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# Variability and drivers of burn severity in the northwestern Canadian boreal forest

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**Citation:** Whitman, E., M.-A. Parisien, D. K. Thompson, R. J. Hall, R. S. Skakun, and M. D. Flannigan. 2018. Variability and drivers of burn severity in the northwestern Canadian boreal forest. Ecosphere 9(2):e02128. 10.1002/ecs2.2128

**Abstract.** Burn severity (ecological impacts of fire on vegetation and soils) influences post-fire stand structure and species composition. The spatial pattern of burn severity may compound the ecological impacts of fire through distances to seed sources and availability of bud banks and seedbeds. Land managers require spatial burn severity data to manage post-fire risks, ecosystem recovery, and assess the outcomes of fires. This research seeks to characterize and explain variability in burn severity in the northwestern boreal forest. We assessed burn severity one year post-fire in six large wildfires that burned in 2014. We measured burn severity using the Composite Burn Index, surface Burn Severity Index, Canopy Fire Severity Index, and percent overstory mortality, describing a range of surface and overstory fire effects. Burn severity was variable, ranging from unburned residuals to complete overstory mortality and intense combustion. We related field measurements to remotely sensed multispectral burn severity metrics of the differenced Normalized Burn Ratio (dNBR), the Relativized dNBR, and the Relativized Burn Ratio. Diagnostic models of burn severity using relativized metrics had lower errors and better (though not significantly so) fits to the field data. Spatial patterns of burn severity were consistent with those observed in other large fires in North America. Stand-replacing patches were large, aggregated, and covered the largest proportion of the landscape. These patterns were not consistent across the four mapped burn severity field metrics, suggesting such metrics may be viewed as related, but complementary, as they depict different aspects of severity. Prognostic models indicated burn severity was explained by pre-fire stand structure and composition, topoedaphic context, and fire weather at time of burning. Wetlands burned less severely than uplands, and open stands with high basal areas experienced lower burn severity in upland vegetation communities. This research offers an enhanced understanding of the relationship between ground observations and remotely sensed severity metrics, in conjunction with stand-level drivers of burn severity. The diverse fuel complexes and extreme fire weather during the 2014 fire season produced the complex patterns and broad range of burn severity observed.

**Key words:** boreal forest; burn severity; fire severity; fire weather; forest fire; fuel structure; topoedaphic context; wildfire.

**Received** 25 January 2018; accepted 29 January 2018. Corresponding Editor: Debra P. C. Peters.



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## Introduction

The boreal forest is the largest biome in Canada, extending from west to east, and as far north as the Arctic coast. Wildfire is the primary stand-renewing disturbance in the boreal forest (Stocks et al. 2002), and such disturbances can determine forest succession and stand composition (Weber and Stocks 1998), post-fire site productivity (Amiro et al. 2000), and may temporarily convert forested lands from carbon sinks to carbon sources, thus driving the regional carbon balance (Bond-Lamberty et al. 2007, Goodale et al. 2016). Although the fire regime of the North American boreal forest is regularly characterized as one of high-intensity, high-mortality crown fire (Johnson 1992), there is substantial variability in burn severity (changes to vegetation and soils from fire), ranging from unburned residuals and areas of low-mortality surface fire to highly charred and combusted areas with complete above-ground mortality (Kafka et al. 2001, Stocks et al. 2001).

Land managers in the boreal forest region require information about burn severity for diverse purposes with different temporal scales, from managing long-term post-fire recovery of ecosystems to addressing more immediate hazards and outcomes resulting from wildfire. For example, locating areas of high-severity burned sites containing mortality of overstory trees from fire is relevant to salvage logging (Greene et al. 2013), whereas the severity of the consumption of the surface organic layer and presence of exposed mineral soil may be more relevant to understory vegetation community development (Wang and Kemball 2005) and erosion risk management (Robichaud et al. 2000). Wildfire impacts to both the overstory and the surface are relevant to post-fire recruitment potential (Lentile et al. 2007) and estimating ecological effects on forest communities (Greene et al. 1999, 2004, Turner et al. 1999), wildlife habitat (Koivula and Schmiegelow 2007, Bond et al. 2009), identifying fire refugia and unburned islands (Kolden et al. 2012, Krawchuk et al. 2016), and estimating combustion for carbon accounting (Kurz et al. 2009, Veraverbeke et al. 2015). Spatial burn severity data are also applied for wildfire management uses, as they allow managers to engage in highly detailed mapping of fire perimeters (Kolden et al. 2012, Kansas et al. 2016), and to assess the role of fuel treatments and prescribed burning in reducing (Parks et al. 2014*a*, Prichard and Kennedy 2014, Lydersen et al. 2017) or promoting (Harvey et al. 2016*a*) subsequent fire intensity and severity through altering fuel loads or post-fire stand structure. Depending on the wildfire effect of interest, managers may require information about overstory mortality, combustion, or a combination of the two. Due to the diversity of management uses for severity data, many field metrics have been developed to measure burn severity. Percent overstory mortality measures mature tree survival following fire, whereas the Composite Burn Index (CBI; Key and Benson 2006) is a generalized measure of burn severity, mortality, and combustion across all strata of forest stands. Other metrics of burn severity aim to measure only combustion of the overstory or soil surface.

The use of multispectral remotely sensed burn severity metrics is widespread across North American forests, but the relationships of such metrics to ground observations of burn severity are variable, especially in the boreal forest (French et al. 2008). The differenced Normalized Burn Ratio (dNBR; Key and Benson 2006) was developed to assess changes in reflectance of healthy vegetation, soils, and soil moisture due to fire. Subsequently, Miller and Thode (2007) adapted this metric to better capture change relative to pre-fire conditions, with the Relativized dNBR (RdNBR). Most recently, Parks et al. (2014*a*) introduced a newer relativized severity metric, the Relativized Burn Ratio (RBR), which remains unassessed in the boreal region. Researchers have primarily assessed burn severity in the boreal forest using CBI, which has demonstrated inconsistent relationships to observed severity in the boreal forest, and studies examining other burn severity metrics such as percent overstory mortality and surface burn severity are limited (French et al. 2008).

Relationships between field measurements of burn severity and remotely sensed severity metrics are used to produce maps of burn severity (Key and Benson 2006, Morgan et al. 2014). Spatial patterns of burn severity can have long-lasting ecological effects on the composition and structure of forests that regenerate following fire (Johnstone and Chapin III 2006). Varying overstory burn severity (ecological impacts on large trees from fire) and surface burn severity (combustion of organic soils, and ecological impacts on understory vegetation) have important direct effects on post-fire forest recovery in the boreal biome. The relative availability and depth of seedbeds (mineral vs. organic soil), and fire intensity and overstory mortality affect seedling recruitment in a manner that can potentially lead to shifts in stand composition (Lavoie and Sirois 1998, Johnstone and Chapin III 2006). The mosaic of burn severity within a fire also influences landscape heterogeneity and stand-age distributions, with implications for both species assemblies and diversity (Chipman and Johnson 2002, Tews et al. 2004), and the flammability of post-fire landscapes due to fuel continuity (Turner and Romme 1994, Parks et al. 2012). Quantifying the relative performance of remotely sensed burn severity metrics in describing diverse field measurements of burn severity will provide insight into the utility and application of multispectral imagery for estimating and mapping meaningful burn severity in the northwestern boreal forest and allow a broader characterization of landscape patterns of burn severity in this region.

In ecosystems dominated by tree species that require live trees for seed sources (non-serotinous), landscape patterns of overstory mortality are important to post-fire vegetation recovery due to limits of seed dispersal (Collins et al. 2017). Analyses of the landscape pattern of stand-replacing fire in such ecosystems show that large fires, like those characteristic of the boreal forest fire regime, tend to incorporate moderately high proportions burned severely (~25%), and that stand-replacing patches are often large, simple in form with substantial core areas, and aggregated, with some variability driven by local climate and vegetation (Cansler and McKenzie 2014, Harvey et al. 2016*b*). It is therefore possible to characterize the landscape patterns of diverse overstory and understory burn severity metrics in the northwestern boreal forest, relative to documented patterns of stand-replacing fire in this (Kafka et al. 2001, Ferster et al. 2016) and other ecosystems (Cansler and McKenzie 2014, Harvey et al. 2016*b*, Collins et al. 2017). In the boreal forest, however, many tree species have adaptations that provide in situ budding rhizomes or seed sources following fire, regardless of tree mortality (Greene et al. 1999), suggesting that ecological characterizations of landscape patterns of burn severity in this region should address other fire effects, in addition to overstory mortality (Bergeron et al. 2014).

Climate acts as a significant top-down control on fire activity and area burned, having a direct effect on fire size. Large fires have larger areas of stand-replacing fire that are simpler in shape than smaller fires (Cansler and McKenzie 2014, Harvey et al. 2016*b*). Burn severity is also a product of both pre-fire vegetation (Collins et al. 2007, Boucher et al. 2016) and topography (Dillon et al. 2011, Krawchuk et al. 2016), which provide bottom-up controls on wildfire. Fire weather at the time of burning influences fire behavior and combustion (FCFDG 1992), and in west-central North America, researchers have demonstrated that extreme fire weather may overwhelm the effects of bottom-up controls on burn severity (Dillon et al. 2011, Harvey et al. 2014, Krawchuk et al. 2016). Linkages between fire weather, fuel structure, and burn severity have been identified for the forests of the western United States (Prichard and Kennedy 2014, Lydersen et al. 2017), but they remain sparsely documented in northern forests. An enhanced understanding of top-down and detailed bottom-up controls on burn severity in the northwestern boreal forest would offer insights for fuel and fire management in this fire-prone region.

The goal of this research is to describe and explain variability in burn severity in the northwestern boreal forest. Our objectives were to (1) assess the performance of three remotely sensed burn severity metrics in characterizing field observations of burn severity from the northwestern boreal forest, (2) contextualize and describe the landscape patterns of burn severity in the sampled fires, and (3) characterize the relative importance of top-down (daily fire weather) and bottom-up (topography and vegetation structure) controls on burn severity in an extreme fire year. Hypotheses related to each objective are reported in Table 1.

## Methods and Data

We measured pre-fire stand structure and burn severity metrics one year post-fire and developed bivariate models of field observations and remotely sensed burn severity metrics, linking satellite imaging of fire effects to ground observations of post-fire environments. These relationships were used to create maps of burn severity, which we analyzed with landscape patch metrics. Finally, we fit models explaining burn severity field metrics from measured stand structure, topoedaphic context, and daily fire weather at the time of burning. All analyses were performed in R (R Core Team 2017), unless otherwise specified.

