University of Alberta

The Effect of Anthropogenic Linear Features on Forest Fire Ignition and Initial Suppression in Northeastern Alberta

by

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ABSTRACT

Anthropogenic linear disturbances have been hypothesized to exert positive and negative effects on fire behaviour. My thesis focuses on quantifying the influence of roads, pipelines and seismic lines on fire ignition frequency and escape from initial suppression. I considered fires that ignited between 1995 and 2002 within a ≈67 000 km^2 region of boreal mixedwood forest in northeastern Alberta where linear features are highly abundant. I found a positive association between the number of lightning-caused fires igniting per unit area and road density, with this effect confined to summer and possibly spring. I found no evidence for a pipeline or seismic line effect, even after accounting for seasonality and scale dependency. Linear features did not influence fire escape probability; rather, this process was strongly affected by forest composition, fire weather and management decisions. My results suggest that, in the face of projected road developments in the region, the potential exists for important changes to the fire regime.

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Chapter One

General Introduction

This thesis explores the influence of anthropogenic linear features on two aspects of fire behaviour, i.e. ignition frequency and escape from initial attack (i.e. the first suppression actions taken to halt fire spread), within a ≈67 000 km^2 region of boreal mixedwood forest in northeastern Alberta, Canada. This region is subjected to a combined forestry and energy extraction regime, whose activities produce an extensive footprint on the landscape consisting of roads, pipelines and seismic lines, among other structures. Understanding the influence of these increasingly widespread anthropogenic linear elements on the natural fire regime is required in order to achieve the sustainable management of these forests.

1.1 Fire in the boreal mixedwood forest

The mixedwood section of the boreal forest extends from south western Manitoba to British Columbia (Rowe 1972). Forest associations within this eco-region are typically dominated by aspen or aspen-spruce mixes on upland mesic sites, while jack pine dominates in xeric sites and black spruce dominates in lowlands. There are also extensive wetlands including bogs, fens and marshes. Succession in upland forests typically start as aspen stands, going through a mixedwood phase, and ends up in stands where white spruce dominates; though the precise pathway followed depends on conifer seed availability, distance from sexually-mature spruce stands, and the suitability of the seedbed (Frelich and Reich 1995, Greene *et al.* 1999, Albani *et al.* 2005, Peters *et al.* 2005). However, the boreal forest is considered a disturbance forest, where frequent stand-replacing disturbances such as fires, insect outbreaks and wind throw, prevent long term conifer dominance in uplands and maintain a heterogeneous forest (Heinselman 1981, Payette 1992, Engelmark *et al.* 1993).

Lightning-caused fires are the predominant natural disturbance in the western boreal forest and, as such, they are a critical component of forest dynamics across a wide range of spatial and temporal scales (Engelmark *et al.* 1993, Armstrong 1999). Fires influence forest stand composition (Bergeron *et al.* 1998) and determine age class structure and patch size (Van Wagner 1978). The boreal forest depends on periodic fires for its renewal: areas subjected to a high fire frequency have an overall younger forest, while areas with longer intervals between fires typically show a mosaic of older and smaller forest patches (Van Wagner 1978, Johnson and Van Wagner 1984, Weir *et al.* 2000). In western Canada, the fire cycle, or 'the expected time required to burn an area equal in size to an area of interest' (Reed 2006), has been estimated to currently range between 155 and 5 155 years, depending on the specific region where it is calculated (Bergeron *et al.* 2004).

In northeastern Alberta, lightning-caused fire represents 53% of all fires recorded in digital provincial archives (ASRD 2004) and accounts for 77% of the total area burned. Human-caused fires also represent a significant and potentially growing source of disturbance, but they differ from lightning-caused fires in phenology and spatial distribution (Stocks *et al.* 2002).

Several studies have demonstrated significant variation in lightning-caused fire ignition frequency between different forest stand types, weather conditions, and topography (Renkin and Despain 1992, Agee 1997, Diaz-Avalos *et al.* 2001, Schoennagel *et al.* 2004, Krawchuk *et al.* 2006). In the forests of northeastern Alberta, lightning fire occurrence and burn rates have been positively related to the abundance of white and black spruce stands, and negatively associated with deciduous forest and recently burned areas (Cumming 2001a, Krawchuk *et al.* 2006). Fire intensity and rate of spread are mostly affected by the conifer composition of the burning stand as well as by extreme weather conditions (Hély *et al.* 2001). Finally, final fire size has been correlated with forest composition in the vicinity of the fire ignition point, increasing with the abundance of pine stands and decreasing with the abundances of recently logged or burned areas (Cumming 2001b).

From a simulation modelling perspective, wildfire has been characterized as a three stage stochastic process: arrival, escape and growth (Cumming 2000, Armstrong and Cumming 2003). An arrival is defined as a detected and reported fire (Cunningham and Martell 1976). Cumming (2000) characterized the spatial pattern of lightning-caused fire arrivals in northern Alberta as a negative binomial process. The escape process represents the effect of initial suppression efforts on fire spread; fires not extinguished within a small target size (typically 2-4 ha) are said to have escaped. The observed number of escapes can be modelled as a binomial random variable determined by the total number of arrivals and the *escape probability p* (Cumming 2005). Finally, the growth process determines the final size, shape and forest composition of escaped fires (Cumming 2001b).

1.2 Industrial development in northeastern Alberta

Alberta's boreal forest is currently subjected to multiple overlapping land uses, including timber harvesting and oil and gas exploration and extraction. Roads, cutblocks, well sites, and other infrastructure that accompany these activities have become prominent features in this part of the world; and have the potential to compromise various ecological functions and the integrity of the forest as a whole.

Industrial activities have already placed a significant footprint on Alberta's boreal landscape, with the creation of an extensive, and in some areas very intensive network of linear features, primarily roads, pipelines and seismic lines (Cumming and Cartledge 2004). These features cause a small direct reduction in the amount of habitat; but can potentially generate functional habitat loss by increasing forest fragmentation and the area influenced by edge effects (Forman and Godron 1986, Bayne *et al.* 2005), which can translate into negative impacts on forest biota (Dyer *et al.* 2001, MacFarlane 2003, Bayne *et al.* 2005). The expansion of this type of footprint may also influence subsequent timber harvesting and other industrial activities as well as human access for hunting and recreation (Schneider 2002, Cumming and Cartledge 2004). Increased human access has also been shown to alter fire regimes by associated increases in the number of human-caused fires (Vega Garcia *et al.* 1995).

In conventional energy sector operations, the majority of the disturbance, as measured by the total length of forest cleared, occurs during the exploration phase, when linear corridors called seismic lines are created for delineating subsurface areas with fossil fuel reserves. As of 2004, there were a total of ≈110 000 km of seismic lines within my study region, with a mean density of 1.70 km/km^2. Conventional approaches to seismic exploration involved construction of 6 to 8 m wide lines; whereas low impact seismic lines cut from 1995-on are 5 m wide on average (Schneider 2002). A conventional seismic line program may have parallel or gridded lines at a spacing of 400 m or more; in a heavy oil extraction area, on the other hand, lines are in a grid pattern often spaced only 100 m or less apart. There are no specific requirements for reclamation of seismic lines, but in general abandoned seismic lines are revegetated using general agronomic seed mixes of grasses and forbs or allowed to revegetate naturally (Revel *et al.* 1984). Approximately 20% of seismic lines are eventually re-cleared for subsequent exploration or tracked access, while 3% are upgraded to roads and 65% completely abandoned (Lee and Boutin 2006).

Pipelines and roads are a far less widespread element in the area, though they tend to be active for longer periods of time and used more intensively than seismic lines. The total length of pipelines and roads in my study region (as of

2004) was ≈18 000 and ≈22 000 km, respectively; while their mean densities were 0.27 and 0.33 km/km^2, respectively. Pipelines act as connectors between wells and intermediate processing facilities, and ultimately export the resources to the markets. Most pipelines are buried underground, but their surface is actively prevented from regenerating to forest in order to facilitate access for maintenance or emergencies. Pipelines are significantly wider than seismic lines, typically measuring 25 m in width (Cumming and Cartledge 2004).

