Running Title: Drivers of wildfire susceptibility in Canada

Factors influencing national scale wildfire susceptibility in Canada

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Abstract

Wildfires are expected to increase as a result of climate change. In order to effectively manage and monitor climate-induced changes in Canadian forests, a national-scale understanding of factors influencing wildfire susceptibility is necessary. The goal of this study is to better understand factors influencing large area wildfire susceptibility in Canada. Using year 2000 Canadian land cover data, we identify locations that burned before and after 2000. Pre- and post-fire landscape patterns were assessed and regression tree analyses were used to identify factors influencing national-scale fire susceptibility. Land cover composition, forest pattern, elevation, and anthropogenic influences were quantified for both pre- and post-fire environments. We examined recovery of forest pattern following wildfire events and derived a large-area fire susceptibility model using decision tree classification. Our results indicate that 11.88% of forested ecozones were impacted by large fires. The majority of large wildfires occur in coniferous forests characterized by high forest cover (greater than 45%), few forest patches, large mean forest patch area, and fragmentation-limited forest. Forests occurring at low to intermediate distances from populated places (50 to 150 km) and roads (12 to 72 km) experienced unexpectedly high amounts of fire, as did lower elevation forests. After fire, percentage forest cover, number of forest patches, forest patch size, and proportion forest patches regenerated to pre-fire forest pattern conditions within approximately 20 years. Anthropogenic influences on wildfire susceptibility indicate that human activity still dictates national fire regimes. Additionally, knowledge of space-time patterns of fire-landscape interaction and landscape pattern regeneration provides useful baselines for future comparisons with responses to climate change.

Keywords: Spatial Temporal Pattern of Wildfire, Burn Susceptibility, Landscape Pattern, Anthropogenic Influence, Forest Fragmentation, Regeneration

1. Introduction

Wildfire is a dominant natural forest disturbance in Canada, burning approximately two million hectares of forest annually (Stocks et al., 2002). While short-term effects of fire include changes to landscape pattern (Hayes and Robertson, 2009), wildlife habitat (Emlen, 1970; Whelan et al., 2002), soil (Giovannini et al., 2001), and air quality (Hardy et al., 2001), the long-term effects drive ecological processes (Whelan, 1995) and impact carbon cycling (Kasischke et al., 1995).

Wildfire regimes are changing due to a combination of climate change and fire suppression. A trend in longer, more severe fire seasons has been attributed to climate change resulting in increased fire occurrence (Stocks et al., 1998; Flannigan et al., 2000) and area burned (Flannigan et al., 2005). In heavily-managed forest areas, long-term fire suppression has resulted in unnatural fuel accumulation, ultimately leading to larger, more severe fires (Keeley et al., 1999). Continued changes to modern wildfire regimes will introduce unanticipated fire activity throughout Canada. Therefore, it is necessary to characterize landscape-scale patterns related to wildfire as a means of understanding potential fire susceptibility and to create a baseline for assessing future change in forest processes.

There are many factors that impact space-time patterns and likelihood of fire. Fire occurrence and spread can be attributed to weather and climate (Flannigan and Harrington, 1988), landscape fuel conditions (Romme, 1982; Finney, 2001), ignition agents (Malamud et al., 2005), and human influence (Rollins et al., 2001). There is also a stochastic aspect to wildfire associated with variability in local weather conditions (e.g., surface moisture and wind speed (Bessie and Johnson, 1995)) and successful ignition events, both of which are difficult to predict. Despite fire clustering due to lightning strikes (Diaz-Avalos et al., 2001; Podur et al., 2003) and human-caused ignitions (Cardille et al., 2001; Yang et al., 2007;), predicting exact locations of ignitions is not possible. The human impact on fire location is primarily a product of population presence and accessibility. The use of proximity measures in assessing anthropogenic influence on wildfires has been successful (Yang et al., 2007).

The relationship between landscape fuel condition (i.e., vegetation type and pattern) and wildfire processes is complex and cyclical (Turner and Romme, 1994). The spatial heterogeneity of landscape characteristics, such as vegetation species, forest age, and landscape pattern (e.g., fragmentation) are often at least partially a result of wildfire. Forest fire processes and susceptibility are also influenced by forest pattern (Rollins et al., 2002). The spatial arrangement of vegetation on the landscape relates to fire spread under ordinary weather conditions (Brown, 1985). For instance, highly connected forest patches aids fire spread (Turner and Romme, 1994), whereas patches of irregular shape reduce the rate of fire spread (Ryu et al., 2007).

When spatial processes cannot be measured explicitly, which is the case for large-area forest processes, characterizing spatial patterns through time is a useful proxy (Getis and Boots, 1978; Haining, 2003). Spatial pattern can be quantified in terms of composition or configuration. Composition metrics are used to measure the number of classes or patch types in a landscape, the proportion of each landscape class, or the diversity of classes as described by evenness or richness (Gustafson, 1998). Configuration metrics are used to quantify spatial pattern and feature arrangement in a landscape (Gustafson, 1998). Both composition and configuration measures are critical for characterizing landscape pattern.

The development of national-scale datasets for wildfire (Stocks et al., 2002) and forest pattern (Wulder et al., 2008b) creates a unique opportunity to integrate and build national-scale assessments of factors that influence and are generated by fire processes. For instance, in Canada the Earth Observation for Sustainable Development of Forests (EOSD) product (Wulder et al., 2008a) was created to characterize forest cover and has been used to quantify forest composition in the year 2000.

Our research goals are to provide a national-scale assessment pre- and post-fire landscape conditions and to quantify factors that influence national scale wildfire occurrence in Canada. We will meet these goals by addressing the following objectives.

1. Quantifying the composition of land cover in pre- and post-fire locations within Canada's forested ecozones.
2. Characterizing spatial pattern of forests and abiotic variables (proximity to populated place, proximity to roads, and elevation) associated with pre-fire locations.

3. Quantifying temporal change in the spatial pattern of forests, as forest regenerate post wildfire.
4. Evaluating how covariates relate to national scale patterns of wildlife.

2. Study area and data

2.1. Study area

The 10 forested ecozones of Canada constitute approximately 700 million ha and define the extent for this study (Fig. 1). An ecozone is “an area of the earth’s surface representative of large and very generalized ecological units characterized by interactive and adjusting abiotic and biotic factors” (Ecological Stratification Working Group, 1995). The majority of wildfires in Canada occur within forested ecozones, often with individual ecozones exhibiting distinctive fire occurrence and area burned (Stocks et al., 2002).

2.2. Wildfire data

The Canadian National Fire Database (NFDB) is a spatial database of wildfires in Canada, aggregated by the Canadian Forest Service from the 13 provincial and territorial fire management agencies (for details see Stocks et al., 2002). The NFDB-polygon database consists of vector polygons that represent the fire perimeter as determined by satellite or aerial imagery, aerial observation, or ground mapping using global positioning system units. Ancillary information about the fire is often included, such as start and end date, size, and cause. Completeness of the NFDB varies between agencies and years. Fire polygons were available for all regions from 1980, but were unavailable for some maritime agencies post-2000, although most provinces contributed data until 2005, 2006, or 2007. We have chosen the temporal range of 1980 to 2007 to optimize completeness and consistency.