### *Study area*

The six studied wildfires were very large (~14,000 to >700,000 ha), lightning-caused fires

Table 1. Research objectives and hypotheses, and associated supporting literature used in hypothesis development.

| Objective | Hypothesis | Supporting references |
|---|---|---|
| 1. Assess the performance of three remotely sensed burn severity metrics in characterizing field observations of burn severity from the northwestern boreal forest | $H_{1a}$: Bivariate relationships between the four field metrics of burn severity and remotely sensed burn severity will have different forms | Miller et al. (2009) |
| | $H_{1b}$: Relativized metrics of burn severity (RBR, RdNBR) will have a significantly stronger relationship to field metrics of burn severity than non-relativized metrics (dNBR) | Hoy et al. (2008), Cansler and McKenzie (2012), Parks et al. (2014*a*) |
| 2. Contextualize and describe the landscape patterns of burn severity in the sampled fires | $H_{2a}$: Greater than 25% of the area burned in the sampled wildfires will have burned at high severity, reflecting the large fire sizes and stand-replacing fire regime of the northwestern boreal forest | Cansler and McKenzie (2014), Harvey et al. (2016*b*), Collins et al. (2017) |
| | $H_{2b}$: High-severity patches will have larger average sizes, larger core areas and less complex patch shapes than unchanged, low, and moderate-severity burned patches, reflecting the large sizes of the sampled fires and stand-replacing fire regime of the northwestern boreal forest | Collins et al. (2007), Cansler and McKenzie (2014), Harvey et al. (2016*b*) |
| | $H_{2c}$: Landscape patterns of burn severity will vary with the different modeled burn severity field metrics | Miller et al. (2009) |
| 3. Characterize the relative importance of top-down (daily fire weather) and bottom-up (topographical and vegetation structure) controls on burn severity in an extreme fire year | $H_{3a}$: Burn severity is significantly related to topoedaphic context, pre-fire vegetation, and fire weather at the time of burning in the northwestern boreal forest | Dillon et al. (2011), Prichard and Kennedy (2014), Harvey et al. (2016*b*) |
| | $H_{3b}$: During the extreme fire year of 2014, top-down controls of daily fire weather were of dominant importance to burn severity, due to the overwhelming of other drivers by extreme weather | Dillon et al. (2011), Harvey et al. (2014), Krawchuk et al. (2016) |

*Note:* dNBR, differenced Normalized Burn Ratio; RdNBR, Relativized dNBR; RBR, Relativized Burn Ratio.

that burned in 2014 within the Northwest Territories or Wood Buffalo National Park (Fig. 1). The study area experiences infrequent, stand-replacing (i.e., lethal) fires every 40–350 yr (Boulanger et al. 2012). Although fires in this region are typically small (<200 ha), rare large fires, such as those studied here, are responsible for the vast majority of the area burned (Stocks et al. 2002). 2014 was an extreme fire year in this region, which took place during a multi-year drought (NTENR 2015). Due to the dispersed and small human population in this area, naturally occurring wildfires are generally managed following an appropriate response philosophy, with limited suppression and control efforts, where acceptable. For these reasons, the fires sampled for this study presented a rare opportunity to study burn severity in multiple concurrent, large, free-burning wildfires, in a broad range of fuel complexes.

The study area is characterized by long, cold winters and short hot summers, with mean annual temperatures between −4.3°C (in the north) and −1.8°C (in the south). It generally receives low-to-moderate annual precipitation, ranging from approximately 300 to 360 mm, primarily in the summer months (ESWG 1995, Wang et al. 2012). In the western part of the study area, glacial deposits have produced a flat to undulating plain. To the northeast of Great Slave Lake, bedrock lies closer to the surface, and the terrain becomes rolling granitic hills on the Canadian Shield (ESWG 1995). Peatlands are a substantial component of the entire study area, covering roughly a third of the area, but locally as much as 75–100% of the land's surface, with a higher cover of peatlands west and south of the Great Slave Lake (Tarnocai et al. 2011). Due to the glacial history of this region, there is minimal topography, and surficial geology and soils may contribute more meaningfully to hydraulic gradients than topography in the boreal plain (Devito et al. 2005). The study area is within the discontinuous and sporadic permafrost zones of northern Canada (NRCan 1993). No field sites had an active permafrost layer in the top 1 m of soil.

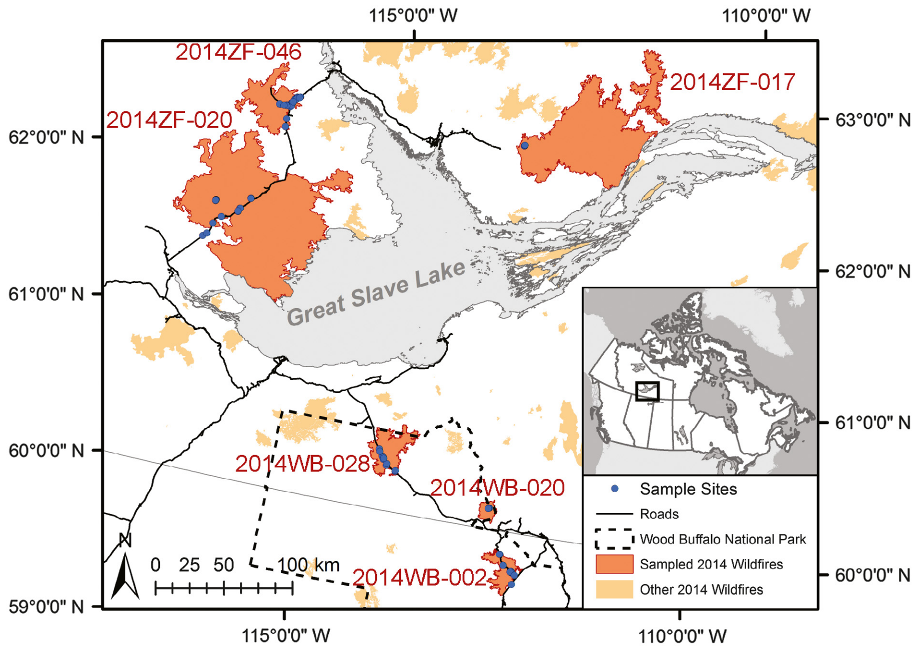


Fig. 1. The study area (extent indicated in black on inset map), located in context within North America. Dark orange areas indicate perimeters of sampled 2014 wildfires, and lighter orange areas are other 2014 wildfires. Sampled fires are labeled in red with the fire name. The 51 burned sampled field plots and 12 unburned control points are identified with blue circles. Detailed descriptions of fires and distribution of sample plots by fire are included in Table 2.

The dominant tree species in this region are black spruce (*Picea mariana*), jack pine (*Pinus banskiana*), white spruce (*Picea glauca*), and trembling aspen (*Populus tremuloides*). Secondary species of eastern larch (*Larix laricina*), balsam poplar (*Populus balsamifera*), and paper birch (*Betula papyrifera*) are also common. Many of these tree species are adapted to recurrent wildfires and have serotinous or semi-serotinous cones, or sucker from roots and rhizomes following fire (ESWG 1995, Greene et al. 1999). These characteristics make distances to live seed sources following fire a less significant driver of post-fire seedling recruitment for many species, with the exception of white spruce, which requires live trees for regeneration. Pre-fire organic soil depths range from sub-centimeter depths in xeric pine stands to meters in peatlands.

### *Field site selection and sampling*

Sampling of pre-fire stand structure and post-fire burn severity took place one year post-fire. Proposed field sites were located in areas >100 m and ≤2 km from roads, with a stratified random sample of burn severity, evenly distributed across low-, moderate-, and high-burn severity classes produced from initial assessment dNBR images (Key and Benson 2006) classified using thresholds reported in Hall et al. (2008). When traveling by helicopter, additional remote field sites were selected to represent the local range of burn severity and vegetation communities accessible from the landing site. We sampled 51 burned field sites and located twelve unburned control points, which we used to identify remotely sensed burn severity and reflectance values of unburned areas. The range of vegetation communities sampled in the burned plots was represented in the sample of unburned controls. At field sites, we placed plot centers randomly within a homogenous area of burn severity, vegetation community, and topoedaphic setting (upland or wetland) that extended ≥60 m in any direction from the plot center. Plot locations were recorded with a differential GPS unit. Plot centers of all field sites were a minimum of 103 m apart, but were on average 170 km apart.

Field sites were 30 × 30 m square plots, oriented so that two 30-m transects aligned with the cardinal directions crossed at the plot center at right angles. The vegetation community and topoedaphic context (upland or wetland class) of a plot were described according to the Field Guide to Ecosites of Northern Alberta (Beckingham and Archibald 1996). Ecosites were generalized into five functional vegetation community classes: Upland Jack Pine, Upland Black Spruce, Upland Mixedwood, Treed Wetland, and Open Wetland.

We described burn severity in each plot using percent overstory mortality, CBI with height thresholds modified for northern forests (Key and Benson 2006, Kasischke et al. 2008), Canopy Fire Severity Index (CFSI; Kasischke et al. 2000), and Burn Severity Index (BSI; Loboda et al. 2013). Composite Burn Index values ranging from 0 (unburned) to 3 (severely burned) were estimated for each forest stratum present in the 30 × 30 m plot and averaged. Canopy Fire Severity Index was used to estimate the level of crown involvement in fire and intensity of overstory combustion, whereas BSI was used to assess the burn severity of the forest floor and ground surface. We estimated the relative area of the seven CFSI classes, ranging from 0 (no tree mortality) to 6 (no primary branches remaining, pole charring occurred) in four 10 × 10 m subplots, at the four corners of the plot. In the same four subplots, we also estimated the relative area of five surface BSI classes described in Dyrness and Norum (1983) ranging from 0 (unburned) to 4 (organic soil ashed, mineral soil exposed). The area of each class was used to calculate weighted sums following the method described in Loboda et al. (2013), and the resulting four CFSI and BSI values per plot were averaged. Percent overstory mortality (MORT) from fire, pre-fire overstory tree species composition, stem density (stems/ha), tree basal area (BA; $m^2$/ha), and estimated pre-fire live conifer crown base height (CBH; m) were measured for 32 trees >3 cm in diameter at breast height with the point-centered quarter method (Cottam et al. 1953, Mitchell 2007) at eight evenly spaced points along the two transects. Where stem density was very low (i.e., open wetlands), a variable radius circle plot with a minimum length of 15 m was taken at the plot center to measure overstory trees. Pre-fire understory stem density of seedlings and saplings was measured using 3 m radius plots at the end points of each transect. The number of understory density plots sampled ranged from one to four, depending on the density and evenness of the seedling and saplings. Understory and overstory stem density were combined for analyses. Pre-fire overstory fuel load (flammable biomass in t/ha) at each site was modeled using allometric equations (Ung et al. 2008, Thompson et al. 2017). Sections from fire-scarred trees were collected to determine stand age and fire history at each plot. If no scarred trees were identified nearby, a section of a mature dominant tree was collected.