There are two broad categories of roads, industrial and provincial. The width, quality and permanence of industrial roads are highly variable. Provincial roads are permanent all-weather public highways which are present at a low density in the region, totalling ≈1 100 km in length as of 2004. The paved surface of roads ranges from 5 to 10 m; in addition, both types of corridors have actively maintained clearings on each side, ranging from 8 to 20 m for industrial roads and 40 m for provincial highways (ASRD 2006). Roadside clearings have conventionally been revegetated with general agronomic seed mixes of grasses and forbs, although there is a recent trend towards the use of native species (AMEC 2001).

1.3 Anthropogenic influences on fire dynamics

Whereas fire behavior may once have been determined entirely by environmental factors, there is ample evidence that fire occurrence is now strongly influenced by patterns of human activity (Frissell 1973, Haines *et al.* 1983, Martell *et al.* 1987, 1989, Johnson *et al.* 1990, Cardille *et al.* 2001, Donnegan *et al.* 2001, Lefort *et al.* 2003). DellaSala *et al.* (2004) report several ways in which anthropogenic land uses influence fire including (1) increases in human-related ignitions; (2) changes in the biomass of fine fuels by alteration of the vegetative cover; (3) modifications in the rate of fire spread related to habitat fragmentation; and (4) active fire suppression.

Several previous studies have developed statistical models quantifying the relationship between the occurrence of human-caused fires and daily weather conditions at various scales (Martell *et al.* 1987, 1989, Todd and Kourtz 1991, Wotton *et al.* 2003). Vega Garcia *et al.* (1995) modeled the probability of daily human-caused fire occurrence as a function of both weather and geographical factors, including measurements of human land use. Lefort *et al.* (2004) found that the area burned by and the occurrence of human-caused fires were positively correlated with the density of the road network and sand deposits. Chou *et al.* (1993) related wildland fire (lightning and human-caused fires combined) occurrence with monthly and annual weather conditions, as well as with anthropogenic factors quantified as distances to nearest campground, road or human infrastructure. Cardille *et al.* (2001) detected a positive relationship between road density and wildland fire occurrence; though they also emphasized the importance of other biotic (e.g. forest composition), abiotic (e.g. weather and lake density), and human (e.g. population density) factors as significant influences on wildland fire patterns.

Diverse human land uses alter the natural relative frequency of different vegetation types within a region, as a result of voluntarily planted species or accidentally introduced exotics. Of particular interest is the invasion of native and non-native grass species into areas disturbed by human activities, such as road (Sumners 2005), seismic line (MacFarlane 2003), and powerline right-of-ways (Cameron *et al.* 1997). It is important to note that in my study region, such introduction of exotics is deliberate, as it results from the use of general agronomic mixes for line revegetation. Several authors have described dramatic effects of this invasion on the regenerating natural plant community as well as on the disturbance regime (D'Antonio and Vitousek 1992, Brooks *et al.* 2004). As noted by D'Antonio and Vitousek (1992), native and exotic grasses increase fuel-bed flammability by changing fuel properties, which can potentially affect the frequency, size, spatial pattern, and, in some cases, intensity of fires. Furthermore, areas that are subjected to consistent disturbance, such as for example road maintenance, favor the establishment and survival of grasses (Cameron *et al.* 1997), creating an environment that sustains a consistent and long-term 'flammable' community. Changes in the fire regime promoted by such shifts in community composition could translate into major ecosystem-level transformations (Mack and D'Antonio 1998).

Landscape connectivity, defined as the degree to which a landscape facilitates or impedes movement of organisms (fire in this case) among patches (Taylor *et al.* 1993), is important in controlling disturbance dynamics (Turner *et al.* 1989). Computer simulation studies have demonstrated that, for very small changes in connectivity, there can be large, sudden changes in the spread of disturbances (Turner *et al.* 1989). Miller and Urban (2000) found that natural connectivity among forest fuels change with elevation, affecting how readily surface fires can spread among low and high elevation sites. Fire spread is very sensitive to the spatial arrangement of fuels under moderate weather conditions; however, under extreme winds and low moisture, the importance of this effect is diminished (Turner and Romme 1994). Connectivity of fuels can also change as a result of human land use patterns. Guyette *et al.* (2002) show that artificial fuel breaks caused by livestock grazing and road building in the Missouri Ozarks can inhibit propagation of low intensity surface fires. Additionally, Duncan and Schmalzer (2004) found that agricultural areas and roads disrupt spatial fire behaviour disproportionately to their actual abundance, with as little as 10% anthropogenic landcover causing a 50% decline in fire extent.

Fire suppression is widely practiced in those countries where wildfires are naturally occurring events. While there is evidence that fire suppression has reduced the number and final size of fires (Ward and Tithecott 1993, Martell 1994, Ward *et al.* 2001, Cumming 2005), some authors claim that suppression has not been effective in stopping extreme fire events (Larsen 1997, Johnson *et al.* 1998, Johnson *et al.* 2001, Miyanishi and Johnson 2001). Regardless of the validity of this last statement, fire suppression influences current fire patterns to some degree, and thus it also affects the ecosystems where fires are part of the natural disturbance regime (Veblen *et al.* 2000, Backer *et al.* 2004).

1.4 Research needs

In the boreal mixedwood forest of northeastern Alberta, anthropogenic linear features produce changes in landscape structure by increasing forest fragmentation (Forman and Godron 1986), changing vegetation cover from forested to non-forested (Revel *et al.* 1984), and facilitating human access to formerly remote areas (Schneider 2002). Thus, the presence and prevalence of roads, pipelines and seismic lines in these forests may alter fire behaviour and regime in many different ways. Fuel fragmentation can impede fire spread and reduce potential final fire size (Duncan and Schmalzer 2004). As noted by Kiil (1970), the herbaceous vegetation on some of these lines may increase fire ignition probabilities and spread rates, especially during early spring and summer when some of these grass species experience a marked decrease in foliar moisture (Van Wagner 1983). This process can be particularly exacerbated in open areas with no forest overstory to reduce direct sunlight and water evaporation (Carlson and Groot 1997). Furthermore, some exotic grass species thriving on these high-light/low-moisture open habitats represent a new source of combustible fuel, which may further facilitate ignition (Stoner *et al.* 2004). As industrial or recreational use follows the developing road and access network, the frequency of human-caused fires is expected to increase. Conversely, the access network may facilitate fire fighting activities, with seismic lines providing a "near-ideal base for fire suppression operations" (Kiil 1970), resulting in the net effect of hindering fire spread and decreasing potential final size. To date, none of these effects have been substantiated for the western boreal forest.

Research on the ecological impacts of linear features is in its infancy. Furthermore, their present density is projected to increase rapidly throughout the boreal forest; thus, any effects that linear features currently have can only be expected to intensify in the future (Schneider *et al.* 2003). Within this context, evaluating and quantifying the effects of linear features on two aspects of fire behaviour, i.e. ignition frequency and escape from initial suppression, are the main focuses of this thesis.

1.5 Thesis overview

This thesis is organized into four chapters. In this first Chapter, I have introduced the background of fire ecology in the boreal mixedwood forest of western Canada, and described potential influences of anthropogenic linear disturbances on fire behaviour. Data chapters (Two and Three) are written in manuscript style; consequently, there is some redundancy in describing the study area and some of the data sources. A version of Chapter Three is currently in press in the *Canadian Journal of Forest Research*: 'Arienti, M.C., Cumming, S.G., and Boutin, S. Empirical models of forest fire initial attack success probabilities: the effects of fuels, anthropogenic linear features, fire weather and management'.

In Chapter Two, I examine whether linear features in the boreal forest affect lightning-caused fire ignition frequency. Linear features have been hypothesized to increase the abundance of fine fuels, i.e. grasses, and thus augment fire ignition potential. To test this, I addressed the following questions:

1- Do roads, pipelines or seismic lines increase lightning-caused fire ignition frequency?

2- Is this effect consistent when analyzed at a coarse (≈10 000 ha landscapes) and a fine (≈2 400 ha landscapes) spatial resolution?

3- Does this effect vary seasonally, as a function of changes in fuel moisture and temperature conditions?

In Chapter Three, I examine the effect of linear features on the effectiveness of initial fire suppression, as measured by the proportion of fires larger than 3 ha (escapes, *sensu* Cumming 2005). Linear features have been hypothesized to act both as fire breaks and as fire spread corridors, which would translate into positive or negative effects on fire suppression effectiveness. They also improve access, which might increase the efficiency of suppression actions. To test these hypotheses, I addressed the following questions:

1- Is the probability of fire escape before any suppression actions are attempted affected by the density of linear features within the fire vicinity?