Only fires greater than 200 ha in size were included in this analysis (Fig. 1). Larger fires are more accurately mapped due to their size and longer duration, making fire data consistent post-1975 with the emergence of remotely sensed data (Murphy et al., 2000). A 200 ha fire size has been the lower limit of the Large Fire Database used in numerous wildfire studies in Canada (Amiro et al., 2001; Stocks et al., 2002; Parisien et al., 2006). Additionally, fires greater than 200 ha account for approximately 3% of ignitions but about 97% of area burned in Canada (Stocks et al., 2002).

2.3. Land cover data

Land cover information was obtained from the EOSD forest product (Wulder et al., 2008a; Wulder et al., 2008b). Land cover conditions in the EOSD are characterized for circa 2000 using over 480 Landsat-7 ETM+ scenes from 1999 through 2002, with 90% of coverage occurring in the year 2000. The EOSD has a spatial resolution of 25 by 25 m. Given the hierarchical classification system of the EOSD (Wulder and Nelson, 2003) the original 23 classes were collapsed into forest and non-forest classes to enable a national-scale assessment of forest pattern. The large-spatial extent, small-spatial grain, and focused-temporal period make the Landsat-derived EOSD land cover product ideal for use as a baseline land cover assessment in Canada.

2.4. Abiotic covariates

Abiotic factors that influence wildfire ignition and spread were included in the analysis. Anthropogenic influences included proximity to road and proximity to populated places, while natural influences were measured using elevation. All abiotic datasets were summarized at a 1 km grain corresponding to the forest pattern coverages. Proximity to road was created by calculating Euclidean distance to road of any size using the 2008 road network file (Statistics Canada, 2008). Proximity to populated places was created similarly, but using persistent nighttime light obtained from the DMSP Operational Linescan System instead (Wulder et al., 2011).

3. Methods

3.1. Composition distributions

To quantify the composition or amount of each land cover in pre- and post-fire locations across Canada we used the year 2000 as a baseline, as the EOSD land cover represents conditions in circa 2000. The baseline represents pre-fire land cover conditions for locations where fires burned after the year 2000 due to EOSD imagery acquisition before fire. Conversely, the baseline represents post-fire conditions for locations where fires burned prior to 2000 due to imagery acquisition after burning. As such, land cover composition is determined for only the year 2000. We define pre-fire conditions for fires between 2003 and 2006 and post fire conditions for fires that burned from1980 to 1999. We chose 2003 as the pre-fire initial year to reduce possible commission and omission errors in the fire data as the EOSD data acquisition ranged between 1999 and 2002.

Frequency distributions of land cover composition were generated for Canada's forested ecozones, for pre-fire locations, and for post-fire locations. Thirteen land cover classes were assessed: broadleaf dense, broadleaf open, broadleaf sparse, coniferous dense, coniferous open, coniferous sparse, mixedwood dense, mixedwood open, mixedwood sparse, wetland treed, non-treed, non-vegetated, and other. Percentage change in land cover composition following fire was calculated for each class to determine which types of landscape burned and to characterize the resultant landscape nationally.

3.2. Configuration and covariate distributions

Several landscape pattern metrics were examined at a 1 km by 1 km grain using a reclassification of the EOSD land cover into three classes: forest, non-forest, and other (See Wulder et al., 2008a). The 1 km by 1 km grain enabled forest pattern to be assessed using 1600 25 m by 25 m spatial units. While the selection of grain size necessitates a component of subjectivity, the 1 km representation enabled robust assessment of landscape pattern due to sufficient sample size. The 1 km landscape also provides a grain size that can be integrated with all other spatial data sets and provides sufficient detail for characterizing national-scale trends. Landscape metrics are useful in quantifying spatial patterns due to their computational simplicity, ease of implementation, and broad-scale applicability (McGarigal and Marks, 1995; Frohn, 1998; Cardille and Turner, 2002). Wildfire influence on spatial pattern has been increasingly documented using landscape pattern metrics for small spatial extents (Lloret et al., 2002; Ryu et al., 2007; Montane et al., 2009; van Leeuwen et al., 2010); however, no national-scale studies exist.

For this study, we chose four metrics that relate landscape pattern to processes of fire (Levin, 1992; Li and Wu, 2004): percentage forest cover (%), number of forest patches, mean forest patch area (ha), and proportion forested patches, all calculated per 1 km cell. Percentage forest cover describes the amount of a given cover type within a cell and is a simple way to quantify pre- and post-fire changes in amount of forest and additionally may be used to characterize forest evenness or dominance (Botequilha Leitao et al., 2006). Number of forest patches and mean patch area represent landscape heterogeneity, fragmentation, and contiguity. Forest fragmentation and complexity has been used to study fire spread (Turner and Romme, 1994; Ryu et al., 2007), prevention

(Finney, 2001), and return interval (Roberts, 1996), and may have important implications for both pre- and post-fire landscapes. Finally, quantifying forested patches provides context for interpreting fragmentation (Wulder et al., 2008b). For example, a highly fragmented forest that is surrounded by a non-fragmented landscape can be differentiated from a highly fragmented forest that is surround by a highly fragmented landscape.

To characterize the impact of forest pattern and abiotic variables on whether a location has a large fire, relative frequency distributions of landscape pattern metrics and abiotic covariates were generated for all locations and for locations that burned after 2003. Differences between relative distributions for all locations and burn locations were calculated and trends in forest pattern and covariates identified. .

3.3. Forest pattern temporal analysis

Forest spatial pattern following fire was also analyzed. Pre- and post-fire landscape conditions were separated by year of fire to examine how forest pattern leads to fire, how a burn alters forest pattern, and the patterns associated with vegetation re-growth. Frequency distributions of landscape pattern metrics were created for locations with similar “time-since-fire” or “time-until-fire” characteristics. A three-dimensional histogram facilitated the representation of a multi-dimensional relationship.

3.4. Decision tree model

Modeling the susceptibility to wildfire of a given landscape requires both *a-priori* knowledge of biotic and abiotic factors influencing fire and the relative importance of each factor. For this study, we classified locations of wildfire ignition density using landscape pattern metrics and abiotic covariates in a decision tree model at the national level. Decision trees recursively partition large datasets into classes based on a set of hierarchical rules (Breiman et al., 1984) and ranks the relative importance of covariates on classification. Given the broad spatial extent of the study, relative ranking of variables that can be mapped nationally is an appropriate level of detail for analysis. As well, the variable ranking and thresholds used by regression trees to classify fire ignition density can be easily applied in a management context when broad-scale mapping is required for strategic-level decision making.

Locations were categorized as fire or non-fire. Due to the absence of post-1999 fire data, Nova Scotia, New Brunswick, and Newfoundland were excluded from the decision tree analysis. The number of burned pixels (n = 79,067) was much fewer than non-burned pixels (n = 5,845,945), indicating a case of class imbalance. Imbalanced datasets may occur with environmental problems (i.e., detection of oil spills (Kubat et al., 1998)) and can result in the classifier having a bias towards the majority class (*non-fire* in this case). Under-sampling of the majority class is suggested to overcome imbalance (Domingos, 1999; Japkowicz and Stephen, 2002). The non-fire class was under-sampled to 1.5 times the fire class size, which maximized non-fire user accuracy and exhibited consistent high fire producer accuracy. Under-sampling was conducted using an ecozone-stratified random sample without replacement.

All four landscape pattern metrics and three abiotic covariates were used in the decision tree analysis along with ecozone and total fire count per pixel. All available data were subset into 70% training data for tree creation and 30% test data for tree validation. Monte Carlo simulations and resultant decision trees were completed to randomize which *fire* pixels occurred in training and test data. Twenty simulations were determined to be acceptable as all decision trees had similar leaf nodes. Final decision tree values were selected by using the simulation with highest *non-fire* user accuracy.