### *Remote sensing of burn severity*

Remotely sensed burn severity within the six fires was estimated using multispectral Landsat 8 OLI (Operational Land Imager) and Landsat 5 TM (Thematic Mapper) images (Landsat Level-1 imagery, courtesy of the U.S. Geological Survey). Image pairs were selected for an extended assessment of burn severity, where post-fire images were captured in the growing season after the fire (Table 2; Key and Benson 2006). Images were converted to at-surface reflectance using dark-object-subtraction in QGIS with the semi-automatic classification plugin (Congedo 2016, QGIS Development Team 2017). Clouded and shadowed areas within fire perimeters were masked by hand in ArcGIS (ESRI 2012), and permanent waterbodies (NRCan 2008) were also masked. The Normalized Burn Ratio (NBR; Eq. 1), dNBR (Eq. 2), RdNBR (Eq. 3), and RBR (Eq. 4) were calculated from at-surface reflectance of near-infrared (NIR) and shortwave infrared (SWIR; Landsat bands 4 and 7 [TM] or 5 and 7 [OLI]) and then multiplied by 1000. All remotely sensed burn severity metrics were calculated in R with the raster package (Hijmans 2016). We included an offset term ($dNBR_{offset}$), normalizing dNBR values in unburned areas to 0 by subtracting the average dNBR in unburned areas to account for phenological differences between images (Eq. 2; Key 2006, Miller and Thode 2007). Values of the remotely sensed burn severity metrics at each field plot were estimated from the four nearest 30 × 30 m pixels using bilinear interpolation.

$$\mathrm{NBR} = \frac{\mathrm{NIR} - \mathrm{SWIR}}{\mathrm{NIR} + \mathrm{SWIR}} \quad (1)$$

$$\mathrm{dNBR} = (\mathrm{NBR}_{\mathrm{prefire}} - \mathrm{NBR}_{\mathrm{postfire}}) - \mathrm{dNBR}_{\mathrm{offset}} \quad (2)$$

Table 2. Pairs of Landsat 8-OLI and Landsat 5-TM images used for measurement of remotely sensed burn severity and summary information describing sampled wildfires.

| Fire name | Start date | Fire size (ha) | Number of field plots | Pre-fire sensor | Path | Row | Pre-fire image date | Post-fire sensor | Path | Row | Post-fire image date |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 2014ZF-020 | June 17, 2014 | 730,855 | 12 | Landsat 8-OLI | 48 | 17 | May 30, 2013 | Landsat 8-OLI | 48 | 17 | 20 May 2015 |
| 2014ZF-017 | June 16, 2014 | 450,207 | 5 | Landsat 8-OLI | 45 | 16 | June 12, 2014 | Landsat 8-OLI | 46 | 16 | 23 June 2015 |
| 2014ZF-017 | – | – | – | – | – | – | – | Landsat 8-OLI | 44 | 16 | 25 June 2015 |
| 2014ZF-046 | July 3, 2014 | 106,485 | 17 | Landsat 8-OLI | 48 | 16 | May 30, 2013 | Landsat 8-OLI | 47 | 16 | 29 May 2015 |
| 2014WB-028 | August 1, 2014 | 66,673 | 8 | Landsat 8-OLI | 45 | 18 | June 13, 2014 | Landsat 8-OLI | 46 | 18 | 23 June 2015 |
| 2014WB-002 | June 15, 2014 | 38,060 | 6 | Landsat 5-TM | 44 | 18 | June 14, 2011 | Landsat 8-OLI | 44 | 18 | 25 June 2015 |
| 2014WB-020 | July 8, 2014 | 13,979 | 3 | Landsat 5-TM | 44 | 18 | June 14, 2011 | Landsat 8-OLI | 44 | 18 | 25 June 2015 |

*Notes:* OLI, Operational Land Imager; TM, Thematic Mapper. Images are listed by the name of the fire analyzed. Two post-fire images for fire 2014ZF-017 were mosaicked together. As this fire appears twice, identical values are not repeated and instead are indicated with a dash.

$$\mathrm{RdNBR} = \frac{\mathrm{dNBR}}{\left|(\mathrm{NBR}_{\mathrm{prefire}})\right|^{0.5}} \quad (3)$$

$$\mathrm{RBR} = \frac{\mathrm{dNBR}}{(\mathrm{NBR}_{\mathrm{prefire}} + 1.001)} \quad (4)$$

### *Daily fire weather*

To assess the potential relationship between weather and burn severity, we interpolated Moderate Resolution Imaging Spectroradiometer (MODIS; CFS 2015) and Visible Infrared Imaging Radiometer Suite hotspots (USDA Forest Service 2014) from the year 2014, using a weighted mean (Parks 2014) to estimate the day of burning (DOB) for each field site. Fire weather conditions were represented using the Canadian Forest Fire Weather Index (FWI) System, which uses daily inputs of temperature, relative humidity, precipitation, and wind speed to produce three fuel moisture codes (Fine Fuel Moisture Code [FFMC], Duff Moisture Code [DMC], and Drought Code [DC]) and three indexes of fire behavior potential (Initial Spread Index [ISI], Buildup Index [BUI], and Fire Weather Index; Van Wagner 1987). Noon (Local Standard Time) weather and FWI System values on the DOB for each site were downscaled from North American Regional Reanalysis data (NARR; Mesinger et al. 2006, Jain et al. 2017) using ordinary kriging. Fire Weather Index System indexes were calculated from the interpolated temperature, precipitation, relative humidity, and wind speed using the cffdrs package (Wang et al. 2017), with starting values from the interpolated values of the FFMC, DMC, and DC from the previous day (Jain et al. 2017).

### *Analysis*

Statistical differences in burn severity and stand structure between vegetation communities were assessed with Wilcoxon signed-rank tests, ANOVA, and post hoc least-squares means tests (lsmeans package; Lenth 2016). We produced scatterplots and computed Spearman's rank correlation coefficients to determine the nature of the relationships between the field measures of burn severity and remotely sensed burn severity metrics. Subsequently, we used generalized linear models (GLMs) and landscape patch metrics to examine landscape patterns and drivers of burn severity in this region. Bivariate GLMs were used to develop diagnostic models describing the relationship between remotely sensed burn severity and field metrics of burn severity. All model fits were assessed using averages of root-mean-square error and $R^2$ (the square of the correlation between observed values and predicted values), calculated following a 10-fold cross-validation (CV) with 100 repeats in the caret package (Khun 2017). All statistical tests in this study were conducted at the 5% level of significance. Continuous values of the four burn severity field metrics were predicted from rasters of remotely sensed burn severity using the bivariate GLMs and subsequently classified into unchanged, low, moderate, and high severity using breaks described in Table 3. The relative quality of each remotely sensed burn severity metric as a classifier of burn severity was assessed using the kappa statistic in the psych package (Revelle 2017).

Burn severity thresholds were identified from field observations of meaningful differences in burn severity for each metric and validated with the distribution of sampled data. Generally, unchanged sites are unburned or lightly burned, where mild and patchy fire effects were intermingled with unburned areas. Low-severity burned areas had scorched or lightly charred surfaces but substantial organic matter still existed post-fire. Some overstory mortality may be evident in the stand, but any crown involvement in the fire did not consume all small branches in the overstory. Moderate-severity burned areas have charred surfaces and may have some exposed mineral soil and ash present. Overstory tree mortality was more common in these stands, with primary branches and some dead non-combusted foliage remaining on the trees despite fire crowning. High-severity burned stands have surfaces substantially composed of exposed mineral soil or ash. There was complete stand mortality, and the majority of primary branches are consumed (illustrative photographs provided in Appendix S1: Table S1). Models using relativized burn severity metrics generally had lower error and higher CV $R^2$ values than models using dNBR to describe burn severity field metrics; however, all bivariate models were significant ($P \leq 0.001$) and none had statistically stronger fits to the field data (Wilcoxon signed-rank tests of model residuals $P \geq 0.44$). Furthermore, because the confidence intervals of the kappa statistic for all three remotely sensed metrics overlapped, we chose to present only RBR in subsequent analyses and visualizations.

Table 3. Breaks used to classify maps of modeled burn severity field metrics of Composite Burn Index (CBI), surface Burn Severity Index (BSI), Canopy Fire Severity Index (CFSI), and percent overstory mortality (MORT).

| Burn severity metric | Unchanged | Low | Moderate | High |
|---|---|---|---|---|
| CBI (0–3) | ≤0.1 | >0.1–1.5 | >1.5–2.25 | >2.25 |
| BSI (0–4) | ≤0.5 | >0.5–1.75 | >1.75–3 | >3 |
| CFSI (0–6) | ≤0.1 | >0.1–2 | >2–4 | >4 |
| MORT (0–100) | ≤10 | >10–50 | >50–95 | >95 |

*Notes:* Modeled values of burn severity metrics were estimated from raster maps of the remotely sensed Relativized Burn Ratio. Representative photographs of burn severity classes are included in Appendix S1: Table S2.

Landscape patterns of classified burn severity within fire perimeters were assessed using a selection of patch metrics calculated in FRAGSTATS (McGarigal et al. 2012). Patches were defined using the eight-neighbor rule. To limit edge effects, landscape patterns of burn severity were assessed within the core area of fire perimeters only, excluding areas ≤100 m from the fire edge (following Parks et al. 2014*b*). Single-pixel patches were removed using a majority filter in ArcGIS, also with an eight-neighbor rule. Patch metrics were selected to characterize the relative dominance, and spatial arrangement and distribution of patches of each severity class. The area-weighted mean patch size and the proportion of the landscape burned in each severity class were used to describe the relative dominance of severity classes on the landscape. Area-weighted means were used as they capture the higher relative probability of a randomly selected point falling into a large patch. The core area of a patch was defined as areas ≥600 m from the edge. This threshold was selected as a conservative estimate of the maximum distance for long-distance seed dispersal for white spruce (Greene and Johnson 2000), in an ecosystem where most trees have in situ bud banks or seed sources, regardless of overstory mortality. The clumpiness index and area-weighted perimeter-to-area ratio (PARA) were selected to characterize how patches are arranged on the landscapes, capturing the relative dispersion and complexity of shape of the different severity classes. Clumpiness is the deviation in the proportion of like adjacencies (pixel edges shared with a pixel of the same class) from that expected in a random landscape. Together, these metrics were used to characterize the post-fire pattern of combustion, seedbeds, and seed sources, and the heterogeneity of the post-fire landscape mosaic.