2- Is the probability of fire escape while it is being suppressed influenced by the density of linear features within the fire vicinity?

Finally, in Chapter Four I present some general conclusions and discuss both the ecological and economical impacts of potential changes in fire dynamics resulting from anthropogenic linear disturbances. I also offer some recommendations for future research and management of fire hazard on linear features in the boreal forest of northeastern Alberta.

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Chapter Two

Lightning-caused Fire Ignition Frequency and Linear Features

2.1 Introduction

Human activities in natural forested ecosystems have the potential to modify not only landscape structure but also alter the disturbance regime intrinsic to such ecosystems (Veblen *et al.* 2000, Lefort *et al.* 2003). In the Canadian boreal mixedwood forest, lightning-caused fires are the dominant natural disturbance, accounting for 85% of the area burned (Weber and Stocks 1998). Lightning fire behaviour is traditionally explained in terms of the interacting factors of weather, fuels, and topography (Agee 1997). Short term weather conditions (wind, temperature, precipitation) determine the moisture properties of forest fuels; the number of lightning strikes increases with elevation; and forest stands are inherently variable in the amount and condition of fuels available for combustion.

Whereas lightning fire ignition locations may once have been determined entirely by these three environmental factors, there is now ample evidence that fire occurrence is increasingly influenced by patterns of human activity (Chou *et al.* 1993; Cardille *et al.* 2001; Donnegan *et al.* 2001; Lefort *et al.* 2003). In my study region, the boreal mixedwood forest of northeastern Alberta, the majority of human disturbances are conducted by the forestry and energy sectors (Schneider 2002). The direct legacy of these activities is an extensive industrial footprint in the landscape in the form of a growing network of linear features, e.g. roads, pipelines and seismic lines (Schneider *et al.* 2003).

Linear features alter landscape structure by increasing forest fragmentation (Forman and Alexander 1998), changing vegetation cover from forested to non-forested (Revel *et al.* 1984), and facilitating human access to formerly remote areas (Schneider 2002). Subsequently, these landscape changes can affect several aspects of fire behaviour, such as fire spread and final size (Duncan and Schmalzer 2004). This chapter is concerned with ignition. The processes by which a lightning strike may result in a detected and reported fire (a fire arrival, *sensu* Cunningham and Martell 1976) are described by Kourtz and Todd (1991). Linear features in the boreal forest are expected to influence these processes by altering the abundance, flammability and spatial distribution of fine fuels, particularly grasses, which experience a marked decrease in foliar moisture content and thus are highly combustible during early spring and summer (Van Wagner 1983). Thus, any effect linear features may have on fire ignition could be expected to vary seasonally, associated with phenology and changes in the flammability of natural and introduced graminoid fuels.

Scale has long been recognized as important in ecology, since the scale at which dependent and independent variables are collected can influence the strength and nature of the relationships that are discovered (Levin 1992). At a landscape level, spatial pattern and heterogeneity depend upon the scale at which those measurements are taken; as a general rule, higher spatial complexity is captured with increasing resolutions (Turner 1989). The frequency and spread of disturbances such as fires is controlled by this spatial heterogeneity in forest fuels and other physical features; thus, our ability to properly describe these processes will ultimately depend on the scale at which we chose to study them.

In this chapter, I developed statistical models to describe spatial variation in the frequency of lightning-caused fires igniting over the period 1995-2002 within a ≈67 000 km^2 region of boreal mixedwood forest in northeastern Alberta.

As described before, this region contains a large and increasing abundance of anthropogenic linear disturbances, thus particular interest was placed on unraveling their current influence on fire ignition patterns. My specific objectives were to: 1) quantify the effect of linear features on the frequency of lightning-caused fires; 2) assess the sensitivity of detected relationships to the spatial resolution of analysis; and 3) test for the expected seasonal variation in the effect of linear features.

2.2 Data and methods

2.2.1 Study region

My study region was a large (≈67 000 km^2) forest estate in the boreal mixedwood forest (Rowe 1972) of northeastern Alberta, Canada (Fig. 2.1a). The area is characterized by a continental climate (Strong and Leggat 1992) and moderate topographic relief, with elevations ranging from 200 to 850 m. Mature mixed stands containing either trembling aspen (*Populus tremuloides*) or balsam poplar (*Populus balsamifera*) and white spruce (*Picea glauca*) are characteristic of well-drained sites (Strong and Leggat 1992) and cover approximately 25% of the area. Rapidly draining sites are vegetated principally by jack pine (*Pinus banksiana*), and some white spruce and aspen. Poorly drained areas are dominated by black spruce (*Picea mariana*) and tamarack (*Larix laricina*). Various wetland types, e.g. sparsely treed or open bogs and fens, cover about 50% of the study region.

The region was relatively undisturbed by human activity until the 1950s; however, most of the study region is now allocated to industrial development by the forestry and energy sectors. These activities have altered spatial distribution and age structure of vegetation communities, and have markedly increased the density of linear features (Schneider 2002). About 3% of the study area has been disturbed by surface developments such as roads, pipelines, seismic lines and permanent clearings associated with the energy sector (Fig. 2.1b), while 2.6% has been directly harvested by forestry companies.

Seismic lines are narrow corridors (≤ 8 m) constructed for the detection of oil and gas reserves and are revegetated using mixes of general agronomic species or allowed to revegetate naturally (Revel *et al.* 1984). Until 1995, most seismic lines were 8 m wide, but their widths now vary from 1-7 m with a mean of 5 m (Schneider 2002). Buried pipelines connect wells with intermediate processing facilities in the area, they are typically 25 m in width and their surface is prevented from regenerating to forest in order to facilitate maintenance. Roads include winter roads, in-block haul roads, permanent industrial roads and all-weather provincial highways. Most of these roads have clearings on each side ranging from 8 to 20 m wide, while provincial highways have 40 m clearings. Roadside clearings are generally revegetated with general agronomic seed mixes of grasses and forbs.

2.2.2 Data

2.2.2.1 Sampling units

I modeled the relationship between the number of lightning-caused fires per unit area and various biotic and abiotic factors at two spatial resolutions. For the coarse-resolution analysis, I used the Alberta township grid. Townships are land survey units with a mean area of 9 230 ha, varying slightly in size due to geographical corrections. This resolution has previously proved adequate to explore several issues in fire ecology (Cardille *et al.* 2001, Cumming 2001, Krawchuk *et al.* 2006). Of 806 townships within the study region, I selected 601 for analysis with a minimum size of 7 000 ha and classifiable terrestrial surface area greater than 70% (Fig. 2.1b). These criteria excluded small townships, those having large water bodies or large areas of unclassified vegetation. For the fine-resolution analysis, I used quarter-townships: each township was divided into four equal sized quarters of approximately 2 370 ha each (Fig 2.2). This spatial resolution was equivalent to the fine-scale grid used by Cardille *et al.* (2001). Using the 70% classifiable surface area criteria, I selected 2 370 quarter-townships for analysis.

2.2.2.2 Fire-day counts

The Government of Alberta's Forest Protection Division has on-line provincial-wide databases of fire records from 1961-2003 (ASRD 2004). I used a consistent version of these data assembled by S. Cumming (Boreal Ecosystems Research Ltd.), containing information on fire cause, date of occurrence, and geographical location for the fire ignition point (Arienti *et al.*, *in press*). From this, I created a count response variable for the total number of lightning-caused fires per sampling unit over the interval 1995-2002 (Fig 2.1b). This sampling interval was chosen as a compromise between minimizing temporal variation in landscape characteristics resulting from increasing industrial activities in the area, and maximizing the time window so as to reduce the effects of inter-annual variation in fire weather conditions. I considered only those fires that occurred during the regional fire season (30 April to 29 September; C. Tymstra, personal communication, 2005). Multiple fires igniting on the same day within the same sample unit were infrequently detected; nevertheless, these were omitted in order to avoid the statistical dependence of events resulting from the same weather/forest conditions (Krawchuk *et al.* 2006). Thus, the response variables are actually fire-day counts per sampling unit.

