.08	0.02	0.03	0.00	0.03	0.16	0.89	1.07
14-16	59.947	-112.448	145 ²	submesic	b	0.20	0.02	0.06	0.00	0.00	0.28	1.11	1.05
14-11	60.316	-114.126	146	submesic	b	0.41	0.02	0.06	0.00	0.01	0.50	0.82	1.34
13-17	59.976	-112.469	1	mesic	d	0.11	0.05	0.01	0.00	0.00	0.17	0.00	0.17
14-04	59.352	-113.713	2	mesic	d	0.30	0.02	0.03	0.00	0.00	0.35	0.01	0.35
12-02	60.045	-112.820	31	mesic	d	0.17	0.01	0.00	0.00	0.00	0.18	1.47	1.66
14-21	60.059	-113.282	57 ³	mesic	d	0.21	0.03	0.04	0.00	0.00	0.28	1.73	0.28
13-21	59.947	-112.490	67	mesic	d	0.13	0.03	0.54	0.01	0.00	0.71	1.33	1.16
13-12	59.834	-112.945	124	mesic	d	0.38	0.04	0.06	0.00	0.00	0.48	1.21	0.48
13-15	59.101	-111.889	145 ³	mesic	d	0.28	0.02	0.05	0.00	0.00	0.35	1.51	0.43
14-05	59.358	-113.718	250 ⁴	mesic	d	0.15	0.04	0.11	0.00	0.00	0.30	2.17	2.13
13-29	58.843	-112.006	155 ⁴	mesic	e	0.72	0.04	0.02	0.00	0.00	0.78	6.04	6.20
12-13	59.786	-112.601	105 ⁴	hygric	f	0.04	0.03	0.02	0.00	0.00	0.09	4.37	4.20
14-14	60.034	-113.116	2	subhygric	g	0.55	0.06	0.06	0.00	0.40	1.07	0.00	1.07
13-25	60.139	-113.660	9	subhygric	g	0.14	0.09	0.07	0.00	0.00	0.30	0.00	0.30
14-13	60.035	-113.105	90	subhygric	g	0.19	0.04	0.08	0.00	0.28	0.59	1.06	1.91
13-03	59.528	-114.559	113 ⁵	subhygric	g	0.00	0.04	0.04	0.00	0.19	0.26	4.85	5.30

S1

13-13	59.835	-112.943	124 ¹	subhygric	h	0.08	0.03	0.14	0.00	0.33	0.58	1.18	2.09
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*most recent fire at site was a surface fire

¹age from nearby site 13-12 used

²used Cieszewski et al (1993) age-height curves with 12-02 as reference

³used Cieszewski et al (1993) age-height curves with 13-21 as reference

⁴used Cieszewski et al (1993) age-height curves from a reference site at 119 years of age and 16.9m stand height, Whitman et al (in prep) site 15s-NT-24

⁵stand age from Larsen (1994)

⁶used Cieszewski et al (1993) age-height curves and stand ages and heights from Johnston et al. (2015) as reference

⁷stand origin of 1941 from Wallenius et al (2011) site 60

⁸same 1941 fire origin as site 13-05 assumed

Draft

S1

Draft

S1

Draft

S1

Draft

S1

Draft

S1

SD_live_ha	SmTreeAllLiveDensity_ha
379	236
0	0
0	2122
0	154203
6127	7427
3427	1627
2603	0
2924	354
850	1061
0	37136
1989	0
0	0
440	0
3435	0
2253	884
901	236
590	6897
569	1533
0	0
98	3890
4466	472
1404	0
1120	0
2384	0
607	2358
547	1650
1498	1297
1439	619
316	67906
0	12909
4329	21928
6631	17330