We also fit prognostic multivariable GLMs to estimate burn severity field metrics from pre-fire stand structure, topoedaphic context, and fire weather. A complete suite of stand structure and age, and fire weather variables were considered for each model. Only those explanatory variables that were significant ($P \leq 0.05$) were retained. If explanatory variables were highly correlated ($|\rho| \geq 0.6$) with one another, then the variable contributing most significantly to the model was retained and the other correlated variable was removed. Upland and wetland datasets were separated and differences in model performance and in burn severity between the two groups were assessed. Finally, the three remotely sensed burn severity metrics were assessed for complementarity to the field data by adding each to the complete prognostic multivariable model and examining model fit metrics and *t*-values of predictor variables.

## Results

### *Field measures of burn severity*

The CBI, BSI, CFSI, and percent or proportion overstory mortality (MORT) field measures of burn severity were sensitive to pre-fire vegetation communities described by dominant upland tree species, and treed or open wetlands (Fig. 2). Of the vegetation communities, Upland Jack Pine tended to incur the higher ranges of burn severities. Burn severity was most variable in Upland Mixedwood stands, which is likely attributable to the variable proportions of conifer and deciduous species that would influence fuels and the likelihood to burn. Not unexpectedly, Open Wetlands had lower values of burn severity compared to Treed Wetlands (Fig. 2).

Of interest was the degree to which remotely sensed burn severity metrics were statistically correlated to the four burn severity metrics. Both dNBR and RBR were most highly correlated to CFSI, followed by CBI. Relativized dNBR was more correlated to CBI than CFSI. All three remotely sensed metrics were less correlated with BSI than CBI and CFSI, and had the weakest correlation to MORT; however, all correlations were statistically significant ($P < 0.001$; Appendix S1: Table S2). These correlations were supported by the scatterplots of the data points between the remotely sensed metrics and the four field burn severity metrics. In particular, there was a distinct sigmoidal relationship between the remotely sensed severity metrics and MORT, which explains the lower correlation coefficient (Fig. 3).

### *Diagnostic models of burn severity*

Burn severity was statistically lower in wetlands than in uplands (Wilcoxon signed-rank test, $P \leq 0.02$) when measured by CBI, and BSI, but not when using the overstory burn severity measures of CFSI and MORT ($P \geq 0.20$; Fig. 2). Composite Burn Index, CFSI, and BSI were explained by pre-fire vegetation communities (Type II ANOVA, $P < 0.003$); however, MORT was not statistically related to vegetation community ($P = 0.5$). Vegetation community classes alone explained 22.6% (CFSI), 41.7% (CBI), and 54.1% (BSI) of the variance ($\omega^2$) in field measurements of burn severity. Post hoc comparisons of least-squares means with a Tukey *P*-value adjustment confirmed some statistical differences in burn severity among vegetation communities for CBI and BSI ($P \leq 0.05$; Fig. 2). Upland Jack pine and Open Wetlands demonstrated distinct levels of CBI and BSI. Other vegetation communities shared similar levels of severity with one or both of these two communities. The forms of the bivariate relationships between the burn severity field metrics and the remotely sensed burn severity metrics were different for each field metric (Fig. 3). Composite Burn Index, CFSI, and MORT were best modeled with nonlinear model forms, whereas BSI had a linear relationship to remotely sensed burn severity (Fig. 3).

All diagnostic bivariate models estimating field observations of burn severity from the three remotely sensed severity metrics were significant (Table 4; $P \leq 0.001$). Models using the relativized burn severity metrics (RdNBR and RBR) generally better described the burn severity field data and had lower error and higher CV $R^2$ values than those using dNBR. Relativized dNBR typically had the lowest error and the highest CV $R^2$; however, RBR had the best fit for the CFSI model. Relativized dNBR had the highest accuracy and reliability (kappa statistic) in classifying burn severity metrics that considered surface burn severity (CBI and BSI), but performed the worst in classifying burn severity landscapes where only overstory impacts were measured (CFSI and MORT). Relativized Burn Ratio had the highest accuracy and reliability in classifying overstory burn severity (Table 4). Although there

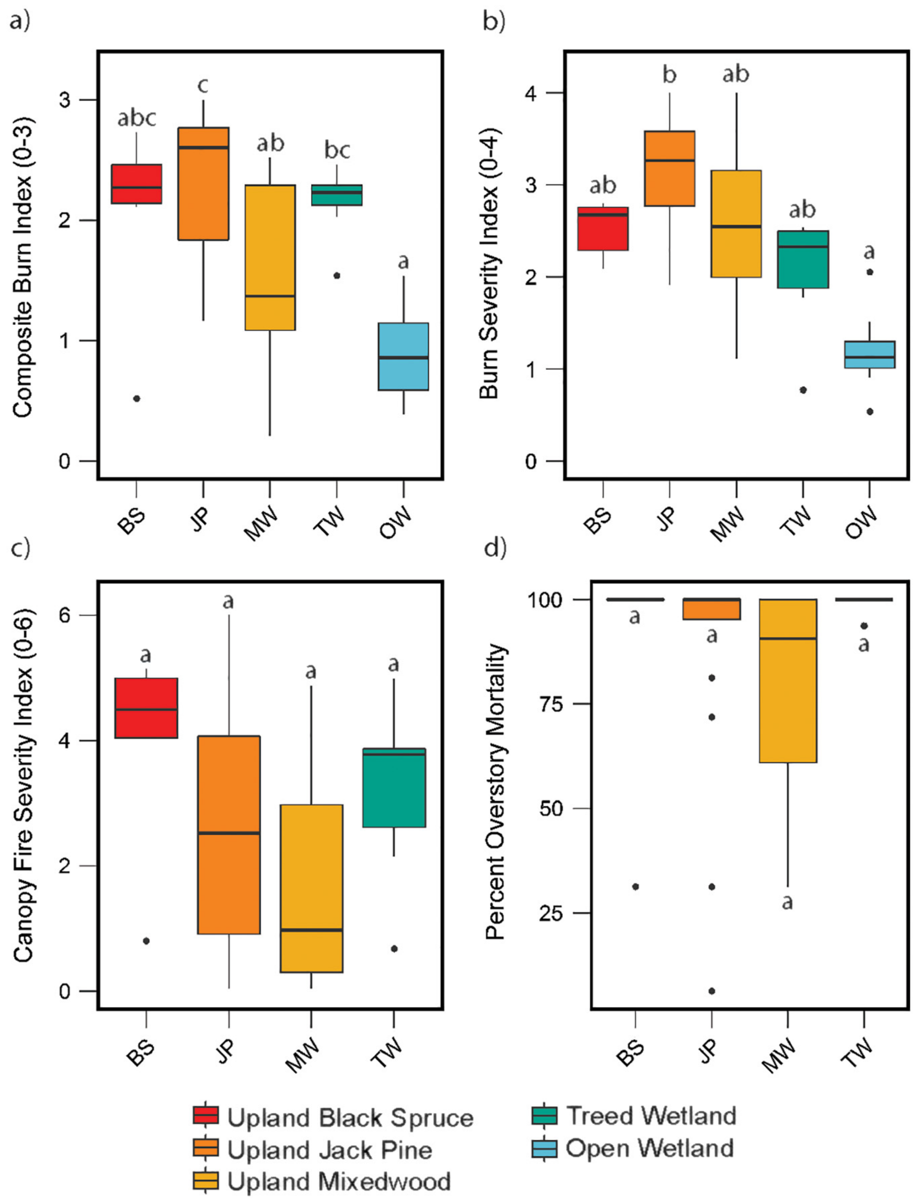


Fig. 2. Distribution of burn severity metrics within vegetation communities of Upland Black Spruce (BS), Upland Jack Pine (JP), Upland Mixedwood (MW), Treed Wetland (TW), and Open Wetland (OW). Burn severity metrics are (a) Composite Burn Index, (b) surface Burn Severity Index, (c) Canopy Fire Severity Index, and (d) percent overstory mortality. Canopy Fire Severity Index and percent overstory mortality are not reported for OWs as these are not forested systems. Letters above each boxplot indicate significant differences ($P \leq 0.05$) in least-squares means with a Tukey *P*-value adjustment.

were differences in error, CV $R^2$ values, and classification accuracy of the three different remotely sensed severity metrics in describing observed burn severity, there was no statistical difference in model fits (Wilcoxon signed-rank tests of model residuals $P \geq 0.44$) and the 95% confidence intervals of kappa statistics overlapped.

When datasets were limited to topoedaphic subsets (wetlands and uplands), dNBR generally had the best fit in wetland-only datasets, whereas