2.2.2.3 Model covariates

I modeled spatial variation in the response variables as functions of four classes of covariates: linear feature densities, land cover class, fire weather and lightning, and geography. Table 2.1 describes each variable, while graphical summaries at both resolutions can be found in Figures 2.3 and 2.4. I measured the explanatory variables from available spatial data layers using Arc Map 9 (ESRI 2004).

I considered three classes of linear features: roads, pipelines and seismic lines. I used linear feature layers from the study area considered valid and complete up to the year 2003. Time-series spatial linear feature data for the area do not exist, and the construction dates of individual features cannot be determined in general; thus, it was necessary to assume that densities had remained constant over the eight-year sampling interval. The road layer did not include detailed attributes such a width or treatment, so it was necessary to consider that all delineated roads were equivalent. The available layers were intersected with the township and quarter-township grids and densities of roads (RO), pipelines (PL), and seismic lines (SL), in units of km/km^2 of terrestrial surface area (hereafter referred to as 'dry land'), were calculated for each sampling unit (Fig. 2.2). For the rest of this chapter, variable names followed by subscript t indicate variables collected at the township scale while variable names followed by subscript q indicate those collected at the quarter-township scale.

I described the forest and vegetation composition of each sampling unit using digital forest inventory data (Alberta Vegetation Inventory (AVI); Nesby 1997). Mapped forested and other vegetated polygons, or stands, were classified mainly by canopy species composition. The proportional areas of six forested classes were calculated for each sampling unit: deciduous (DE), white spruce (WS), black spruce (BS), jack pine (PI), recently ($\leq$ 30 yr) burned (BU) and harvested areas (cutblocks; CU) (Fig. 2.2). Non-forested landscape elements, including water bodies, agricultural land and permanent human infrastructure, were omitted from analysis. I assumed that there were minimal cover type changes due to natural succession or forest harvesting over the eight-year time interval. However, forest composition varied markedly across the region, providing the spatial contrast required for analysis (Figs. 2.3 and 2.4).

The ignition of a lightning-caused fire requires the co-occurrence of a lightning strike under favorable fire weather conditions (i.e. low moisture) (Nash and Johnson 1996). To quantify this, I derived two joint fire weather and lightning (JWL) indices (Krawchuk *et al.* 2006) from spatially registered lightning flashes and two spatially interpolated fire weather indices, the Duff Moisture Code (DMC) and the Fine Fuel Moisture Code (FFMC). These two components of the Canadian Forest Fire Weather Index System (FWI; Van Wagner 1987) measure moisture content of litter or fast-drying fine fuels and loosely compacted organic layers of moderate depth, respectively. Following Krawchuk *et al.* (2006), fuel moisture conditions were considered 'favorable' or 'receptive' to fire ignition if DMC > 34 or FFMC > 87. For each cell, I identified days with DMC or FFMC above these threshold values and at least one lightning strike. These days were summed over the eight-year period to compute D-L and F-L (Table 2.1), which represent the number of days over the study interval where fuel moisture conditions were low (DMC > 34 or FFMC > 87) and there was at least one lightning strike detected within a cell. Recorded lightning flashes were referenced to the coarse and fine-scale grids; however, fire weather indices were interpolated only to the coarse grid since I assumed there would be small variation in these indices within townships. For details on underlying data sources, interpolation procedures and consideration of spatial accuracy, see Krawchuk *et al.* (2006).

Geographical location was included as a proxy for unmeasured spatial gradients. The NAD 27 UTM measure of northing (N) and easting (E), in m, were calculated at each sample unit's centroid. E and N were incorporated as covariates after scaling them down by a factor of 10^5 and 10^6, respectively, in order to improve convergence of certain numerical algorithms used for statistical analysis (Kéry *et al.* 2005). Total township and quarter-township area (TA), in ha, were also calculated. These were incorporated into analysis after being scaled down by a factor of 10^3.

2.2.2.4 Statistical analysis

I modeled the response variables, lightning-caused fire-day counts per cell, as a function of the covariates within a general count regression framework. The R environment for statistical computing (R Development Core Team 2005), version 2.2.0 for Windows, was used for statistical analyses and to prepare graphics.

The simplest regression model for count data is the Poisson distribution (hereafter P; Cameron and Trivedi 1998), where the responses are assumed to be samples from a Poisson process with conditional mean given by some function of the covariates. I used a Generalized Linear Modeling (GLM) framework (McCullagh and Nelder 1989), where the log-conditional mean is a linear function of the covariates (the linear predictor). P data have equal mean and variance; however, real-life data such as mine frequently display overdispersion (variance greater than mean). This may result from neglected or unobserved heterogeneity that is inadequately captured by the covariates in the mean function. A standard approach in such cases is to use a Negative Binomial (hereafter NB) distribution, where the variance is assumed to be a function of the square of the mean ($\sigma^2 = \mu + \mu^2 * \theta^{-1}$) (Cameron and Trivedi 1998). NB data would be expected to have an excess of both low and high counts relative to the mean. I implemented NB regression within a Generalized Linear Modeling (GLM) framework. Alternatively, the data may display overdispersion if the incidence of zero counts is greater than that predicted by P, which in my case might be expected because appropriate fire weather conditions and lightning are both pre-requisites for a lightning-caused fire to occur. Such data can alternatively be modeled using Zero-inflated Poisson (hereafter ZIP) regression (Lambert 1992).

ZIP models assume that a two-stage process generated the observations: the *transition stage* occurs when the observation moves from a state where the event of interest does not or cannot occur to one in which the event does occur according to a P process with rate λ (the *events stage*). The probability of observation i making such a transition is denoted p, and the observed dependent variable takes on a value of zero for all the observations that do not make the transition; p is modeled using logistic regression. The covariates used for the logistic and count components may differ (Lambert 1992); in my case, I included only JWL indices to model the zero inflation, while allowing all the explanatory variables (including JWL indices) to enter into the count component of the model. The absence of a lightning flash or high fuel moisture will tend to hold a cell in the transition stage; if conditions are receptive, then ignition may occur (the transition is made), and the number of lightning-caused fires actually igniting may be influenced by landscape characteristics such as vegetation or linear features. I implemented ZIP regression using the R function *zeroinfl* contained in the package **pscl** (Jackman 2005). I also fitted P and NB GLMs and used the Vuong test (Vuong 1989) to compare the best ZIP with the best P or NB model. This test is a formal test for non-nested models that determines statistically whether the zero-inflated model is a significant improvement over the other alternatives.

I verified the linearity of the relationship between response and explanatory variables using Generalized Additive Models (GAMs), specifying a Poisson or a Binomial error. These tests were conducted at both spatial resolutions as well as for three seasons separately. The results of such models were used to plot adjusted non-parametrically smoothed estimates of the effect of the covariates, which were visually inspected for nonlinearities. If these were detected, appropriate non-linear transformations were included for the corresponding covariate terms when fitting other model structures (P, NB, or ZIP). I assessed collinearity between regressors using Spearman pair-wise correlation coefficient. The correlation between road and pipeline density was high at the coarse ($r_s = 0.78$) and the fine ($r_s = 0.72$) scales. According to Bonate (1999), when covariates showing a high degree of correlation ($r_s >> 0.5$) are included in the same model, standard error estimates may be inflated, indicating that one or both are not relevant to the structural model when in fact they are. In order to account for this confounding effect, I compared models including pipeline or road density, both, and their interaction. Remaining variables had pairwise correlation coefficients $r_s \leq |0.6|$, indicating multi-collinearity in terms should not affect the analysis (Bonate 1999).