The accuracy of the fire susceptibility model was evaluated. A confusion matrix was created to assess the decision tree accuracy for predicting fire location. User and producer error was calculated for both *fire* and *non-fire* classes and the Kappa coefficient (Cohen, 1960) was used to assess overall accuracy. While overall accuracy is important, misclassification was anticipated to occur as forest with potential to burn has not yet. For this reason, high non-fire user accuracy may be more relevant in assessing model accuracy as it indicates lower non-fire commission error, or few fires accidentally classified as non-fire.

Model accuracy was also evaluated by ecozone using historic burn areas. Ecozone-specific values of fire-susceptible area and total area burned between 1980 and 2007 were standardized by total ecozone area. A Pearson correlation was used to assess the relationship between historic area burned and model susceptible area.

4. Results

4.1. Composition distributions

Across Canada's forested ecozones 44.07% are classified as *treed* and have a relatively equal distribution of broadleaf, coniferous, mixedwood, and wetland trees (Fig. 2). In comparison, there is slightly less *non-treed vegetation* area (i.e., shrubs, bryoids, wetland) (39.49%), and *non-vegetation* classes and *other* consist of 12.85% and 3.59% of the study area, respectively.

The fraction of forested ecozones burned by large fires between 2003 and 2007 is 1.81%. Large forest fires predominately burned in *treed* locations, and coniferous forest accounts for over half of the large burns (55.15% of area). Dense and open coniferous stands burned at a greater frequency than sparse stands, and sparse conifers are also a large fraction of the pre-burned landscape. Broadleaf, mixedwood, and wetland treed forests comprise a small percentage (12.03%) of the area burned by large fires. *Non-vegetation* and *other* classes have few large burns.

Between 1980 and 1999, 11.88% of forested ecozones was burned by large forest fires. The post-fire land cover composition is similar to the typical distribution of classes in forested ecozones (Fig. 2a). The largest reduction in land cover occurs within the coniferous classes (34.19% total decrease) with open stands exhibiting the largest post-fire decrease, followed by dense stands, and then sparse. Landscapes following burn tend to be higher in *non-treed vegetation* (22.22% increase) and *non-vegetation* classes post-fire. A 3.08% composition change is seen in all non-conifer forest classes combined.

4.2. Composition and covariate distributions

Fires occurred in all landscapes regardless of the amount of forest (Fig. 3). Areas with a high percentage of forest cover, fewer patches, and small patch area are most frequently associated with burns. Small patches are more plentiful in Canada, while large patch areas are more at risk to fire. Fire occurred consistently through all landscapes regardless of forest to landscape fragmentation ratio. Burns preferentially occurred in less fragmented forests.

Pixels representing areas 50 to 150 km away from populated places and within 12 km to 72 km of a road contain the highest likelihood of burning (Fig. 4). While fires were most frequent near roads, occurrence is less than expected based on random fire processes. Fires also burned most frequently at elevations between 290 m and 580 m with a negative preference in the lowest elevation range: 1–280 m. Most large fires were restricted to elevations below 1000 m.

4.3. Forest pattern temporal analysis

Landscape pattern metrics show change and gradual return to pre-fire distributions after large fire events (Fig. 5). Not surprisingly, percentage forest cover is highest prior to burn, and post-fire landscapes have relatively little forest cover. Number of patches increases following burn and mean forest patch area shifts from a bi-modal distribution with equal large and small patch landscapes to a landscape dominated by small-patches. The proportion of patches that are forest is found to have changed from a weak bi-modal distribution pre-fire to one of increased fragmentation compared to surrounding landscape. While the rate of change varied by metric, all frequency distributions had similar shapes to pre-fire landscapes at 19 to 20 years after burn. As such, after a fire it takes about 20 years for forest pattern to return to pre-fire conditions.

4.4. Decision tree model

By rank, factors that most influenced the presence of forest fires include: mean forest patch size, proximity to populated place, proximity to road, and elevation (Fig. 6). Fires were most likely to occur when patch area was greater than 0.64 ha and the location was further than 61 km from populated places, lower than 1105 m and within 98 km of a road.

An error matrix determined that the total accuracy for the decision tree model is 61.30% with a Kappa coefficient of 0.27 (Table 1). The producer accuracy of *fire* is 82.14%, and *non-fire* is 47.34%. The user accuracies of *fire* and *non-fire* are 50.98% and 79.90%, respectively. There are low errors of commission for the *non-fire* class (20.10%) and low errors of omission for *fire* class (17.86%). The largest misclassification occurred when *non-fire* pixels were incorrectly classified as *fire*.

Application of Pearson correlation indicates a relationship between susceptible area and actual area burned by ecozone (r = 0.75) (Fig.7). The decision tree model correlates best with area burned in the Taiga Plains, Boreal Shield (both large area susceptible, large area burned), and Montane Cordillera (low area susceptible, low area burned). The model correlates poorly with Hudson Plains and Pacific Maritime (high area susceptible, low area burned).

5. Discussion

Identifying landscape characteristics that precede large fire events allows us to understand the conditions present at fire-prone locations. In Canada, large fires most often occur in coniferous forests of all densities, and fire is important in the evolutionary history of certain conifers (Moore et al., 1999). The fire-dominated northern forested ecozones possess an increased proportion of coniferous forest and large fires in these areas are often not suppressed (Ward et al., 2001). In addition, there is a high occurrence of crown fires in conifers due to low crown moisture (Van Wagner 1977). Wildfires that occur in non-treed vegetation (grass or shrubland) are typically collateral damage from forest-centric fires; however, a small number of grassland-centric fires do occur in Canada (Bond and van Wilgen, 1996).

Landscape pattern metrics were related to fire processes to characterize national-scale trends. As expected, burns were most often associated with landscapes that had a high percentage forest cover. Similarly, a low number of forest patches and larger patch size enables large fires to propagate though a landscape easily (Turner and Romme, 1994). The sharp decline in frequency of wildfires as the number of patches increased indicates how critical low patch numbers are for large fire occurrence. While small patch areas were most frequently associated with burns, this is due to the large number of landscapes in Canada with small patch area. When the difference in relative frequency is considered, the susceptibility of larger patch areas is highlighted. Forests with less fragmentation than the surrounding landscape are also at greater risk. Our findings are consistent with the predisposition for fire in non-fragmented landscapes that has often been observed at regional-spatial scales (Ryu et al., 2007).

Examination of the relationship between fire and anthropogenic covariates indicates that settlement and transportation networks (i.e., accessibility) are important drivers of fire susceptibility. Large fires occur more often than expected between 100 and 300 km from populated places. Areas less than 100 km away are less likely to burn due to increased pressure for suppression when close to human interests. Proximity to roads is similar, as fires close to roads are a priority to manage due to increased accessibility aiding the suppression effort. The difference in magnitude observed between road and populated place proximities can be explained as roads are more wide-spread than populated places. By connecting populated locations, roads themselves do not correspond as well with highly protected human interests, but do reflect increased accessibility to areas. The reduced occurrence of fire starting at 84 km to roads and 400 km to populated place is likely an artifact of reduced road network occurrence in low burn areas within the northern Taiga Shield and Hudson Plains. In these regions natural influences such as wetlands are controlling landscape fragmentation (Wulder et al., 2011) and may be decreasing fire susceptibility.