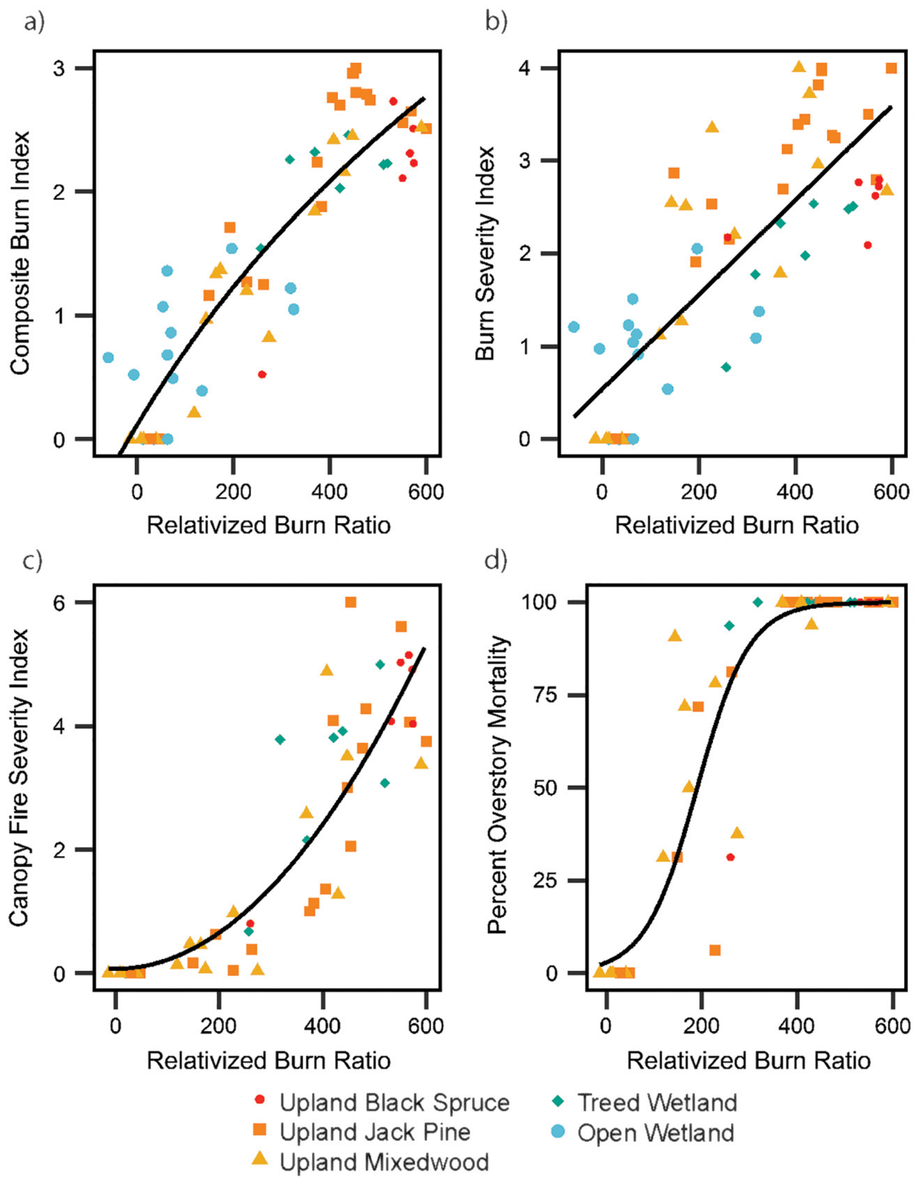


Fig. 3. Relationships between (a) Composite Burn Index, (b) surface Burn Severity Index, (c) Canopy Fire Severity Index (CFSI), and (d) percent overstory mortality (MORT) and the Relativized Burn Ratio across all vegetation communities (indicated by point color and shape). Open Wetlands are excluded from the CFSI and MORT models as these are not forested systems. Unburned control sites are classified into the same vegetation communities and are identifiable as points with y-axis values of zero. Model fit statistics are reported in Table 4, as are models of the same form with independent variables of the differenced Normalized Burn Ratio (dNBR) and Relativized dNBR (RdNBR).

models with RdNBR and RBR had stronger relationships to the field data and lower error when fit with only upland field sites. Model significance (*P*) decreased slightly from the significance of the full models when fit with wetlands only, but remained <0.05 (Appendix S1: Table S3). For severity metrics that considered surface burn severity, partitioning into uplands and wetlands generally improved the model fit, but increased error in the wetland-only datasets. When the CFSI

Table 4. Diagnostic generalized linear models of Composite Burn Index (CBI), surface Burn Severity Index (BSI), Canopy Fire Severity Index (CFSI), and proportion of overstory mortality (MORT) predicted from the remotely sensed burn severity metrics differenced Normalized Burn Ratio (dNBR), Relativized dNBR (RdNBR), and Relativized Burn Ratio (RBR).

| Severity metric | Formula | Distribution | CV MAE | CV RMSE | CV $R^2$ | Overall accuracy | Kappa | Kappa CI ± |
|---|---|---|---|---|---|---|---|---|
| dNBR | CBI = dNBR × $(0.1651\ [\text{dNBR}] + 161.8346)^{-1}$ | Gaussian | 0.38 | 0.45 | 0.83 | 49.21 | 0.31 | 0.16 |
| RdNBR | CBI = RdNBR × $(0.52379\ [\text{RdNBR}] + 46.13491)^{-1}$ | Gaussian | 0.34 | 0.40 | 0.87 | 57.14 | 0.41 | 0.16 |
| RBR | CBI = RBR × $(0.1267\ [\text{RBR}] + 140.1737)^{-1}$ | Gaussian | 0.36 | 0.43 | 0.85 | 53.97 | 0.36 | 0.16 |
| dNBR | BSI = 0.6178 + (dNBR × 0.0037) | Gaussian | 0.71 | 0.80 | 0.66 | 36.51 | 0.12 | 0.15 |
| RdNBR | BSI = 0.484 + (RdNBR × 0.0021) | Gaussian | 0.61 | 0.71 | 0.72 | 39.68 | 0.19 | 0.15 |
| RBR | BSI = 0.537 + (RBR × 0.0051) | Gaussian | 0.66 | 0.75 | 0.70 | 38.10 | 0.16 | 0.15 |
| dNBR | CFSI† = −0.2069 + ($\text{dNBR}^2$ × 0.000008) | Gaussian | 0.78 | 0.99 | 0.80 | 73.47 | 0.65 | 0.18 |
| RdNBR | CFSI† = 0.1324 + ($\text{RdNBR}^2$ × 0.000002) | Gaussian | 0.81 | 1.04 | 0.76 | 57.14 | 0.43 | 0.18 |
| RBR | CFSI* = −0.0714 + ($\text{RBR}^2$ × 0.0001) | Gaussian | 0.70 | 0.93 | 0.82 | 73.47 | 0.65 | 0.18 |
| dNBR | MORT† = −3.361 + (dNBR × 0.0134) | Quasibinomial | 0.09 | 0.15 | 0.83 | 59.18 | 0.37 | 0.21 |
| RdNBR | MORT† = −3.5882 + (RdNBR × 0.008) | Quasibinomial | 0.08 | 0.14 | 0.86 | 57.14 | 0.33 | 0.21 |
| RBR | MORT† = −3.508 + (RBR × 0.0185) | Quasibinomial | 0.09 | 0.15 | 0.85 | 59.18 | 0.35 | 0.21 |

*Notes:* Model fits are described using averages of 10-fold cross-validated (CV) mean absolute error (MAE), root-mean-square error (RMSE), and $R^2$, derived from 100 repeats. Error terms are expressed in the units of the predicted variable. *P*-values of models were derived from $\chi^2$ tests of model deviance explained relative to a null model. The classification accuracy of mapped burn severity produced using each model and kappa statistic and associated 95% confidence intervals (CI) are also reported. $P \leq 0.001$ for all models.

† Subset of forested sites only, excluding data from Open Wetlands.

model was fitted with upland sites the error decreased and the CV $R^2$ increased; however, the MORT model fit worsened slightly, but non-significantly (Appendix S1: Table S3). Models of CFSI and MORT were not significant when fitted using wetland data only due to the small sample size of forested wetlands ($n$ = 8), and thus, these models are not reported in Appendix S1: Table S3.

### *Landscape patterns of burn severity*

Although all classified maps were derived from the same RBR rasters, the landscape pattern metrics differed, depending on the modeled burn severity metric (Fig. 4). When all fires were considered together, maps of CBI, BSI, and CFSI classified the majority of the burned landscape as moderate severity, and this class tended to have the largest mean patch size. MORT demonstrated a different trend, where high-severity classes were the majority of the area, and also had the largest mean patch size (Fig. 4; Appendix S1: Table S4). Unchanged and low-severity patches tended to be more complex in form for all modeled severity metrics. Mapped CBI was an exception to this and had more complex moderate-severity patches, although unchanged patches had a similar mean PARA value to that of moderate-severity patches (Fig. 4; Appendix S1: Table S4). Relatively few patches had significant core areas (>600 m from the patch edge). Core area of severity classes varied from fire to fire, and with the mapped burn severity field metric. Unchanged and low-severity patches had the largest total core area when landscapes were classified by CBI, BSI, and CFSI. Once again, MORT exhibited a substantially different trend and each fire had patches with quite substantial core areas, especially within high-severity burned patches. High-severity patches were broadly the most aggregated and were substantially more likely to have like neighbors, when classified severity landscapes were derived from modeled values of CBI, BSI, or MORT. Canopy Fire Severity Index differed from this pattern, with unchanged patches being the most likely to share like adjacencies; however, this varied substantially between fires (Fig. 4; Appendix S1: Table S4).

### *Prognostic models of burn severity*

The four burn severity field metrics were significantly explained by multivariable prognostic

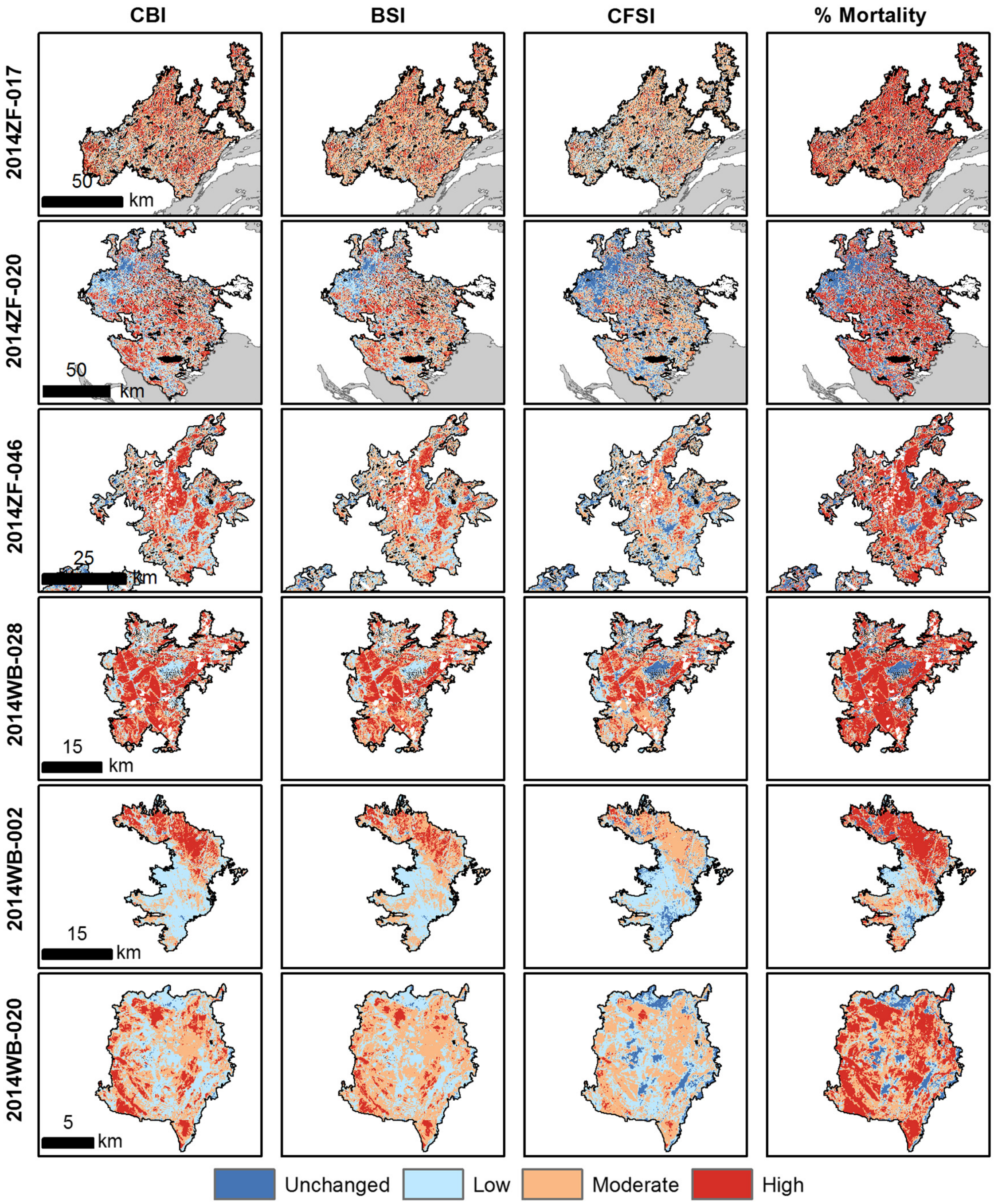


Fig. 4. Remotely sensed burn severity maps produced by estimating Composite Burn Index (CBI), surface Burn Severity Index (BSI), Canopy Fire Severity Index (CFSI), and percent overstory mortality (% Mortality) from bivariate relationships with the Relativized Burn Ratio within six fires: 2014ZF-020, 2014ZF-017, 2014ZF-46, 2014WB-28, 2014WB-02, and 2014WB-20. Continuous burn severity values were classified using thresholds reported in Table 3. Clouded areas inside fires (fire perimeters outlined in black) are masked and appear in white. Waterbodies inside fire perimeters are shown in black, and Great Slave Lake is shown in gray. Statistical summaries of landscape patterns of burn severity are reported in Appendix S1: Table S4.