The main motivation for this research was to assess the effect of linear features on lightning-caused fire ignition; however, I also included forest composition, weather and geography given their previously demonstrated importance (Krawchuk *et al.* 2006, and references therein) and assumed this would yield models with higher explanatory and predictive power. Furthermore, I had no explicit prior hypothesis about interactions between covariates, so no interaction terms were included. I used a hierarchical model selection approach using Akaike's Information Criteria (AIC; Burnham and Anderson 1998) to determine the best model overall. A change in AIC ≥ 5 was considered evidence of a substantial change in the model's descriptive ability; if this change was < 5, I selected a simpler model over a more parameterized model. First, a global linear feature model (LF_M) with all three classes of linear features was reduced by backward elimination to the most parsimonious model LF_{BM}. Similarly, a global vegetation model (VE_M) was reduced to a simpler model VE_{BM}. Finally, a global geography model (GEO_M), including both E and N, was reduced to GEO_{BM}. LF_{BM}, VE_{BM} and GEO_{BM} were combined into LVG_M, where one of the weather variables (D-L or F-L) and total area (TA) were also included. From these two LVG_M models, I selected the one with the lowest AIC value and further reduced it in order to yield the final best model LVG_{BM}. This process was repeated three times in order to select the best model fitted using P (P-LVG_{BM}), NB (NB-LVG_{BM}), and ZIP (ZIP-LVG_{BM}) regression; which were subsequently compared using the Voung test to determine the best model overall $\mathbf{LVG_{BM}}$. I also present the null and global (including all covariates) models only for comparison purposes.

I checked for outliers in the selected best model $\mathbf{LVG_{BM}}$ using graphs of deviance and Pearson residuals (Cameron and Trivedi 1998). If large residuals were identified, I assessed the change in the magnitude and significance of the coefficients in a model fitted to a dataset without such identified observations. Further graphical assessment of residuals was conducted following Cameron and Trivedi (1998, p. 144).

I evaluated the adequacy of the best-fitting model $\mathbf{LVG_{BM}}$ using a parametric bootstrapping procedure (Kéry *et al.* 2005). The number and identity of covariates was fixed to the one obtained for the best model, and 1 000 replicate P, NB or ZIP data sets were generated, depending on the distributional assumption of $\mathbf{LVG_{BM}}$. For each bootstrapped data set, covariate coefficients were re-estimated and a fit statistic (sum of squared errors (SSE)) was computed. This collection of simulated values of SSE forms the reference distribution to which the observed SSE was compared. I calculated the proportion of the bootstrapped SSE distribution that had equal or more extreme values than the one obtained for $\mathbf{LVG_{BM}}$; if this value was smaller than 0.05, it was decided that the model fit was not satisfactory. Model performance was assessed by comparing observed proportions versus predicted probabilities for each count category (Lambert 1992). Finally, the selected top models were validated using a 1:7 testing to training *k*-fold partitioning procedure (Harrell

2001). This technique randomly divides the original dataset into seven groups; the model is trained iteratively on a dataset without each group, and then tested on the omitted group. The mean prediction error (observed - predicted) from this testing procedure was calculated and reported with respect to the mean prediction error obtained using the null model.

I assessed seasonality of fire-day counts at the coarse-spatial resolution by classifying each fire as a spring (30 April to 31 May), summer (1 June to 31 July) or fall fire (1 August to 29 September) (dates follow C. Tymstra, personal communication, 2005). Seasonal fire-day counts were analyzed with respect to the four covariates classes described earlier, except that the JWL indices were replaced with their seasonal analogues. These are denoted with subscripts SP, SU, and FA for spring, summer and fall, respectively.

Finally, I used the selected best model obtained from the coarse-resolution analysis to estimate the number of fire-days that would have occurred in the study area during the eight-year interval under two hypothetical scenarios: (1) no roads present, and (2) a 'road-saturated landscape', where every cell was assigned a road density of 0.85 km/km^2. This value corresponds to the upper 95 percentile of the current road density distribution in my study area.

2.3 Results

Of the 601 coarse-scale cells initially selected for analysis, three adjacent townships had unusually high values of $D\text{-}L_t$ and $F\text{-}L_t$ but no fires. Most of these identified receptive days occurred during the summer of 1998, a year with mean annual temperature 1.2°C above provincial normals and below normal precipitation for the northeast region of the province (ASRD 2001). This was accompanied by a marked increase in the number of wildfires, with 1 698 fires occurring that year (ASRD 2004), compared to a long-term provincial average of 852 fires per year. The three identified cells did not follow such general pattern, and in fact were drivers of a negative relationship between JWL indices and fire-day frequency; such a relationship is entirely implausible. Thus, I considered these cells to be outliers and dropped them from analysis.

Over the study interval, the final sample of 598 townships had 1 043 lightning-caused fires which burned a total area of 1 905 km^2. Only 13% of cells recorded multiple fires on a single day (see section 2.2.2.2). After removing these multiple fire events, a total of 919 fire-days remained for analysis (Fig. 2.1b). The mean fire-day count was 1.54 (s.d. = 1.49) and the variance to mean ratio was 1.44, indicating the data was overdispersed. Almost 60% of the observations fell within the zero and one categories, with no apparent difference, indicating an excess of zeros compared to that expected under Poisson (Fig. 2.5). Fire-day counts varied among seasons, with 13%, 65% and 22% occurring during spring, summer and fall, respectively (Fig. 2.6).

2.3.1 Do linear features influence the frequency of lightning-caused fire-days per township?

Seismic lines were the most abundant linear feature class at the coarse-resolution grid, with a mean of 1.77 km/km^2 and a maximum value of 16.1 km/km^2 (Table 2.1). Mean road and pipeline density was 0.35 km/km^2 and 0.29 km/km^2, respectively (Table 2.1). The proportional areas of the six forest types ranged from 0 to 92% cover, with black spruce the most abundant ($\overline{x} = 0.45$, s.d. = 0.20) and cutblocks the least abundant class ($\overline{x} = 0.03$, s.d. = 0.05). D-L_t had a maximum of 30 and a mean value of 15.4 days; the maximum and mean values for F-L_t were 34 and 16.8, respectively.

The top two models were ZIP-LVG_{BMt} and NB-LVG_{BMt}, which showed a ΔAIC = 3 between each other (Table 2.2). The use of NB over a P model was justified (ΔAIC = 9) (Table 2.2), consistent with the overdispersion detected in the raw data. Predictions from models P-LVG_{BMt}, NB-LVG_{BMt} and ZIP-LVG_{BMt}

fell very close to the observed values (Fig. 2.7), but the P model showed the largest discrepancy, particularly for the zero and one classes. The ZIP model predictions were the closest to the observed proportions; however, this model was not a significant improvement over the NB model (Vuong test statistic = 1.22, $p = 0.11$). Thus, I selected NB-LVG_{BMt} as the best model overall.

The maximum likelihood estimates and standard errors for each term included in NB-LVG_{BMt} are shown in Table 2.3. Increasing road density and the proportion of white spruce stands per cell leads to an increase in the expected number of fire-days (Figs. 2.8a and d). Increasing proportions of recently burned areas and deciduous-dominated forest have a negative effect on the conditional mean (Figs. 2.8b and c). Finally, I found a negative influence of easting, indicating that the expected fire-day count increases by a factor of 1.86 from the western end to the eastern border of the study area. However, the magnitude of this effect was not large, particularly when compared to the rest of the covariates in the model: road density increased the conditional mean by a factor of 2.04 over its range of recorded values, while the analogue measure for WS_t was 3.08. BU_t and DE_t decreased the conditional mean by a factor of 0.38 and 0.20, respectively, over their range of recorded values. The relative importance of individual covariates was assessed by the difference in AIC value resulting from their removal from the final model (Table 2.3, last column). The effects of forest composition, particularly the proportion of white spruce forest, were more important than anthropogenic linear features and easting.

Seven-fold validation of model NB-LVG_{BMt} yielded a mean prediction error of 1.09, which represents a 9% reduction with respect to the error associated with the null model. Graphical inspection of residuals identified one potential outlier. When I refitted the selected best model to a dataset excluding this point, the significance and signs of the coefficients did not change while their magnitudes remained within one S.E. of the original value, supporting the robustness of NB-LVG_{BMt}. Based on a parametric bootstrap, the selected best model fit the data adequately, as evidenced by a SSE value of 1 127, which fell well within the simulated distribution of SSEs ($p = 0.68$).

Using model NB-LVG_{BMt} (Table 2.3), I estimated that if there had been no roads in the study area, only 732 (S.E. = 69) fire-days would have occurred during the study interval, which represents a 20% reduction with respect to the original value recorded. Similarly, a 'road-saturated landscape' (see section 2.2.2.4 for definition) would experience 1 255 (S.E. = 127) fire-days, representing a 37% increase with respect to the actual value recorded.