The final abiotic covariate examined, elevation, likely influences fire frequency through surface moisture and species composition, and fuel moisture has been demonstrated to increase with elevation (Hayes, 1941). While the relationship between elevation and large fires will vary at a regional scale, the observed 1000 m threshold is similar to limits found in other studies (1500 m in the Washington Cascades (Camp, 1999); 800 m in Alaska (Kasischke et al., 2002); 1000 m in the Mediterranean (Diaz-Delgado et al., 2004)). The reduced occurrence of fire at the lowest elevations can be explained in part by the increased wetland prevalence and decreased fire occurrence in the Hudson Plains.

Using a decision tree model to rank the relative importance of national-scale variables, we found mean forest patch area to be the most influential factor on large fire susceptibility. Large fires are less likely to occur in forest patches smaller than 0.63 ha. Northern regions with sparse tree coverage, regions of high elevation, and regions where fire has recently occurred are less susceptible. The decision tree thresholds for proximity to populated place and elevation corroborate findings observed in the relative frequency distribution analysis. Reduced fire susceptibility with far distances to roads, though counterintuitive, is accounting for the lightly burned northern Taiga Shield and Hudson Plains without removing the heavily burned northern Taiga Plains and Taiga Cordillera. Considering similarities in fragmentation drivers in these ecozones (Wulder et al., 2011), the importance of roads on fire is emphasized and may indicate an anthropogenic influence on fire activity in the Taiga Plains and Taiga Cordillera.

At the national scale examined in this study, anthropogenic activities have a strong influence on large fire susceptibility. The low number of forest pattern variables in our model of fire susceptibility drivers is perhaps not unexpected. This model has been created for a national extent to assess large-scale drivers of wildfire, but the relationship between forest pattern and fire process varies spatially. There are numerous fire-behavior regimes within Canada (Stocks et al., 2002; Parisien et al., 2006; Gralewicz et al., in press) and the different landscapes have adapted to each regime, influencing inter-fire landscape variation. Additionally, intra-fire variation may result from collateral damage of large fires or extreme fire-weather causing burn of non-normal landscapes.

Anthropogenic influence appears to shape fire regimes at many spatial scales. Our drivers of fire susceptibility model has similarities with a study in the Ozarks Highlands Region of Kansas (Yang et al., 2008), an extent approximately 1.1% the size of our study area. Despite the differences in scale, both studies found human accessibility to be the primary driver of burn susceptibility and fire occurrence. Biotic and topographic factors were considered secondary. Elevation, however, was a greater descriptive factor in our model than Yang et al. (2008), likely due to our larger study area.

While understanding conditions that lead to fire are important for modeling, management also requires knowledge of the landscape's response to forest fire. Large compositional changes occur after wildfire. For instance, in Canada, fire reduces the amount of coniferous forest in the short term. Conifer regeneration is expected given their co-evolution with fire, although regeneration times will vary by location (Shatford et al., 2007) and are dependent on fire severity (Key and Benson, 2005).

While conifers may be evolutionarily adapted to high fire regimes, increased disturbance from fire could transform inexperienced environments (broadleaf and mixedwood) to non-treed or non-vegetation dominated regions (Ogden et al., 1998; D'Antonio and Vitousek, 1992). Many non-conifer forest classes (broadleaf, mixedwood, wetland) experienced less than 2% change in amount of forest following burn. Natural fire regimes in these regions likely involve fewer fires due to climatic controls. Increased anthropogenic-related ignitions or more severe fire weather, however, heighten fire risk for non-conifer species.

Wildfire also changes landscape pattern by increasing fragmentation: percentage forest cover decreases, number of forest patches increases, mean forest patch area decreases, and forest to landscape patch ratio increases. The ability to accurately predict regeneration times for composition and configuration is essential for forest management, carbon modeling, and habitat analysis. Previous studies have examined regeneration compositionally as vegetation regeneration with normalized difference vegetation index (NDVI; Goetz et al., 2006) or net primary productivity (Amiro et al., 2000), and indicated regeneration time as five years or 20–30 years, respectively. Both of these measures take non-treed vegetation into account and reflect establishment of pioneer species and saplings. In this study, the regeneration of forest pattern to pre-fire levels took approximately 20 years. Forest cover increased, patches decreased in number, patch area increased, and forests became less fragmented. The 20–30 year recovery period (Amiro et al., 2000) matches our response in percentage forest cover. The short five year recovery period has been justified by accounting for spatial variability in burn severity (Goetz et al., 2006), which has demonstrated influence in vegetation recovery post-fire (Diaz-Delgado et al., 2003).

While national studies are important in understanding the overarching, broad-scale controls and results of wildfire, the existence of multiple fire-behavior regimes within Canada (Parisien et al., 2006) necessitates multiple management strategies. Wildfire expectation and suppression requires region-specific analysis, and fire management must be tailored to unique regions. The drivers of fire susceptibility model and map should be used as preliminary and exploratory tools. Local-level susceptibility would include regionally specific expectations of fire behavior, anthropogenic influence, and ignitions, as well as temporally specific estimates of fuel, moisture, and fire weather (Wotton, 2009).

The model, and thus the drivers of fire susceptibility, is considered accurate based on a producer accuracy of *fire* of 82%. The poor producer accuracy of *non-fire*, poor user accuracy of *fire*, and overall poor Kappa coefficient can be explained by the absence of future fire information. The model was especially accurate for the Taiga Plains and Montane Cordillera. Taiga Plains is an ecozone where large fires occur infrequently, whereas the Montane Cordillera experiences frequent, smaller fires. Conversely, the model performed less well for the Hudson Plains and Pacific Maritime. Both Hudson Plains and Pacific Maritime have much less fire than the model anticipated, likely due to the amount of wetlands and amount of precipitation, respectively. The reduced susceptibility of mountainous, Cordillera ecozones compared to boreal forest ecozones is also evident, similar to findings by Parisien et al. (2006).

6. Conclusions

Using a national spatial extent and a spatial resolution of 1 km, we found that fire burned predominantly in coniferous landscapes. Landscape pattern metrics and abiotic covariates were used to demonstrate that large fires were dominant in non-fragmented landscapes and at intermediate distances to anthropogenic influence. Fire caused reduced percentage forest cover and increased fragmentation, with regeneration to pre-fire landscape conditions taking approximately 20 years. Finally, a model of national-scale drivers of susceptibility was created for Canada and identified non-sparse forest, anthropogenic proximity, and elevation as influential factors. Fire severity can influence all of these results, yet the NFDB does not currently contain accurate information on fire severity. We suggest emphasis should be put on improving the estimation of fire severity with remote sensing techniques (e.g., Soverel et al., 2010).

Development of this national model of fire drivers provides a starting point for susceptibility modeling in Canada and emphasizes the influence of human activity on fire regimes. The distributions of land cover indicate that fire has broadly shaped the land cover composition and that fire due to climate change or anthropogenic activities may negatively affect non-fire adapted forest. Additional insight has been given into landscape pattern regeneration after fire. This work provides a baseline for comparing future climatic influence on fire and landscape behavior. Future research should examine regional and multi-scale trends, fire severity, and extreme fire weather impacts on landscape pattern and susceptibility.

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Table 1. Error Matrix for accuracy assessment of the drivers of wildfire susceptibility decision tree. Number of pixels classified as *fire* or *no fire* by the decision tree are compared to corresponding outcome in reality (fire or no fire).