S1

2572

23696

SD_live_ha SmTreeAllLiveDensity_ha

0 354

0 0

0 9726

0 13086

225 10257

1578 5836

1249 4598

248 0

4539 16033

727 1650

0 0

0 0

0 0

0 0

0 0

0 0

0 0

29 472

0 1650

0 0

0 0

0 0

0 0

0 2829

0 0

0 0

0 0

S1

0	0
0	0
2293	6013

Draft

S1

SmTreeConiLiveDensity_ha	SmTreeLiveBA_m2_ha	LrgTreeLiveBA_m2_ha	TotalLiveBA	ConiBA	LiveLoreyH	LiveLoreyCBH	LiveConifer_CBD
236	0	14.4	14.4	14.4	24.38	18.76	0.17
0	0	0	0	0			
2122	0.2	0	0.2	0			
154203	10.7	0	10.7	0			
7427	2.2	10.1	12.3	12.3	6.49	4.45	0.47
849	0.3	31.1	31.4	31	13.31	5.63	0.27
0	0	14.9	14.9	14.9	12.57	6.68	0.2
118	0	19.6	19.6	19.6	13.62	6.13	0.2
1061	0.1	28.1	28.3	28.3	18.58	10.12	0.21
26526	2.6	0	2.6	0			
0	0	20.1	20.1	20.1	14.36	6.59	0.18
0	0	0	0	0			
0	0	8.8	8.8	5.6	19.86	13.32	0.06
0	0	15.3	15.3	0	10.75	6.12	0
884	0.5	27.3	27.8	15.8	13.02	7.74	0.28
236	0	13.5	13.5	13.5	16.55	8.94	0.12
2476	1.7	17.2	18.8	13.1	20.97	13.02	0.1
1533	0.2	12.9	13.1	13.1	18.37	9.92	0.1
0	0	0	0	0			
1415	0.1	0.1	0.2	0	20.97	16.71	0
236	0.1	22.1	22.2	14.3	10.36	5.49	0.2
0	0	24.9	24.9	0	18	13.27	0
0	0	17	17	5.6	17.63	10.95	0.06
0	0	20.9	20.9	0	12.35	8.19	0
354	0.4	19.4	19.8	1	23.43	15.18	0.01
472	0.4	21.7	22.1	18.6	24.89	11.52	0.14
707	0.1	54.4	54.5	48.9	20.89	8.71	0.44
619	0	35.4	35.4	33.3	16.8	9.49	0.56
67906	7.1	0	7.1	6.2	3.36	0.05	0
8665	0.2	0	0.2	0			
21221	4.6	7.4	12	11.8	5.21	2.17	0.34
14147	4.2	23.1	27.2	27.2	6.58	2.15	1.1

S1							
22282	5.8	8	13.7	13.7	6.85	2.23	0.26
SmTreeConiLiveDensity_ha	SmTreeLiveBA_m2_ha	LrgTreeLiveBA_m2_ha	TotalLiveBA	ConiBA	LiveLoreyH	LiveLoreyCBH	LiveConifer_CBD
354	0.1	0	0.1	0			
0	0	0	0	0			
9726	0	0	0	0			
12025	0.2	0	0.2	0			
10257	1.8	0.2	2	2	3.65	0.07	0.01
5836	1.4	2.4	3.8	3.8	3.87	1.29	0.23
3183	1	1.6	2.6	2.3	4.17	0.63	0.06
0	0	0.6	0.6	0.6	4.82	0.57	0.03
8488	1.5	12.4	13.9	13.1	6.75	5.06	1.12
1650	0.3	2.8	3.1	3.1	6.14	1.28	0.11
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			
472	0.1	0	0.1	0.1	3.07	0.74	0
1061	0.5	0	0.5	0			
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			
2829	0.6	0	0.6	0			
0	0	0	0	0			
0	0	0	0	0			
0	0	0	0	0			

S1							
0	0	0	0	0			
0	0	0	0	0			
6013	0.8	11	11.8	11.8	9.64	3.16	0.25

Draft

S2

Site	Lat	Long	Time Since Fire	Vegetation Class	Drainage	Ecosite
13-19	59.973	-112.472	1*	Treed peatland	subhydryc	i
15-02	59.780	-112.350	1	Treed peatland	subhydryc	i
13-07	59.383	-113.367	6	Open peatland	subhydryc	i
13-08	59.382	-113.368	6	Treed peatland	hydryc	i
14-17	60.195	-113.751	10	Treed peatland	subhydryc	i
13-09	59.382	-113.374	83 ⁶	Shrub peatland	subhydryc	i
14-03	60.138	-113.657	86	Treed peatland	subhydryc	i
15-13	60.270	-113.973	1	Treed peatland	subhydryc	j
14-06	60.048	-112.823	31	ruce dominated mix	subhydryc	j
13-20	59.945	-112.491	32*	Treed peatland	hydryc	j
14-20	60.195	-113.753	69	Treed peatland	subhydryc	j
13-04	59.527	-114.561	113 ⁵	Treed peatland	hydryc	j
13-18	59.975	-112.471	1	Sedge marsh	hydryc	k
12-10	58.607	-111.691	2	Wet meadow	subhydryc	k
12-06	58.627	-111.679	8	Semi-open prairie	subhydryc	k
12-07	58.628	-111.677	8	Semi-open prairie	subhydryc	k
12-08	58.630	-111.672	8	Sedge marsh	hydryc	k
12-09	58.630	-111.673	8	Wet meadow	subhydryc	k
13-14	59.028	-111.874	14	nally submerged m	hydryc	k
13-06	59.436	-112.362	32	Shrub peatland	subhydryc	k
13-22	59.280	-112.473	32	Shrub peatland	subhydryc	k
12-04	59.515	-111.459	64	nally submerged m	subhydryc	k
13-24	59.689	-111.530	72	Sedge marsh	hydryc	k
12-12	59.786	-112.602	105 ⁴	Rich fen	subhydryc	k
13-11	59.828	-112.864	124 ¹	Shrub peatland	hydryc	k
13-27	58.841	-111.991	155 ⁴	Semi-open prairie	subhydryc	k
13-28	58.858	-111.992	155 ⁴	Willow-alder thickets	subhydryc	k
13-30	58.851	-111.992	155 ⁴	Willow-alder thickets	hydryc	k
14-18	58.839	-111.978	1	Wet meadow	hydryc	l
14-01	59.812	-111.967	61	ilt flats saline mead	subhydryc	l
13-05	59.961	-112.363	72 ⁷	ilt flats saline mead	subhydryc	l
14-09	59.551	-111.468	73 ⁸	Wet meadow	hydryc	l
14-02	59.359	-113.717	250 ⁴	Treed peatland	subhydryc	l