2.3.2 Does the apparent effect of linear features change when the spatial resolution is increased?

Of 2 370 quarter-townships considered for analysis, one had an extreme value for D-L_q and F-L_q. This cell corresponds to the south west quadrant of one of the three adjacent townships identified as outliers at the coarse-scale analysis (see section 2.3.1). Accordingly, this cell was dropped from the fine-scale analyses.

There were 916 fire-days within the 2 369 quarter-townships, with a mean of 0.39 (s.d. = 0.68) and a variance to mean ratio of 1.21. Figure 2.9 shows that the maximum number of fire-days recorded per quarter-township was five, though this was fairly uncommon representing less than 1% of the cells. Also, it is evident from the graph that there is a slight excess of zeros with respect to that expected under a Poisson distribution (Fig. 2.9).

Linear features were widespread at the fine resolution, with only 4% of the cells completely free of roads, pipelines or seismic lines. Maximum seismic line density was 31 km/km^2; mean road and pipeline densities were similar to the values recorded at the coarse scale, though their standard deviations were higher (Table 2.1). As expected, more spatial heterogeneity in linear feature densities was captured at the fine resolution. This was consistent for forest composition, where there was higher variability in the recorded proportions among quarter-townships (Table 2.1); at this scale, the proportion of all forest classes ranged from 0 to 99% cover. D-L_q had a maximum of 12 and a mean of 3.9 days; F-L_q had a maximum and mean value of 15 and 4.2, respectively (Table 2.1).

Models ZIP-LVG_{BMq} and NB-LVG_{BMq} were equivalent based on AIC and included the same covariates (Table 2.4). Predictions from all the three top models followed closely the observed proportions (Fig. 2.10). The P model was relatively poor, based on AIC (ΔAIC = 13; Table 2.4); while the ZIP model did not provide a significant improvement over the NB model (Vuong test statistic = 0.02, p = 0.49). Accordingly, I selected NB-LVG_{BMq} as the best model overall; its parameter estimates and standard errors are shown in Table 2.5.

In general, the effect of the covariates was very similar to that observed at the coarse resolution (Table 2.3), though at this scale the dispersion parameter θ declined to half the value obtained at the township scale. Additionally, I detected a quadratic relationship between the abundance of harvested areas and the conditional mean (Fig. 2.11). Based on the change in model AIC, forest composition was the most important influence on fire-day counts when compared to that of the other covariate types (Table 2.5, last column), consistent with the findings at the coarse-scale resolution (Table 2.3).

Seven-fold validation of model NB-LVG_{BMq} yielded a mean prediction error of 0.52, which represents a 5% reduction with respect to the error derived from the null model. Graphical inspection of residuals suggested three potential outliers; however, when these were removed from the dataset, there was no change in the sign or significance of the coefficients in the best model and their magnitudes remained within one S.E. of the original value. The selected best model fit the data adequately (SSE = 1 038, p = 0.50).

2.3.3 Does the influence of linear features change seasonally?

This analysis was conducted only at the coarse scale, thus the term 'cell' should be understood to refer to township throughout this section. There were 117 fire-days in spring, 598 in summer and 204 in fall. Figure 2.6 shows the distribution of the number of fire-days per township for each season. When these counts were compared to a P model with parameter lambda set to the mean fire-day count for each season, only summer counts showed an apparent excess of zeros. During summer and fall, the maximum per cell fire-day count was five; while in spring the maximum was three fire-days per cell (Fig. 2.6).

Mean spring fire-day count was 0.20 (s.d. = 0.48), and the variance to mean ratio was 1.20. The three top models had very similar AIC scores and selected mostly the same covariates, except that RO was missing from NB-$LVG_{BM,SP}$ (Table 2.6). There was little evidence in favour of ZIP over P (Vuong test statistic = 1.02, p = 0.15) or NB (Vuong test statistic = 0.24, p = 0.41) models, so it was discarded. The remaining top two models had ΔAIC of $\approx$18.5 with respect to the null model but explained only 6% of the null deviance. Model P-$LVG_{BM,SP}$ had an SSE value of 136 (p = 0.08) while NB-$LVG_{BM,SP}$ had an SSE value of 136 (p = 0.5). Overall, there was insufficient evidence to select between P-$LVG_{BM,SP}$ and NB-$LVG_{BM,SP}$. Both of them included a negative effect of recently burned areas and deciduous-dominated stands, and a positive effect of easting; only model P-$LVG_{BM,SP}$ included a significant positive effect of road density (Table 2.7).

Mean summer fire-day count was 1.0 (s.d. = 1.14), and the variance to mean ratio was 1.29. Model ZIP-$LVG_{BM,SU}$ and NB-$LVG_{BM,SU}$ had a ΔAIC = 4 (Table 2.6). However, there was insufficient evidence to prefer the ZIP model (Vuong test statistic = 1.02, p = 0.15), so I selected NB-$LVG_{BM,SU}$ as the best overall model. It included a positive effect of road density and proportional area of white spruce forest; while deciduous-dominated stands, recently burned areas and easting were negatively related to the conditional mean (Table 2.7). $F\text{-}L_{t,SU}$ had a positive effect, though this was not very strong and only significant at the 1% level. This model explained 9% of the null deviance and fit the data adequately (SSE = 696, p = 0.53).

In fall, the mean fire-day count was 0.34 (s.d. = 0.43), and the variance to mean ratio was 1.26. The top three ZIP, P and NB models had very similar AIC values, and there was no support for any one in particular (Table 2.6). Each included the same covariates with the same signs and similar parameter estimates. Mainly for inferential purposes, I chose the simplest model $P\text{-}LVG_{BM,FA}$ as the best of the three. Fall fire-day counts were significantly influenced by deciduous and white spruce dominated stands (Table 2.7). Intermediate abundance of black spruce within a township was associated with increased frequency of fire-days. Intermediate values of $F\text{-}L_{t,FA}$ were associated with decreased fire-day counts, while under high $F\text{-}L_{t,FA}$ the conditional mean was increased (Table 2.7). Model $P\text{-}LVG_{BM,FA}$ explained 16% of the null deviance and fit the data adequately (SSE = 218; p = 0.18).

2.4 Discussion

Lightning-caused fire ignition results from the interaction between broad-scale weather patterns and the flammability properties of the fuels on which lightning flashes strike. I found that this process is also positively influenced by at least one class of anthropogenic linear feature, namely roads. Their influence was consistent between the fine (≈2 400 ha cells) and coarse (≈10 000 ha cells) scales, indicating that the inclusion of this covariate in the best models was not arbitrary, and supporting the robustness of the analysis across the two spatial resolutions. Analysis of seasonal variation in fire-day frequencies indicated that the effect of roads was strongest in summer, possibly weaker in spring and non-detectable during the fall.

Roads are landscape elements that have been demonstrated to produce significant changes in landscape structure and composition (Forman and Alexander 1998; McGarigal *et al.* 2001). Of all the disturbances that humans have introduced into natural terrestrial environments, roads may be the most ubiquitous and significant legacy of our activities (Forman and Alexander 1998). In western boreal forests, roads have been previously shown to influence fire regimes through increased human-caused fire ignitions (Vega Garcia *et al.* 1995).

Roads can also affect many other ecological processes; for example, roads may serve as a conduit for the movement of organisms across the landscape, including the deliberate or accidental spread of non-native plant species (Trombulak and Frissell 2000). Of particular interest here is the increase in fuel-bed flammability resulting from native and exotic grass invasions along roadsides, which can potentially affect the frequency, size, spatial pattern, and, in some cases, intensity of fires (D'Antonio and Vitousek 1992). Some of these effects have been thoroughly described in the plains of western North America and the Mojave Desert, where the invasion of alien annual grasses has increased fire frequency and produced other changes in the fire regime (Brooks *et al.* 2004).

In the boreal mixedwood forest of Saskatchewan, Sumners (2005) found that vegetation quadrats next to roads contained the highest amount of exotics compared to mature mixedwood stands and recently disturbed areas by timber harvesting and wildfires, with 94% of these quadrats containing at least one non-native plant. He found a total of 22 non-native species, with nine species belonging to the *Gramineae* family. This addition of non-native species of grasses to the native pool already thriving along roadsides represents a new source of fuel (Stoner *et al.* 2004), since many perennial grasses tend to produce litter that is highly combustible (Hogenbirk and Sarrazin-Delay 1995). I suggest that this might be the underlying mechanism for the positive road effect I detected. Thus, it is not the case that lightning flashes are somehow attracted to roads, since roads are not flammable *per se*. What this means is that when a lightning flash discharges close to a roadside, it strikes an area with a higher proportion of flammable fine fuels than what is generally available within forested areas.