Table 5. Prognostic generalized linear models of Composite Burn Index (CBI), surface Burn Severity Index (BSI), Canopy Fire Severity Index (CFSI), and proportion of overstory mortality (MORT) predicted from pre-fire fuel structure, topoedaphic context, and fire weather at the time of burning.

| Formula | Distribution | $P$ | CV MAE | CV RMSE | CV $R^2$ |
|---|---|---|---|---|---|
| CBI = 1.883 − (BA† × 0.03451) + (CBH‡ × 0.1361) − (WET§ × 0.9656) + 0.00009 (WET × STEMS¶) + (STEMS × 0.000009) | Gaussian | 0.004** | 0.44 | 0.54 | 0.62 |
| BSI = 0.5082 − (WET × 1.1514) − (FL# × 0.0507) + (CBH × 0.0979) + (CON‖ × 0.0272) + (BUI × 0.0212) − 0.0003 (CON × BUI) | Gaussian | <0.001*** | 0.51 | 0.63 | 0.63 |
| CFSI† = 4.2193 − (BA × 0.0707) + 0.2192 (CBH × log(STEMS)) − (CBH × 1.5453) − (log(STEMS) × 0.2215) | Gaussian | <0.001*** | 1.27 | 1.46 | 0.62 |
| MORT† = 1.9986 − (BA × 0.0872) + (CBH × 0.3972) | Quasibinomial | 0.019* | 0.13 | 0.17 | 0.65 |

*Notes:* BA, basal area ($m^2$/ha); BUI, Buildup Index; CBH, crown base height. Independent variables in each equation appear in order of importance (*t*-values in Appendix S1: Table S5). Model fits are described using averages of 10-fold cross-validated (CV) mean absolute error (MAE), root-mean-square error (RMSE), and $R^2$, derived from 100 repeats. Error terms are expressed in the units of the predicted variable. $P$-values of models were derived from $\chi^2$ tests of model deviance explained relative to a null model. Levels of significance are expressed as *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

† Subset of forested sites only, excluding data from Open Wetlands

‡ Median live CBH of conifer species (m)

§ Wetland (binary factor; 1 = wetland, 0 = upland)

¶ Stem density (understory and overstory stems/ha)

\# Pre-fire overstory fuel load (t/ha)

‖ Stand percent non-deciduous conifer by fraction of fuel load

models, with CV $R^2$ values ranging from 0.62 to 0.65 (Table 5). Burn severity metrics that integrated overstory impacts (CBI, CFSI, and MORT) were predominantly related to stand total BA ($m^2$/ha), median conifer live CBH (m), and stem density (STEMS; stems/ha). Time since last fire and time since stand origin were not significant variables ($P \geq 0.05$) in any model. Burn Severity Index was the only severity metric for which model fit improved with the inclusion of fire weather variables. Severity metrics that considered surface impacts (BSI, CBI) also included topoedaphic context (whether a site was an upland or wetland) as an important explanatory variable (Table 5; Appendix S1: Table S5). The stand structure, composition, and pre-fire overstory fuel load variables retained in models were significantly different between sampled uplands and wetlands (Wilcoxon signed-rank test, $P \leq 0.04$). Post hoc comparisons of stand structure and fuel load using least-squares means with a Tukey $P$-value adjustment confirmed statistical differences between vegetation communities, primarily associated with dominant tree species and overstory density in each community (Fig. 5).

The explanatory power of all prognostic multivariable models could be improved with the addition of remotely sensed burn severity metrics (Appendix S1: Table S6), with RBR providing the most significant improvement in model fit to field measurements of burn severity. When models were fitted using both field and remotely sensed burn severity metrics RBR was typically the most important predictor of burn severity, whereas dNBR and RdNBR were typically less important than measured pre-fire variables. For the BSI model whether the site was a wetland or not remained the most important variable after RBR was added (Appendix S1: Table S7). The multivariable linear models did not represent a significant improvement in predictive power compared to the bivariate models of severity (Tables 4, 5); however, they elucidate significant relationships between pre-fire stand structure and composition, and fire weather drivers of observed severity. Models fitted with both field data and RBR had similar fits to the models using remotely sensed burn severity metrics alone (Table 4; Appendix S1: Table S6).

## Discussion

### *Selecting multispectral remotely sensed burn severity products for application in the northwestern boreal forest*

Multispectral remotely sensed imagery is widely available, making it an appealing source of spatial burn severity data. The utility of such

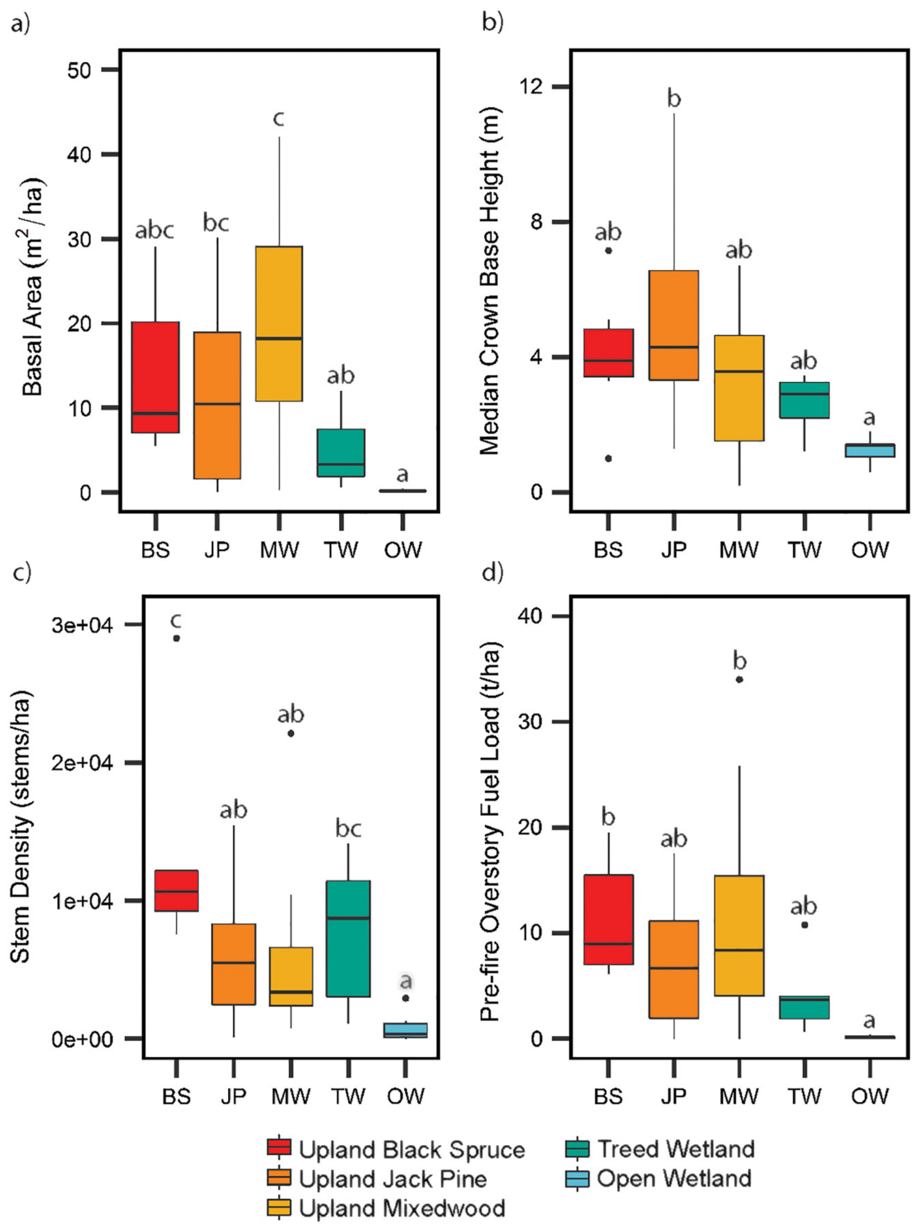


Fig. 5. Distributions of pre-fire stand structural characteristics of (a) basal area ($m^2$/ha), (b) median live crown base height of conifers (m), (c) stem density (stems/ha), and (d) pre-fire overstory fuel load (t/ha) within pre-fire vegetation communities of Upland Black Spruce (BS), Upland Jack Pine (JP), Upland Mixedwood (MW), Treed Wetland (TW), and Open Wetland (OW). Detailed relationships between burn severity metrics, stand structure, and pre-fire overstory fuel load are shown in Appendix S1: Fig. S1. Letters above each boxplot indicate significant differences ($P \leq 0.05$) in least-squares means with a Tukey $P$-value adjustment.

imagery for burn severity assessment in the northwestern Canadian boreal forest is supported by our results, as all four measured field metrics of burn severity can be estimated from remotely sensed burn severity. Each field metric of burn severity had a different form of relationship to the remotely sensed severity metrics, supporting hypothesis $H_{1a}$. Miller et al. (2009) identified nonlinear relationships between RdNBR and various measures of fire severity. We also found that primarily nonlinear regressions best fit the relationship between the field data and the remotely sensed severity metrics, with the exception of BSI. Burn Severity Index had a linear relationship to the remotely sensed variables, and had the worst modeled fit and classification accuracy, regardless of the remotely sensed severity metric employed.