	Decision Tree Results			
Reference Data	**No Fire**	**Fire**	**Producer Accuracy**	**Errors of Omission**
No Fire	16592	18454	*47.34%*	*52.66%*
Fire	4172	19193	*82.14%*	*17.86%*
User Accuracy	*79.90%*	*50.98%*		
Errors of Commission	*20.10%*	*49.02%*		

Fig. 1. The ten Canadian forested ecozones and large wildfires (greater than 200 ha) between 1980 and 2007.

Fig. 2. (a) Land cover composition distribution for all forest ecozones, pre-fire locations, and post-fire locations in Canada. Pre-fire locations refer to landscapes where fire burned between 2003 and 2006 (year 2000 conditions represent those preceding fire). Post fire locations refer to landscapes where fire burned 1980-1999. (b) Post-fire percentage change by land cover composition class.

Fig. 3. (a) Relative frequency distribution histograms for landscape pattern metrics in all forested ecozones (grey) and pre-fire locations (green; where landscape represents conditions that burn 2003-2006). (b) Relative frequency difference between all forested ecozones and pre-fire locations for the same landscape metrics.

Fig. 4. (a) Relative frequency distribution histograms for abiotic covariates in forested ecozones (grey) and pre-fire locations (green; where landscape represents conditions that burn 2003-2006). (b) Relative frequency difference between forested ecozones and pre-fire locations for the same abiotic covariates.

Fig. 5. Three-dimensional histograms of landscape pattern metric distribution by year of fire for (a) percentage forest cover, (b) number of forest patches, (c) mean forest patch area, and (d) proportion of all patches that are forest. Green denotes pre-fire conditions: landscapes that burned 2003-2006.

Fig. 6. Decision tree model for identifying national drivers of large fire susceptibility in Canada. Decisions are made proceeding to the left if the statement is true.

Fig. 7. Ecozone-based comparison of area previously burned by large wildfires (between 1980 and 2007) and area susceptible to fire (determined by national-scale drivers). Area is represented as percentage of total ecozone area.

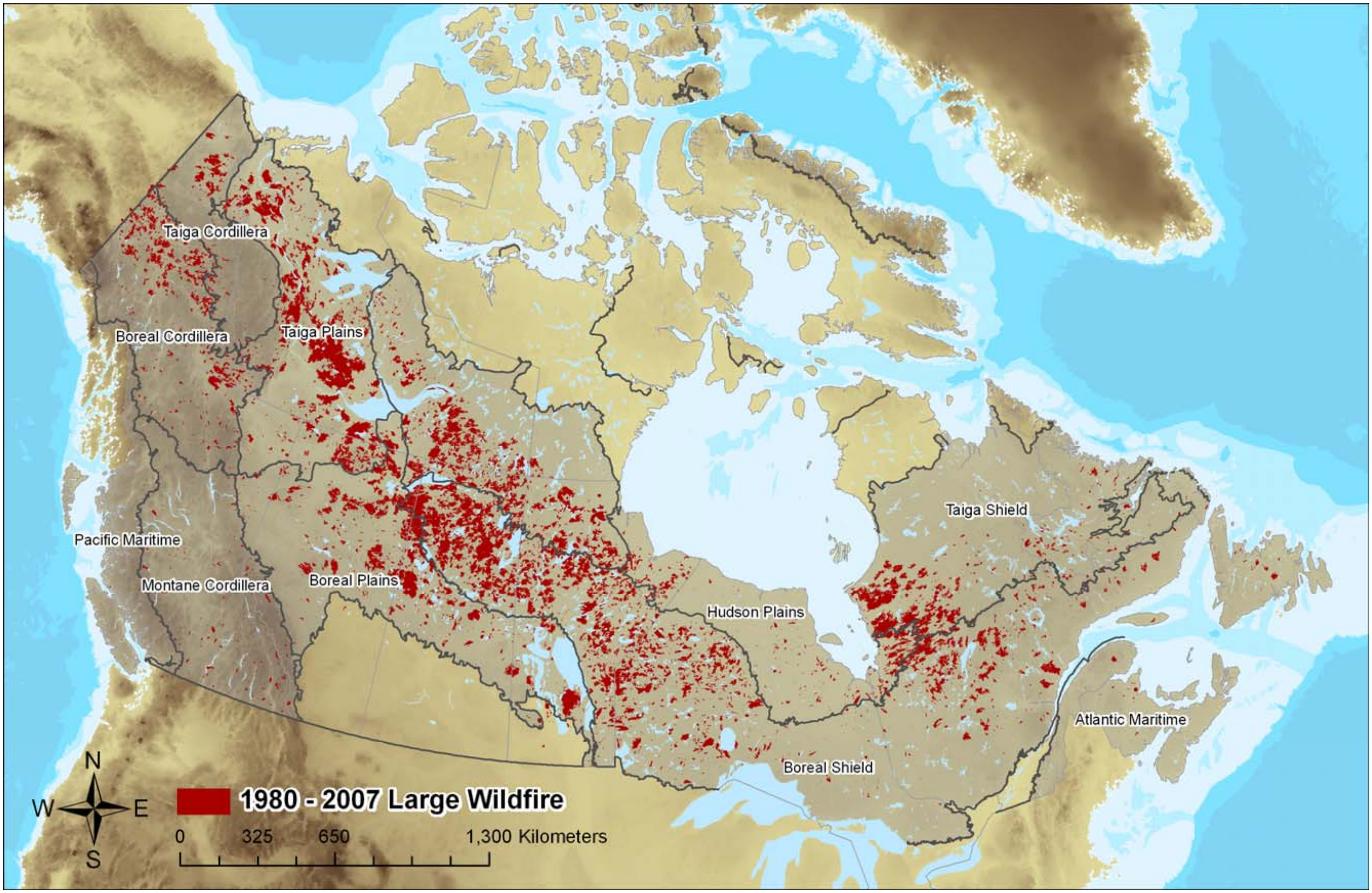

Taiga Cordillera
Boreal Cordillera
Taiga Plains
Pacific Maritime
Montane Cordillera
Boreal Plains
Hudson Plains
Taiga Shield
Atlantic Maritime
Boreal Shield
N
W
E
S
1980 - 2007 Large Wildfire
0
325
650
1,300 Kilometers

Fig. 1.