Pipelines and seismic lines are revegetated with similar general agronomic grass mixes as roadsides (Revel *et al.* 1984); and several non-native species of grasses have been found established on seismic lines (MacFarlane 2003). Thus, one might expect pipelines and seismic lines to have a positive effect on lightning fire ignition as well. However, I did not detect such effect.

Cameron *et al.* (1997) noted that areas that are subjected to consistent disturbance favor the establishment and survival of exotic and weedy species; roadsides are continuously disturbed for maintenance purposes (e.g. to maintain visibility for wild animals crossing), while seismic lines experience a broad range of disturbance frequencies, from complete abandonment after initial construction to yearly reuse for oil and gas exploration (MacFarlane 2003). This suggests that seismic lines and roads should differ in the relative abundance of grasses growing on them, with grasses being a more consistent and long-term inhabitant of the latter. Seismic lines are also narrower than roads, so it is expected that their microclimate conditions will be strongly affected by the adjacent forest canopies (Carlson and Groot 1997). Thus in some mixedwood stands, the shadow provided by nearby tall trees might help retain plant moisture; while in more open areas, adjacent to cutblocks or on line intersections for example, direct sunlight might reduce the water content of the vegetation on the seismic line (Revel et al. 1984). Additionally, Lee and Boutin (2006) noticed that tracked use of seismic lines adjacent to black spruce bogs alter their local hydrology, converting them into a very wet fen dominated by sedges. This suggests that the influence of individual seismic lines on lightning-caused fire ignition will be positive or negative depending on the details of their moisture condition, an attribute which would be hard to measure at the scale required for this analysis.

Pipelines are less abundant than roads within my study region; but their spatial distributions are similar, and pipelines are also consistently disturbed for maintenance. I found that road and pipeline densities were positively correlated at both scales of analysis (see section 2.2.2.4), which could confound statistical inference. I addressed this issue by assessing the relative strengths of evidence for alternate models with covariates for road density, pipeline density, both, as well as their interaction. I found that the effect of roads consistently dominated that of pipelines; with pipeline density being a significant predictor only when no other linear feature covariates were present in the same model. This suggests that the road effect disguises the influence pipelines may potentially have; and indicates that the approach I followed was not able to conclusively tear apart each separate effect, but more likely identified roads as a surrogate for the effect other anthropogenic linear features such as pipelines may have on the spatial patterns of lightning-caused fire ignition.

All of my models incorporated covariates for fire weather and lightning through a joint fire weather and lightning index (JWL). Krawchuk *et al.* (2006) unequivocally demonstrated the influence of JWL indices on lightning-caused fire ignition; however, I only detected evidence for a weak effect constrained to summer and fall. I suspect the reason lies in the differences in temporal scale between these studies. JWL indices quantified the total number of days where lightning and appropriate low moisture conditions were detected in a cell; in prior studies, these have been measured at annual scales, while I used values summed over eight years. Thus, inter-annual variation in weather-lightning conditions within cells, summed over several years, might even out the true effect. This is supported by Krawchuk *et al.* (2006), who found no evidence for stable inter-annual spatial patterns in JWL indices in this region over an earlier interval (1983-1993). A possible solution to describe weather-lightning patterns beyond an annual scale might be to weight each year's contribution by a measure of the regional climatic conditions for that year, such as the Seasonal Severity Rating index (SSR; Van Wagner 1987).

I found that the effect of forest composition on spatial variation in fire-day counts dominated that of all other covariates. Previously burned areas and deciduous-dominated stands decreased the expected mean, while white spruce-dominated stands had a positive influence. These results are consistent with findings by Krawchuk *et al.* (2006) for the same study region. I also detected a quadratic relationship between the frequency of fire-days and the proportion of harvested areas, but only at the fine-scale resolution. This relationship was not very strong, as evidenced by a ΔAIC = 3 after the deletion of the linear and quadratic terms on CU_q (Table 2.5). However, I decided to retain them in the final best fine-scale model.

In northeastern Alberta, industrial-scale forestry began during the 1940s, focusing mainly on leading white spruce stands; but intensified markedly after 1993 when deciduous dominated stands became the harvest target after the construction of the AlPac pulp-mill (AlPac 2001). In mesic stands, regenerating cutblocks are initially inhabited by grasses (Lieffers *et al.* 1993) and gradually replaced by aspen, which typically dominates for the following half century or so regardless of the pre-harvest canopy composition or all but the most extreme silvicultural treatments (Navratil *et al.* 1990). Thus, recently harvested areas have been suggested to experience increased fire hazard as a result of dry and abundant harvest debris and grass fine fuels (Kiil 1968, Baxter 2003). However, such effect is considered transitory, decreasing with the increase in aspen dominance, a tree species that has been shown to reduce fire ignition potential (Krawchuk *et al.* 2006). In my study area, I found that the proportion of cutblocks per quarter-township was negatively correlated with the year when harvesting started within that cell ($r_s = -0.22$, $p < 0.001$); and that most of these activities occurred, on average, during the four years following the onset of harvesting. This supports the idea that quarter-townships with a high proportion of cutblocks represent areas harvested more than 10 years ago, while cells with a low proportion of cutblocks have been targeted fairly recently for forestry activities. In such case, the detected quadratic relationship is supported, whereby increased ignition frequency is associated with recent cutblocks in medium harvested cells, while decreased ignitions are associated with old cutblocks in highly harvested cells.

Observed spatial patterns and the relationship between pattern and process are a function of the spatial scale of analysis (O'Neill *et al.* 1996). Rare landscape elements are typically less well represented as resolution decreases and spatial complexity decreases as well (Cain *et al.* 1997). As a general rule, processes at finer scales tend to be driven by a larger number of factors (O'Neill *et al.* 1996). In this instance, increasing the spatial resolution did not reveal an effect of pipelines or seismic lines, which was what I expected. On the other hand, increasing the spatial scale from ≈10 000 ha to ≈2 400 ha did reveal an effect of harvested areas. Despite the 50+ years of harvesting history in the study region, and its recent intensification, harvested areas are not yet a widespread element in this landscape, covering only 2.6% of the area as of the year 2004. However, harvested areas do tend to be spatially aggregated where they occur and, as might have been predicted from Cain *et al.* (1997), the effect of this 'rare' fuel type was evident only after increasing the spatial resolution of analysis.

Laboratory studies have shown that moisture content is an important determinant of lightning-caused fire ignition potential (Latham and Schlieter 1989). Fuel moisture varies seasonally as a consequence of phenology (i.e. cured grasses, changes in canopy cover, etc.) and changes in temperature, precipitation and relative humidity. I found seasonal variation in the influence of natural and anthropogenic factors on lightning-caused fire-day counts. In particular, the effect of roads was strongest in summer and possibly spring. Grasses on roadsides typically produce standing dead fuels that dry rapidly in response to low soil moisture and atmospheric humidity, increasing their flammability and promoting fire ignitions earlier in the spring as well as during the heart of the fire season (Brooks *et al.* 2004). In fall, I found the number of fire-days per cell to be determined exclusively by fuels and weather-lightning conditions, with no detectable effect of linear features. This was unexpected, as grasses have been found to promote fire ignition in the fall as well (Brooks *et al.* 2004). Similarly, I expected to detect an effect of pipelines or seismic lines during spring, as several fire managers have suggested these features to be highly fire-prone during that time of the year (C. Tymstra, personal communication, 2005). However, my analyses did not support such hypotheses, suggesting that this observed effect is not widespread.

I conceived lightning-caused fire-day frequency per unit area as a two-stage process: a *transition* stage, where a particular cell might or might not experience appropriate weather-lightning conditions conducive to fire ignition; and a subsequent *events* stage, where the number of fires per 'transitioned' cell is then determined by local landscape characteristics. Accordingly, I fit Zero-inflated Poisson regression models (Lambert 1992), but found no evidence to favor these over simpler Poisson or Negative Binomial models. The lack of significance of JWL indices at both analyzed spatial resolutions might be responsible for the poor performance of the ZIP models I detected. I conclude though that, over the long term, spatial variation in fire-day frequency per unit area is better described by a simpler one-stage point process determined by a single parameter, the process intensity or mean, which varies among landscapes depending on their local characteristics and, to some degree, regional climatic trends.