Both of the relativized metrics had better fits to the field data, providing some support for our hypothesis $H_{1b}$. Differences in model fit as measured by residuals and the classification accuracy of the different metrics (kappa statistic) were not statistically different, leading us to reject our hypothesis that the relativized metrics have significantly stronger relationships to the field data. These findings generally match those of researchers who have examined the explanatory power and classification accuracy of dNBR and RdNBR in southwestern North America, with relativized metrics offering some improvement over dNBR for modeling and classifying burn severity (Miller and Thode 2007, Cansler and McKenzie 2012, Parks et al. 2014*a*), as well as some research in boreal Alaska (Hoy et al. 2008). When Soverel et al. (2010) compared the performance of RdNBR and dNBR in Canada, they found that dNBR was a better classifier of burn severity. Although the relativized transformations of dNBR had better fits to field data in our study, we examined more burn severity field metrics than just CBI and the geographic extent of our study was narrower than that of Soverel et al. (2010), possibly accounting for these differences. Model fits between CBI and dNBR and RdNBR were similar (in terms of $R^2$) within the three northern boreal fires they assessed, and generally demonstrated a higher accuracy when RdNBR was used for classification, despite their finding that dNBR best represented their complete suite of fires. We found no statistical improvement in model fits and classification accuracy when using relativized metrics, supporting the assertion that dNBR remains a useful and applicable severity metric in this region (Hall et al. 2008, Soverel et al. 2010).

We failed to consistently replicate the significant improvement in model fit when comparing RBR to RdNBR, observed by Parks et al. (2014*a*); however, our sample size was much smaller. Of the two relativized metrics, RdNBR typically had the best fit to the field data and the lowest error, but RBR had a better or equivalent model fit and higher classification accuracy when describing overstory burn severity metrics (CFSI, MORT). Relativized Burn Ratio also had the strongest explanatory power when pre-fire forest structure, landscape, and weather variables were controlled for in models. It is possible the difference in importance when pre-fire variables are considered is due to the slightly higher correlation of RBR to pre-fire NBR and thus to the pre-fire landscape, relative to the less correlated RdNBR (Parks et al. 2014*a*). If they are available or easily calculated, it is likely preferable to use relativized multispectral remotely sensed burn severity metrics (RdNBR, RBR), rather than dNBR, to map burn severity in the northwestern boreal forest. When surface burn severity impacts are of interest, RdNBR may offer a slight improvement over RBR, whereas overstory burn severity impacts may be best represented by RBR.

As all four field metrics were statistically related to remotely sensed burn severity metrics, the interpretation of remotely sensed burn severity pixel values and maps can therefore vary, depending on the ground-based measurement of interest. For example, in this dataset, an RBR value of approximately 400 indicates near-complete stand mortality, yet this same value corresponds to a tree with secondary branches remaining (in terms of CFSI), and moderate scorch and charring of the soil surface (in terms of surface BSI). Therefore, multispectral estimates of burn severity can and should be interpreted relative to specific management or research interests. Although modeled relationships to CBI offer insight into general fire effects and combustion, it is also beneficial to use remotely sensed burn severity to characterize other specific ecological impacts of fire (Morgan et al. 2014), as in this study. Whereas stand mortality and landscape patterns of stand-replacing fire are of great importance and interest in mixed-severity fire regimes (Dillon et al. 2011, Cansler and McKenzie 2014, Harvey et al. 2016*b*, Collins et al. 2017), the same maps of remotely sensed burn severity can be used to estimate post-fire seed viability due to combustion (CFSI), as well as seed bed and bud-bank availability (BSI), in an ecosystem adapted to recurrent stand-replacing fires, such as the northwestern boreal forest.

*Landscape patterns of burn severity in the northwestern boreal forest*

The 2014 northwestern boreal fire season occurred in a year of extreme drought. Fires burned for months at a time, often with limited suppression, and in a broad range of weather conditions and fuels (NTENR 2015). Although the landscape patterns of burn severity in these fires are the result of extreme conditions, and in some cases of dramatic fire behavior, the variable weather and fuels produced a complete range of burn severity, including unburned residual stands. The spatial burn severity patterns within these fires result from the clear predominance of crown fire in this ecosystem, but the product of such lethal fires is not a uniformly homogenous level of burn severity. By altering the local stand-age distribution and post-fire forest structure (Brassard et al. 2008) large wildfires such as these cause persistent changes in heterogeneity and spatial pattern across both landscape and local scales (Weir et al. 2000, Burton et al. 2008), affecting species richness and diversity of vegetation (Hart and Chen 2008), animal (Smucker et al. 2005), and invertebrate (Buddle et al. 2006) community assemblies.

When considered in terms of overstory mortality, the spatial patterns of burn severity in the six studied fires were broadly consistent with those expected from a high-severity stand-replacing fire regime, providing support for hypotheses $H_{2a}$ and $H_{2b}$. Stand-replacing patches in the six sampled fires were large, with substantial core areas, aggregated, and simple in form. Approximately 40% of the burned area experienced stand-replacing fire. Others have observed that with increasing fire size the proportion burned severely also increases, as does the mean area-weighted patch size (Cansler and McKenzie 2014, Harvey et al. 2016*b*, Collins et al. 2017). The fires studied here are of a substantial size, with only six fires burning more than 1,400,000 ha in a single year, an area equivalent to that burned by 295 fires over 26 yr in the montane study area examined by Harvey et al. (2016*b*), and over four times the area burned by 125 fires over 24 yr in a study of the northern Cascade Range (Cansler and McKenzie 2014). The proportion experiencing complete mortality was much higher than the average proportions of stand-replacing fire observed in the western United States (Cansler and McKenzie 2014, Harvey et al. 2016*b*, Collins et al. 2017); however, this proportion was smaller than the 64% observed by Ferster et al. (2016) in the same boreal plains ecoregion. Although area burned severely increases with increasing fire size, it has also been suggested that larger fires tend to have larger residual stands and a higher proportion of unchanged patches in the boreal forest (Eberhart and Woodard 1987, Madoui et al. 2010). A substantial proportion of the area within the fires (~15%) was unchanged, offering live seed sources for white spruce and a large area of residual habitats for species requiring mature forests, within the fire perimeters.

We did not consistently observe the same patterns of high-severity patches dominating the landscape when the landscape patterns of burn severity were quantified using metrics other than overstory mortality, providing support for hypothesis $H_{2c}$. Serotinous and semi-serotinous tree species, such as jack pine and black spruce, respectively, have in situ seed sources, and if viable seeds are available stand mortality is not of substantial importance to post-fire seedling recruitment. More important to such species is the level of overstory combustion (Arseneault 2001). Landscape patterns of canopy combustion, represented by CFSI, indicate that much of the area burned was of moderate severity (45%), and it was rare for severely combusted stands (CFSI >4) to be especially large, or to have core areas >600 m from lower severity burned stands, suggesting that lower densities of recruitment of serotinous species will occur only in small patches. Patterns of seed bed availability and bud-bank persistence, as characterized by BSI, are also different from the landscape pattern of stand-replacing fire derived from the overstory mortality model. Given the importance of seedbeds to interspecific competition among seedlings in the boreal region (Johnstone and Chapin III 2006), the presence of both substantial areas with remaining organic soils in peatlands as well as exposed mineral soils in uplands suggests a potentially less severe ecological outcome of burn severity that is relevant to conifer seedling recruitment (Kemball et al. 2006). The observed differences in landscape patterns of burn severity depending on the modeled metric of choice will lead to different conclusions about relative levels of burn severity and potential ecological impacts from fires, and thus, it is important to select meaningful burn severity metrics for the local fire regime when modeling spatial patterns of burn severity.

### *Vegetation, topoedaphic, and weather drivers of burn severity*

The patterns of burn severity within the six wildfires are largely explained by a combination of fuels, topoedaphic context (uplands and wetlands), and fire weather, leading us to accept our hypothesis $H_{3a}$. Previous studies have demonstrated the importance of land cover and forest type to fire frequency (Cumming 2001) and burn severity both in this ecosystem and others (Collins et al. 2007, Hall et al. 2008, Boucher et al. 2016). Our results provide further insight into the importance of stand structural and fuel loading characteristics of vegetation in driving these differences. Severity increased with stem density and median conifer CBH and decreased with increasing BA of mature trees. The effect of overstory and understory stem density on burn severity observed here is consistent with that observed in studies examining burn severity following prescribed fire and fuel treatments in the western United States; however, the same effect of pre-fire BA was not observed in this region (Prichard and Kennedy 2014, Lydersen et al. 2017). The measured differences in burn severity reflect the crowning and rate of spread potential of different fuel complexes (FCFDG 1992), suggesting that the burn severity of fires in the northwestern boreal forest may be largely due to the direct influence of stand structure on fire behavior. As burn severity was substantially explained by vegetation communities and their associated characteristics, the role of land cover in producing landscape patterns of burn severity should be controlled for when using remote sensing to monitor landscape patterns of burn severity (Collins and Stephens 2010), and trends in burn severity over time and across fires in the northwestern boreal forest. The inclusion of pre-fire vegetation and topographic variables in addition to remotely sensed severity metrics may improve estimates of burn severity and combustion by incorporating variability in drivers of severity (Barrett et al. 2010).

Detailed data about stand structure and topoedaphic context were necessary to produce robust predictions of severity. Others have found that topographic variables are important, and even dominant, drivers of burn severity (Dillon et al. 2011). In this study, wetlands consistently burned at lower severities than uplands. Site moisture likely plays a role in this effect, but wetlands also had significantly lower pre-fire overstory fuel loads and BA of trees than uplands, as is common in this region (Thompson et al. 2017). The surface burn severity-limiting effect of wetlands decreased with increasing stem density, in a gradient of increasing burn severity from open to increasingly well-stocked treed wetlands. The effect of estimated pre-fire live CBH on severity was counter to that expected, given the documented role of ladder fuels in conducting surface fires into the canopy and enabling transitions from surface to crown fire (Agee and Skinner 2005). This evidently conflated effect is explained by the wide range of vegetation communities sampled. Although lower CBHs are associated with severe fire behavior in densely stocked stands, in our dataset the lowest CBHs were in poorly stocked wetlands and mixedwood stands with suppressed, shade-tolerant coniferous understories—ecosystems that burned at lower severity.