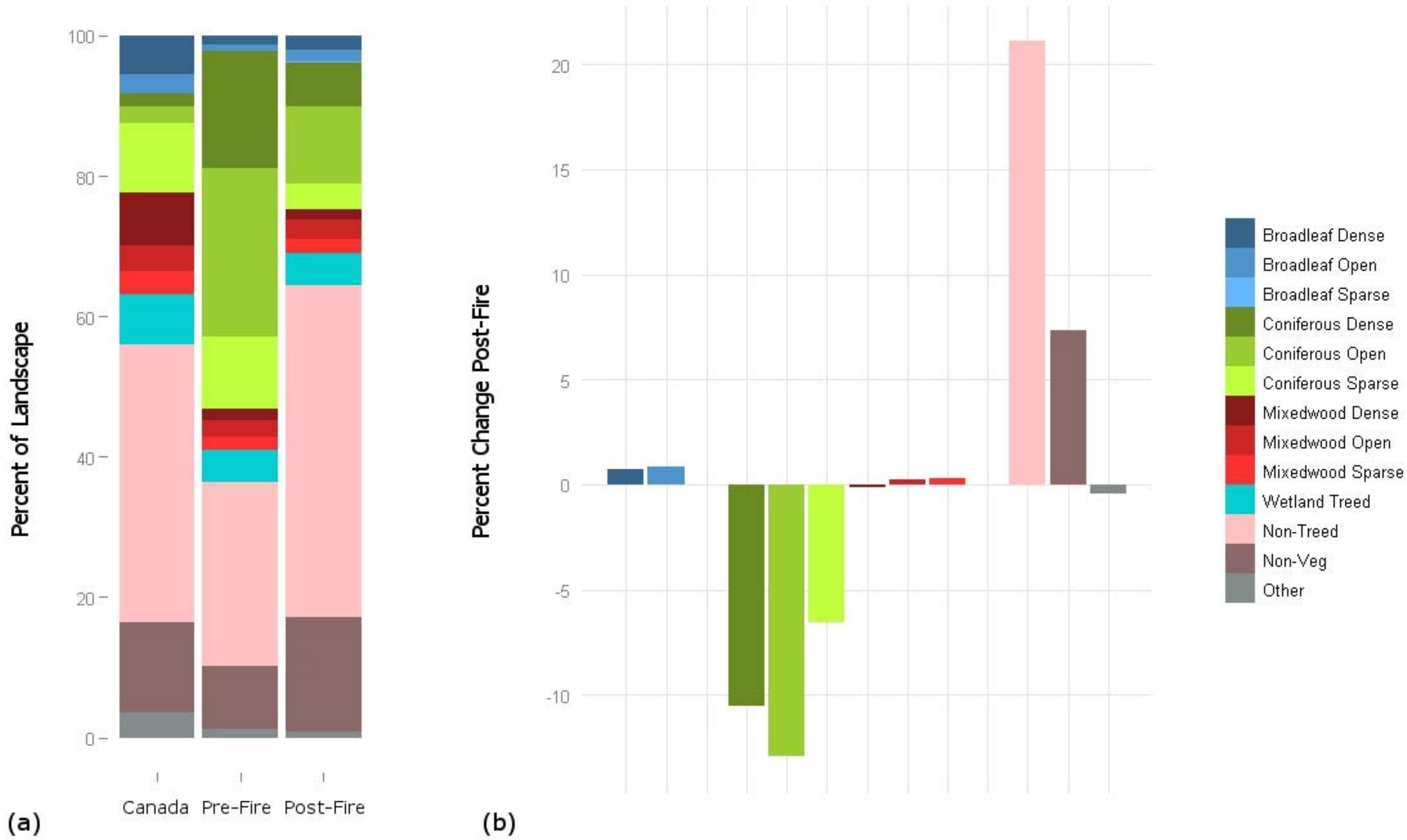

Percent of Landscape
100
80
60
40
20
0
Canada
Pre-Fire
Post-Fire
(a)
Percent Change Post-Fire
20
15
10
5
0
-5
-10
(b)
Broadleaf Dense
Broadleaf Open
Broadleaf Sparse
Coniferous Dense
Coniferous Open
Coniferous Sparse
Mixedwood Dense
Mixedwood Open
Mixedwood Sparse
Wetland Treed
Non-Treed
Non-Veg
Other

Fig. 2.

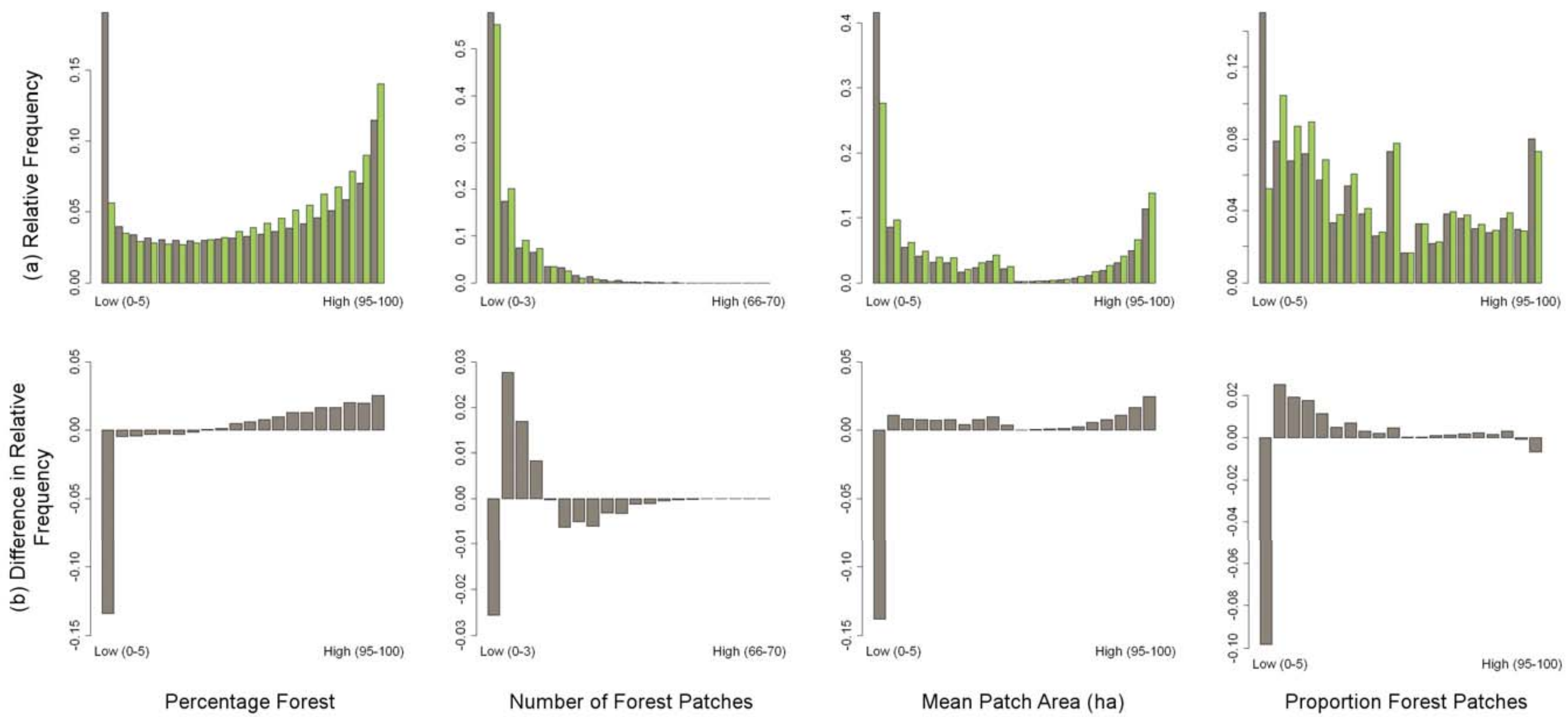

(a) Relative Frequency
(b) Difference in Relative Frequency
Low (0-5)
High (95-100)
Low (0-3)
High (66-70)
Percentage Forest
Number of Forest Patches
Mean Patch Area (ha)
Proportion Forest Patches

Fig. 3.