S2						
Litter	Wood	Live	Shrub	Under kg m-2	Σ Surface	Σ Crown
0.08	0.00	0.03	0.00	0.25	0.36	0.00
0.08	0.04	0.08	0.00	0.00	0.20	0.00
0.07	0.00	0.10	0.00	0.00	0.17	0.00
0.20	0.05	0.19	0.00	0.08	0.52	0.00
	0.00		0.00	0.01	0.01	0.00
0.11	0.00	0.52	0.00	0.12	0.75	0.04
0.56	0.03	0.05	0.00	0.10	0.74	0.60
0.04	0.01	0.01	0.00	0.00	0.05	0.00
0.10	0.00	0.02	0.06	0.05	0.22	0.24
0.13	0.00	0.29	0.01	0.00	0.43	0.11
0.06	0.00	0.09	0.00	0.05	0.20	2.00
0.06	0.04	0.10	0.00	0.02	0.22	0.55
0.00	0.00	0.25	0.01	0.00	0.26	0.00
0.29	0.00	0.43	0.00	0.00	0.72	0.00
0.63	0.00	0.11	0.00	0.00	0.74	0.00
0.63	0.00	0.10	0.00	0.00	0.73	0.00
0.44	0.00	0.02	0.00	0.00	0.46	0.00
0.36	0.00	0.20	0.00	0.00	0.56	0.00
0.08	0.00	0.25	0.00	0.00	0.33	0.00
0.14	0.00	0.03	0.00	0.00	0.17	0.00
0.24	0.00	0.08	0.23	0.01	0.55	0.00
0.22	0.00	0.15	0.00	0.02	0.39	0.00
0.84	0.00	0.35	0.10	0.00	1.29	0.00
0.07	0.00	0.00	0.00	0.00	0.07	0.00
0.22	0.03	0.30	0.16	0.00	0.71	0.00
0.49	0.00	0.23	0.00	0.00	0.72	0.00
0.32	0.06	0.04	5.87	0.05	0.47	0.00
0.36	0.00	0.17	2.06	0.00	0.53	0.00
0.06	0.00	0.47	0.00	0.00	0.53	0.00
0.09	0.00	0.10	0.00	0.00	0.19	0.00
0.40	0.00	0.13	0.00	0.00	0.53	0.00
0.17	0.01	0.09	0.00	0.00	0.27	0.00
0.21	0.05	0.16	0.00	0.05	0.47	1.61

S2

Σ Fuel

- 0.36
- 0.20
- 0.17
- 0.52
- 0.02
- 0.90
- 1.44
- 0.05
- 0.54
- 0.56
- 2.13
- 0.79
- 0.26
- 0.72
- 0.74
- 0.73
- 0.46
- 0.56
- 0.33
- 0.17
- 0.79
- 0.39
- 1.39
- 0.07
- 0.86
- 0.72
- 6.39
- 2.59
- 0.53
- 0.19
- 0.53
- 0.27
- 2.12