Burn severity of a site is highly related to pre-fire, bottom-up drivers, such as fuels and topography, but weather conditions may override or shift this relationship, leading to variability in observed burn severity (Harvey et al. 2014, Birch et al. 2015, Krawchuk et al. 2016). A fire weather variable describing long-term drying and surface combustion potential (BUI; Van Wagner 1987) tempered the importance of stand structural effects on surface burn severity under more extreme fire weather in our model of BSI, but was not important in the other prognostic burn severity models. The very severe fire weather in 2014 may explain the relatively low importance of weather in determining overstory burn severity in this study. Of the 51 sampled field sites, only six burned under weather conditions with a DC of 300 or less, whereas the majority of sites burned under weather conditions with a DC of 500 or higher, indicating extreme long-term drying in deep layers of the soil (Amiro et al. 2004). Fire weather is important to wildfire occurrence and burn severity, but the restricted range of variability in weather in this study likely reduced predictive power gained from weather variables (Stocks et al. 2004, Parks et al. 2015, Krawchuk et al. 2016). This may also reflect the potentially weather-limited nature of the boreal forest fire regime, where fire occurrence is highly weather-dependent and episodic (Meyn et al. 2007, Podur and Martell 2009). The lack of variability in weather conditions prevented us from observing an overwhelming effect of fire weather on bottom-up controls on fire activity, leading us to reject our hypothesis $H_{3b}$, as we were predominantly able to detect an effect of bottom-up controls on burn severity.

Although pre-fire stand structure, composition, and topoedaphic context have a role in determining burn severity in this region (Hall et al. 2008, Ferster et al. 2016), there is still substantial variability in observed severity. A typical median level of burn severity is apparent within individual vegetation communities, but measured burn severity was quite variable, with some communities capable of burning at particularly broad ranges of severity. For example, there is substantial evidence for surface and mixed-severity fire in mixedwood and mature jack pine stands; a distinct and potentially underemphasized component of the local fire regime. The observed ranges around characteristic levels of severity for each vegetation community may be a product of pre-fire variability in stand structure and composition, and the inhibiting effect of fire weather on the influence of fuels on burn severity.

Given the importance of pre-fire stand structure and composition to burn severity outcomes, levels of burn severity falling outside of those expected for a certain stand structure and changes to both spatial and temporal patterns of fire occurrence may produce unexpected and persistent ecological outcomes from wildfires (Freeman and Kobziar 2011, Brown and Johnstone 2012). For example, some of the measured sites experienced very short-interval high-severity reburning (10 yr between stand-replacing fires), despite the substantially reduced fuel load from the previous fire. These sites had extremely low densities of any species of seedlings, a characteristic that will likely carry forward in time (Johnstone et al. 2004), potentially causing a shift away from the dense conifer forest previously found at the site. Although time since last fire and time since stand origin did not significantly contribute to the multivariable models, they were nearly significant in some cases (e.g., $P = 0.06$ MORT model). Our results do not support a clear relationship between stand age and field measurements of burn severity, but the other variables selected in the models are partially products of stand age and may simply relate more directly to burn severity outcomes (e.g., fuel load; Thompson et al. 2017).

### *Management implications*

Remotely sensed multispectral burn severity was meaningfully related to diverse field measurements of overstory and understory burn severity in the northwestern boreal study area. Managers can use existing field datasets (where available) to build region-specific models and calibrate remotely sensed severity metrics. Predicting values of diverse post-fire burn severity metrics that are tailored to specific management objectives (e.g., estimating post-fire recruitment, erosion risk, planning salvage logging, assessing prescribed burning outcomes) adds value and facility of use to these products, and recognizes the different relationships between remotely sensed burn severity metrics and overstory, understory, and mortality impacts from fire. Multispectral burn severity metrics can provide significant and rapidly available ecological information about wildfire effects in northern forests, where access for field visits is limited or expensive.

Burn severity was explained by topoedaphic context (uplands and wetlands), pre-fire stand structure and composition (vegetation communities), and fire weather. Fire managers can use this information to make rapid estimates of severity to inform management decisions about active fires where burn severity and ecological impacts are an important consideration or intended outcome, before remote sensing data are available. For example, in a vegetation community not characterized by high-severity fire, which may be a biologically relevant fire refugia, the selective burning of unburned areas for fire control (i.e., burnouts) could be avoided. Where vegetation tends to naturally burn severely, suppression efforts may be of limited effectiveness and also ecologically undesirable, and thus, suppression resources could be redirected elsewhere, if safe to do so. This understanding of the characteristic ranges of severity in different vegetation communities could also be used in combination with modeled values of burn severity field metrics to identify areas that have burned outside of expected characteristic levels of severity, and where interventions may be necessary to manage atypical ecological impacts, for example, in severely burned stands that previously experienced recurrent low-intensity surface fires or in even-aged stands reburning at very short fire return intervals.

In variable-retention forestry, it is common to intentionally leave some residual stands and trees unharvested to act as biological legacies, provide habitat, and maintain forest diversity (Gustafsson et al. 2012). In the Canadian boreal forest, this strategy often aims to replicate patterns and outcomes of wildfire, in an attempt to emulate natural disturbance (Bergeron et al. 2002, Long 2009). In light of these results, where foresters wish to mimic natural patterns of wildfires when harvesting by leaving residual forested areas and individual trees, it is valuable to understand that partial mortality and unburned residuals within the studied natural fires were not random, but instead were associated with certain vegetation communities (Upland Jack Pine and Upland Mixedwood) and stand structures (mature, open stands, with high BAs) that enabled surface fires and low-intensity burning. Such sites may represent old-growth fire refugia, within a predominantly high-intensity stand-replacing fire regime. The landscape patterns produced by retention areas will differ from those produced by natural wildfires if they are not located and planned in a manner that considers the causes and probability of natural residuals (Dragotescu and Kneeshaw 2012).

### *Limitations and future research*

Although we were successful in producing prognostic and diagnostic models of burn severity, there are some limitations to the conclusions that can be drawn. The NARR data product used for fire weather was downscaled significantly from its original resolution. This product also integrates modeled precipitation over northern North America, rather than observed precipitation (Mesinger et al. 2006). Although these limitations may have reduced the importance of fire weather in the explanatory models of burn severity, the low weather station density in northern Canada justified this choice. A larger sample of sites, or sampling in fires that burned in other, non-drought, years may be required to better characterize the influence of fire weather on burn severity in the study area. The multispectral images used to produce remotely sensed burn severity in this study were predominantly from satellite overpasses early in the growing season. Although spring and early summer imagery may over-estimate burn severity, the lack of cloud and smoke-free images in later seasons and years necessitated the choice of these images. The multivariable explanatory models of burn severity did not directly consider topography or elevation, which are known bottom-up controls of wildfire spread and severity (Birch et al. 2015, Krawchuk et al. 2016). The landscape of the study area is primarily composed of gently rolling plains; however, more subtle topography such as transition zones between wetlands and uplands may still have affected burn severity in ways that are not captured in these models. Finally, the pre-fire stand structural characteristics of understory stem density and live conifer CBH were measured in the year after fire. There is a possibility that understory trees and lower branches were fully consumed and thus not measured, thereby leading to underestimates of understory density and overestimates of CBH, but this type of error is likely minimal. Even in the most severe burns, we were able to distinguish remaining stems of consumed saplings, and it is generally possible to determine whether branches were alive or dead prior to a fire's arrival due to the persistence of bark and less deep charring on live wood.

Future research in the northwestern boreal forest could include explanatory spatial models of burn severity, representing the in-stand (patch- and plot-level) drivers of burn severity identified here with mapped fuels, topography, and weather, potentially improving predictive ability across the diverse boreal landscape. The inclusion of additional fires for spatial analysis of landscape patterns of burn severity in the northwestern boreal forest would provide a more robust dataset and would permit a broader characterization of typical patterns of burn severity across fire sizes and between fires in this region. In addition, future research into drivers of burn severity using structural data from paired pre-fire and post-fire sample plots could more robustly characterize these relationships than is possible with the parameters collected post-fire in this study.

## Conclusion

Overstory and surface burn severity in the northwestern Canadian boreal forest was significantly explained by the multispectral remotely sensed metrics of burn severity, in an ecologically diverse sample of burned sites. Burn severity metrics that were relativized to pre-fire conditions (RdNBR, RBR) were more related to observed burn severity than non-relativized metrics (dNBR). Burn severity was adequately predicted by pre-fire forest characteristics of stand structure, fuel load, species composition, and topoedaphic context. Although fire weather was also related to observed burn severity, this was only significant in determining surface burn severity and we did not observe an overwhelming effect of extreme fire weather on bottom-up drivers of burn severity, despite the drought-driven nature of the sampled fires. Differences in stand structure and fuel loading translated to different characteristic levels of burn severity within vegetation communities, which explains in part why there can be considerable variation in the degree of burn severity within northwestern boreal forest wildfires. This variability influenced the range and landscape patterns of burn severity observed, despite the dominance of stand-replacing crown fire in this ecosystem. High- and moderate-severity burned patches were large, simple in form, and made up the majority of area burned, whereas low-severity burned patches were small and complex. Despite these general trends, landscape patterns of burn severity differed depending on the modeled field metric of burn severity assessed, and conclusions about characteristic spatial arrangement and overall "severity" of landscapes were not transferrable between burn severity field metrics.

Differences in severity observed between uplands and wetlands and the influence of stand structure and composition on burn severity in this study highlight the importance of considering wetlands, and all major vegetation communities, when attempting to capture the range of burn severity. The strong association between land cover and post-fire burn severity should be controlled for when using remote sensing to monitor landscape patterns and trends in burn severity over time, and across multiple fires. The prognostic models built using continuous stand structural variables demonstrate that burn severity in the northwestern boreal forest is predictable and characteristic of different ecotypes. Relationships between remotely sensed burn severity and ground observations of severity in this region allow forest and fire mangers to address refined management goals, such as the estimation and management of post-fire recruitment, assessment of prescribed burning outcomes, post-fire erosion control, salvage logging planning, and assessment of wildfire effects on habitat for wildlife. The influence of pre-fire variables on burn severity could also be incorporated into forest harvesting, and prescribed and active fire management by choosing to leave residuals where they may naturally occur, preserving probable fire refugia. The relationships between burn severity, and pre-fire drivers and fire weather presented here offer potential areas for future exploration to improve spatial modeling of burn severity and the scaling of these effects from in-stand to landscape levels.

## Acknowledgments

This research was funded by the Natural Sciences and Engineering Research Council of Canada (Funding Reference Number: CGSD3-471480-2015) and the Government of the Northwest Territories. Parks Canada Agency and Jean Morin provided in-kind support. We thank Xinli Cai, G. Matt Davies, Kathleen Groenewegen, Derek Hall, Koreen Millard, and Doug Stiff for assistance in the field. We also acknowledge Xianli Wang and Sean A. Parks for their assistance with fire progression mapping. Scott E. Nielsen and two anonymous reviewers provided helpful feedback and comments on earlier versions of this manuscript, and greatly improved this work.

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## Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2128/full