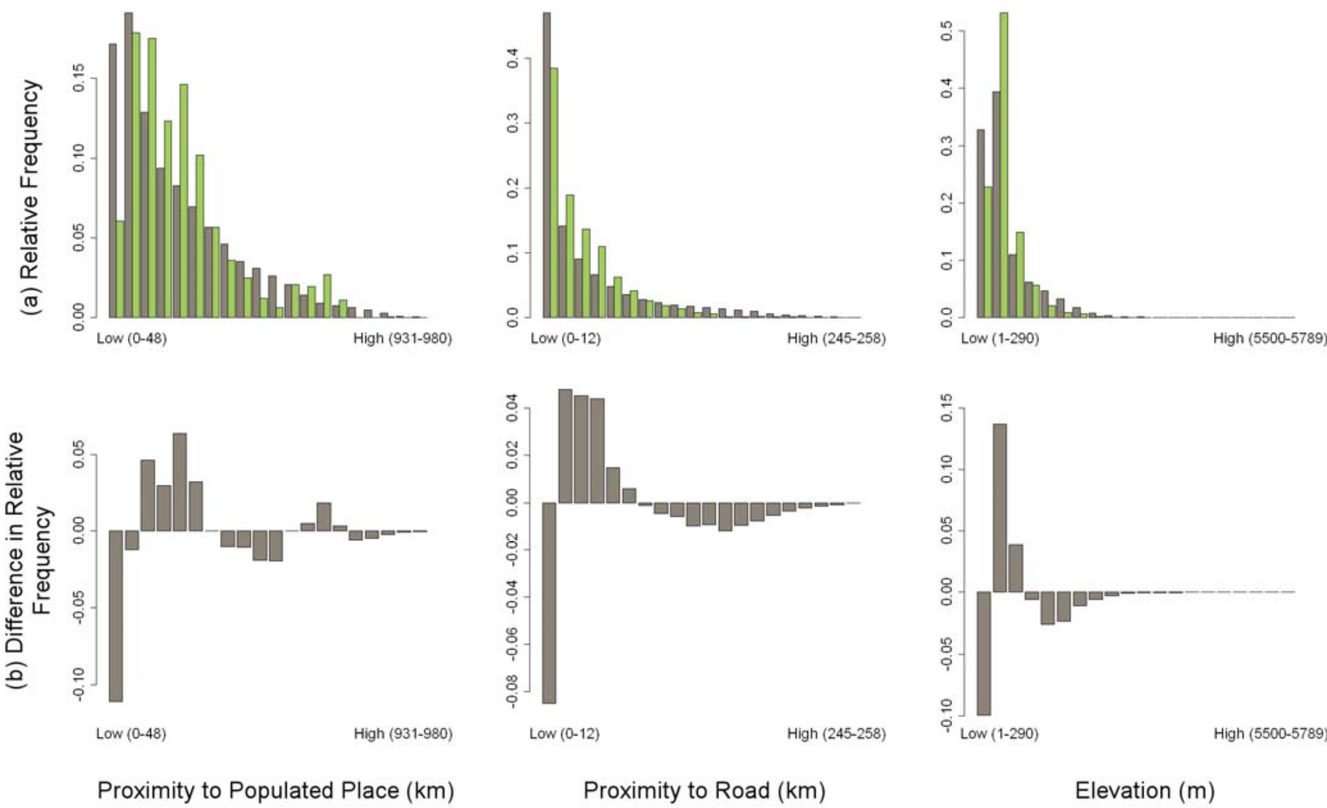

(a) Relative Frequency
(b) Difference in Relative Frequency
Low (0-48)
High (931-980)
Low (0-12)
High (245-258)
Low (1-290)
High (5500-5789)
Proximity to Populated Place (km)
Proximity to Road (km)
Elevation (m)

Fig. 4.

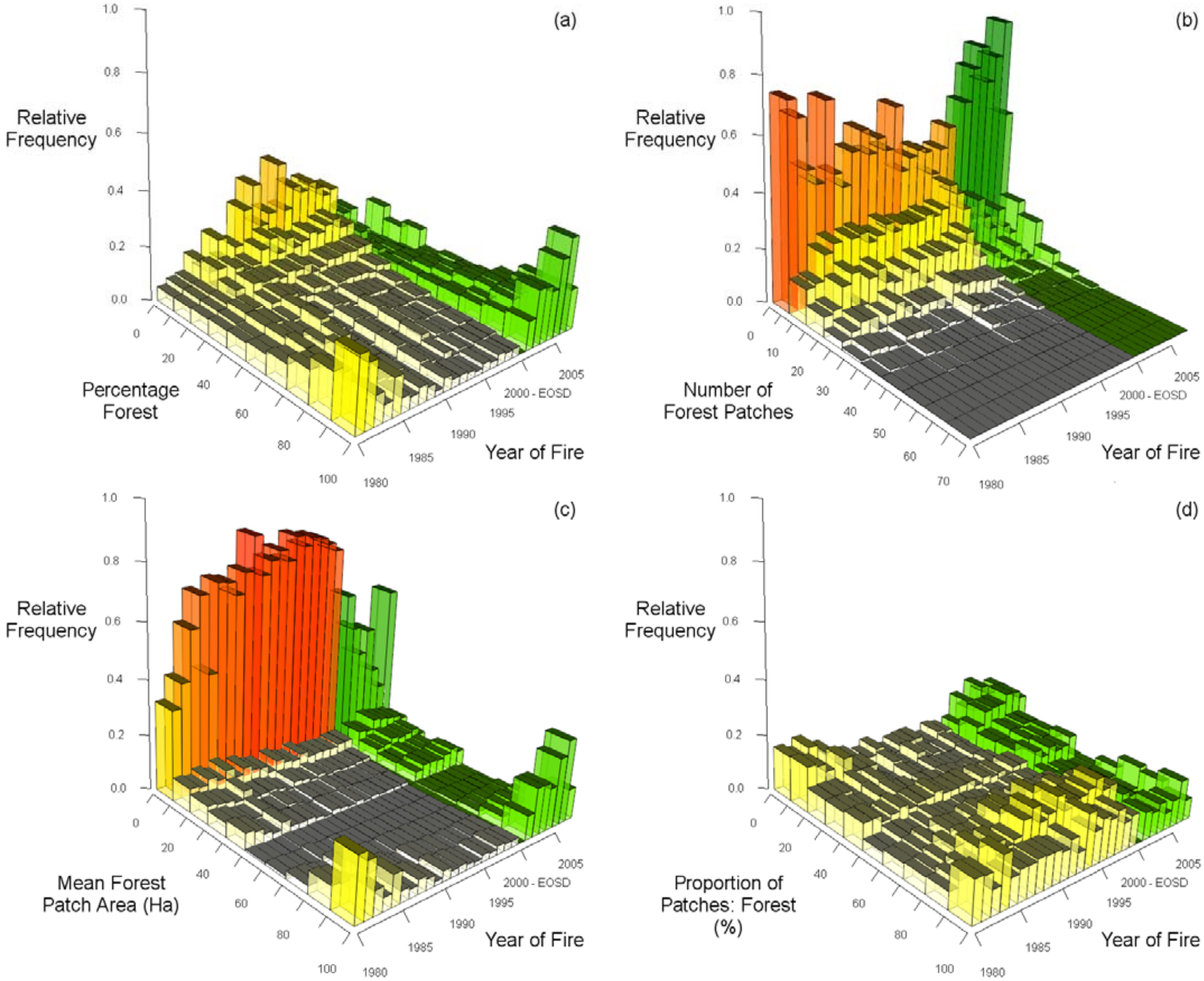

(a)
Relative Frequency
Percentage Forest
Year of Fire
2000 - EOSD
(b)
Number of Forest Patches
(c)
Mean Forest Patch Area (Ha)
(d)
Proportion of Patches: Forest (%)

Fig. 5.