Draft

S3

Site	CWD < 7cm (kg m ⁻²)	CWDTotal kg m ⁻²	Stems > 7 cm (stems ha ⁻¹)	Stems < 7 cm (stems ha ⁻¹)	Total Live Basal Area (m ² ha ⁻¹)	Conifer Live Basal Area (m ² ha ⁻¹)
14-07	0.09	0.09	379	236	14.4	14.4
14-08	0.13	0.13	0	0	0	0
13-01	1.17	1.17	0	2122	0.2	0
12-03	0.08	0.08	6127	7427	12.3	12.3
12-05	0.05	0.05	3427	1627	31.4	31
13-16	1.58	5.24	0	154203	10.7	0
13-02	0.45	0.45	2603	0	14.9	14.9
13-10	0.66	0.66	2924	354	19.6	19.6
12-01	0.31	1.29	850	1061	28.3	28.3
14-10	0.6	0.6	0	37136	2.6	0
14-19	0.91	0.91	1989	0	20.1	20.1
13-26	0.26	0.26	0	0	0	0
14-15	0.37	1.16	440	0	8.8	5.6
13-23	0.59	0.88	3435	0	15.3	0
12-11	0.23	0.23	2253	884	27.8	15.8
14-12	0.52	0.52	901	236	13.5	13.5
14-16	0.17	0.29	590	6897	18.8	13.1
14-11	0.16	0.16	569	1533	13.1	13.1
13-17	0.89	1.97	0	0	0	0
14-04	0.36	2.36	98	3890	0.2	0
12-02	0.68	10.03	4466	472	22.2	14.3
14-21	1.01	1.14	1404	0	24.9	0
13-21	0.63	0.73	1120	0	17	5.6
13-12	0.51	0.51	2384	0	20.9	0
13-15	0.02	3.08	607	2358	19.8	1
14-05	0.37	1.51	547	1650	22.1	18.6
13-29	0.77	2.59	1498	1297	54.5	48.9
12-13	0.16	1.92	1439	619	35.4	33.3
14-14	1.28	1.4	316	67906	7.1	6.2
13-25	1.81	4.43	0	12909	0.2	0
14-13	1.01	1.11	4329	21928	12	11.8
13-03	0.18	0.18	6631	17330	27.2	27.2
13-13	0.11	0.11	2572	23696	13.7	13.7

S3

Live Lorey Height (m)	Live Lorey Crown Base Height (m)	Live Conifer Crown Bulk Density (kg m ⁻³)
24.38	18.76	0.17
6.49	4.45	0.47
13.31	5.63	0.27
12.57	6.68	0.2
13.62	6.13	0.2
18.58	10.12	0.21
14.36	6.59	0.18
19.86	13.32	0.06
10.75	6.12	0
13.02	7.74	0.28
16.55	8.94	0.12
20.97	13.02	0.1
18.37	9.92	0.1
20.97	16.71	0
10.36	5.49	0.2
18	13.27	0
17.63	10.95	0.06
12.35	8.19	0
23.43	15.18	0.01
24.89	11.52	0.14
20.89	8.71	0.44
16.8	9.49	0.56
3.36	0.05	0
5.21	2.17	0.34
6.58	2.15	1.1
6.85	2.23	0.26

S4

Site	CWD < 7cm (kg m ⁻²)	CWDTotal kg m ⁻²	Stems > 7 cm (stems ha ⁻¹)	Stems < 7 cm (stems ha ⁻¹)	Total Live Basal Area (m ² ha ⁻¹)	Conifer Live Basal Area (m ² ha ⁻¹)
13-19	0.04	0.04	0	354	0.1	0
15-02	0.065	0.065				
13-07	0	0	0	0	0	0
13-08	0.32	0.32	0	9726	0	0
14-17	0.13	0.13	0	13086	0.2	0
13-09	0	0	225	10257	2	2
14-03	0.12	0.12	1578	5836	3.8	3.8
15-13	0.03	0.03				
14-06	0.19	0.19	1249	4598	2.6	2.3
13-20	0.07	0.07	248	0	0.6	0.6
14-20	0.11	0.39	4539	16033	13.9	13.1
13-04	0.07	0.07	727	1650	3.1	3.1
13-18	0	0	0	0	0	0
12-10	0	0	0	0	0	0
12-06	0	0	0	0	0	0
12-07	0	0	0	0	0	0
12-08	0	0	0	0	0	0
12-09	0	0	0	0	0	0
13-14	0	0	0	0	0	0
13-06	0	0	0	0	0	0
13-22	0	0	29	472	0.1	0.1
12-04	0	0	0	1650	0.5	0
13-24	0	0	0	0	0	0
12-12	0	0	0	0	0	0
13-11	0.21	0.21	0	0	0	0
13-27	0	0	0	0	0	0
13-28	0.87	0.87	0	2829	0.6	0
13-30	0	0	0	0	0	0
14-18	0.13	0.13	0	0	0	0
14-01	0	0	0	0	0	0
13-05	0	0	0	0	0	0
14-09	0.13	0.13	0	0	0	0
14-02	0.11	0.11	2293	6013	11.8	11.8

S4

Live Lorey Height (m)	Live Lorey Crown Base Height (m)	Live Conifer Crown Bulk Density (kg m ⁻³)
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3.65	0.07	0.01
3.87	1.29	0.23
4.17	0.63	0.06
4.82	0.57	0.03
6.75	5.06	1.12
6.14	1.28	0.11

3.07	0.74	0
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9.64	3.16	0.25
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