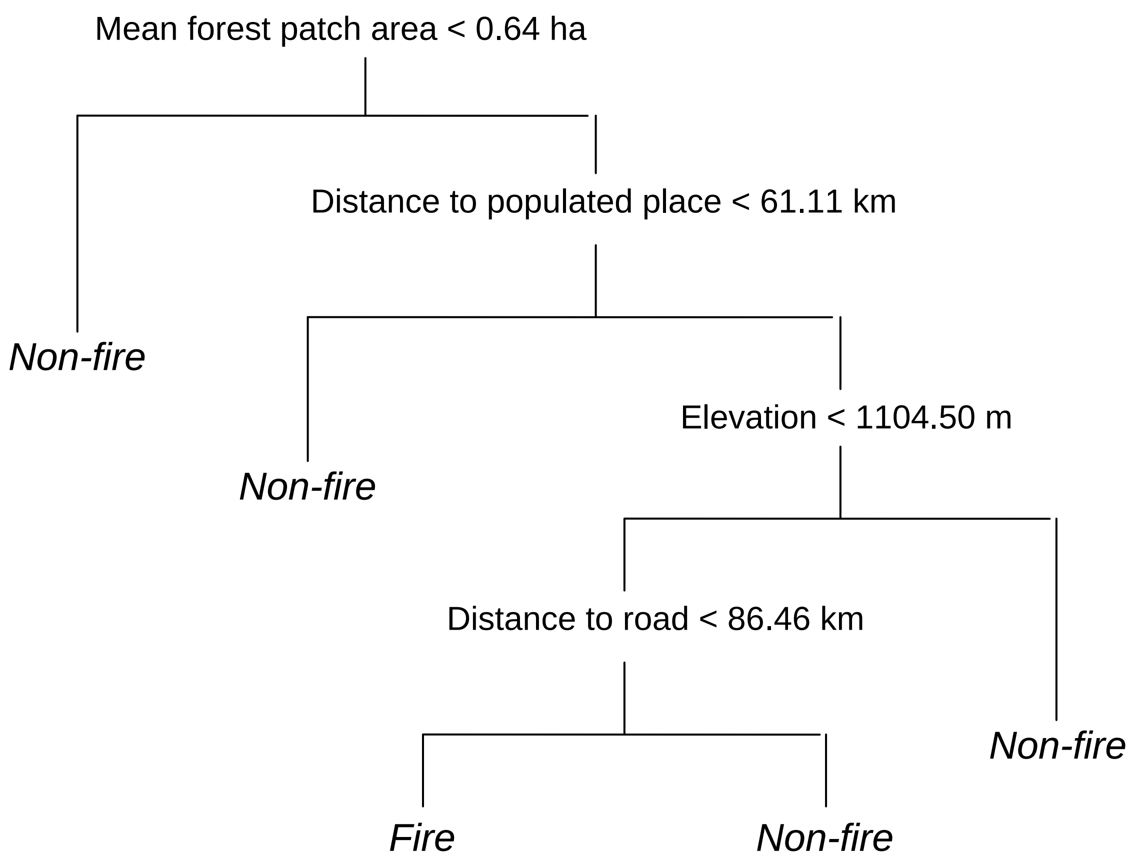
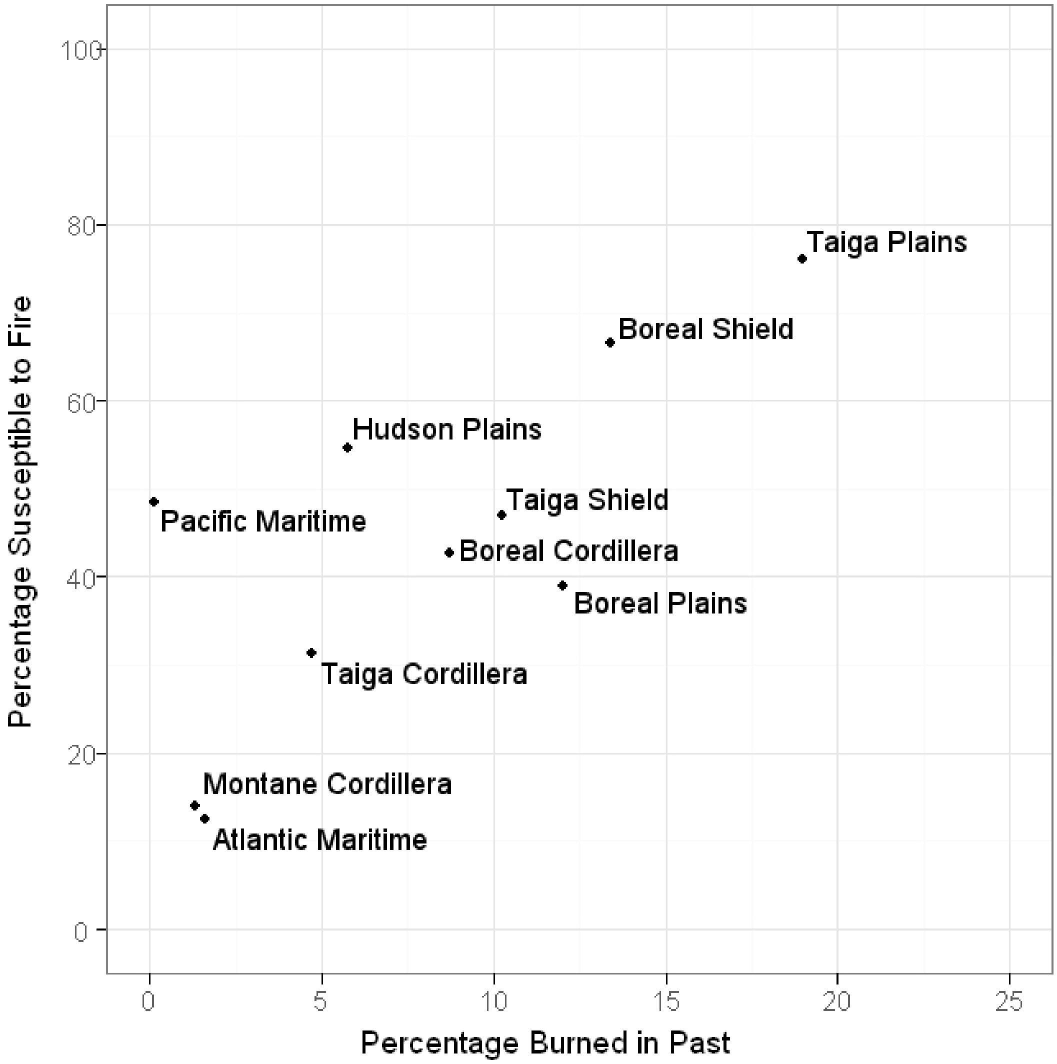

Mean forest patch area < 0.64 ha
Distance to populated place < 61.11 km
Non-fire
Non-fire
Elevation < 1104.50 m
Distance to road < 86.46 km
Non-fire
Fire
Non-fire

Fig. 6


Percentage Susceptible to Fire
Percentage Burned in Past
0
20
40
60
80
100
0
5
10
15
20
25
Taiga Plains
Boreal Shield
Hudson Plains
Taiga Shield
Pacific Maritime
Boreal Cordillera
Boreal Plains
Taiga Cordillera
Montane Cordillera
Atlantic Maritime
Fig. 7.