Well-quantified relationships between the biophysical environment and the frequency of fire ignitions are essential to form a complete understanding of fire regimes in boreal and other ecosystems. After accounting for the influences of forest fuels, weather and lightning conditions, and geography; I found a significant effect of one anthropogenic linear disturbance, roads, on the frequency of lightning-caused fires in the forests of northeastern Alberta. My results also showed that a hypothetical 'road-saturated' landscape would experience a 37% increase in the frequency of days where lightning-caused fires occur. This suggests that projected increases in road density, associated with future industrial developments in the region (Schneider *et al.* 2003), may lead to major changes in the fire regime and possibly to subsequent ecosystem-level transformations (Mack and D'Antonio 1998). In the face of these projected landscape changes, the increased fire disturbance regime resulting from human related activities will prove an important consideration in the pursuit of sustainable forest and ecosystem management.

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Table 2.1: A summary table of explanatory variables used for analyses.

Name	Variable	$\bar{x}^{a}$	s.d.[a]	5 – 95%[a]	$\bar{x}^{b}$	s.d.[b]	5 - 95%[b]
RO	Road density (km per km^2 of dry land)	0.35	0.28	0 - 0.85	0.35	0.34	0 - 1.00
PL	Pipeline density[d]	0.29	0.35	0 - 1.08	0.29	0.49	0 - 1.17
SL	Seismic line density[d]	1.77	1.67	0.12 - 3.88	1.77	2.02	0.04 - 3.86
DE	Deciduous (proportional area)	0.20	0.15	0.01 - 0.48	0.21	0.18	0 - 0.53
WS	White spruce[d]	0.05	0.05	0 - 0.16	0.05	0.66	0 - 0.19
BS	Black spruce[d]	0.45	0.20	0.12 - 0.79	0.45	0.24	0.08 - 0.85
PI	Jack pine[d]	0.08	0.09	0 - 0.27	0.08	0.11	0 - 0.30
BU	Recent burns[d]	0.07	0.17	0 - 0.54	0.07	0.20	0 - 0.62
CU	Cutblocks[d]	0.03	0.05	0 - 0.14	0.03	0.06	0 - 0.15
D-L	Joint DMC-lightning index	15.4	4.65	8 - 24	3.87	2.14	1 - 8
F-L	Joint FFMC-lightning index	16.8	4.29	10 - 24	4.21	2.16	1 - 8
E[c]	NAD 27 UTM easting at cell centroid (in units of 10^5 m)	4.25	0.77	2.90 - 5.47	4.25	0.76	2.89 - 5.45
N[c]	NAD 27 UTM northing at cell centroid (in units of 10^6 m)	6.24	0.07	6.11 - 6.37	6.24	0.08	6.11 - 6.37
TA[c]	Sampling unit area (in units of 10^3 ha)	9.49	0.12	9.45 - 9.54	2.37	0.01	2.36 - 2.38

[a] Values reported correspond to the coarse-scale grid (≈10 000 ha cells). $\bar{x}$ = mean, s.d. = standard deviation, 5 - 95% = 5 and 95 percentiles of the observed distribution.

[b] Values reported correspond to the fine-scale grid (≈2 400 ha cells).

[c] E, N and TA were included in analyses after scaling them down by a factor of 10^5, 10^6 and 10^3, respectively. See section 2.2.2.3 for details.

[d] PL and SL have the same units as RO. WS, BS, PI, BU, and CU have the same units as DE.

Table 2.2: Zero-inflated (ZIP), Poisson (P) and Negative Binomial (NB) models for the number of lightning-caused fire-days per township (≈10 000 ha) over the period 1995-2002, with relative measures of model support. Models were fitted to a sample of 598 townships located in northeastern Alberta.

Model	Terms[a]	k[b]	AIC[b]	ΔAIC[b]
$Null_t$	~1	1	2020	124
$ZIP\text{-}GL_t$	$x \sim D\text{-}L_t + F\text{-}L_t$; $z \sim RO_t + PL_t + SL_t + BU_t + DE_t + WS_t + BS_t + BS_t^2 + PI_t + CU_t + D\text{-}L_t + F\text{-}L_t + E_t + N_t + TA_t$	22	1907	12
$ZIP\text{-}LVG_{Mt}$	$x \sim D\text{-}L_t$; $z \sim RO_t + BU_t + DE_t + WS_t + BS_t + BS_t^2 + CU_t + D\text{-}L_t + E_t + N_t + TA_t$	14	1898	3
$ZIP\text{-}LVG_{BMt}$	$x \sim D\text{-}L_t$; $z \sim RO_t + BU_t + DE_t + WS_t + BS_t + BS_t^2 + D\text{-}L_t + E_t$	11	1895	0
$P\text{-}GL_t$	$\sim RO_t + PL_t + SL_t + BU_t + DE_t + WS_t + BS_t + BS_t^2 + PI_t + CU_t + D\text{-}L_t + F\text{-}L_t + E_t + N_t + TA_t$	16	1917	22
$P\text{-}LVG_{Mt}$	$\sim RO_t + BU_t + DE_t + WS_t + BS_t + BS_t^2 + CU_t + F\text{-}L_t + E_t + N_t + TA_t$	12	1910	15
$P\text{-}LVG_{BMt}$	$\sim RO_t + BU_t + DE_t + WS_t + D\text{-}L_t + E_t$	7	1907	12
$NB\text{-}GL_t$	$\sim RO_t + PL_t + SL_t + BU_t + DE_t + WS_t + BS_t + BS_t^2 + PI_t + CU_t + D\text{-}L_t + F\text{-}L_t + E_t + N_t + TA_t ; \theta$	17	1909	14
$NB\text{-}LVG_{Mt}$	$\sim RO_t + BU_t + DE_t + WS_t + BS_t + BS_t^2 + CU_t + D\text{-}L_t + E_t + N_t + TA_t ; \theta$	13	1902	7
$NB\text{-}LVG_{BMt}$	**$\sim RO_t + BU_t + DE_t + WS_t + E_t ; \theta$**	**7**	**1898**	**3**

Note: The best model overall is shown in bold.

[a] Variable names are described in Table 2.1.

[b] k = number of parameters estimated in the model. AIC = Akaike Information Criteria. ΔAIC = Increase in AIC score in any one model with respect to the model with lowest AIC score.

Table 2.3: Maximum likelihood estimates, confidence intervals and p-values for each term included in model NB-LVG$_{BMt}$ (Table 2.2).

Term[a]	β	S.E.	p-value	ΔAIC[b]
β_0	1.27	0.24	<0.01	
RO_t	0.63	0.13	<0.01	20.7
BU_t	-1.14	0.29	<0.01	14.7
DE_t	-1.97	0.31	<0.01	39.6
WS_t	3.96	0.74	<0.01	24.2
E_t	-0.20	0.05	<0.01	13.5
Theta (θ)	6.67	2.22		

[a] Variable names are described in Table 2.1.

[b] ΔAIC = Increase in AIC when each term is removed from the model.

Table 2.4: Zero-inflated (ZIP), Poisson (P) and Negative Binomial (NB) models for the number of lightning-caused fire-days per quarter-township (≈2 400 ha) over the period 1995-2002, with relative measures of model support. Models were fitted to a sample of 2 369 quarter-townships located in northeastern Alberta.

Model	Terms[a]	k[b]	AIC[b]	ΔAIC[b]
$Null_q$	~1	1	3915	160
$ZIP\text{-}GL_q$	$x \sim D\text{-}L_q + F\text{-}L_q$; $z \sim RO_q + PL_q + SL_q + BU_q + DE_q + WS_q + BS_q + PI_q + PI_q^2 + CU_q + CU_q^2 + D\text{-}L_q + F\text{-}L_q + E_q + N_q + TA_q$	20	3767	12
$ZIP\text{-}LVG_{Mq}$	$x \sim F\text{-}L_q$; $z \sim RO_q + BU_q + DE_q + WS_q + PI_q + CU_q + CU_q^2 + F\text{-}L_q + E_q + N_q + TA_q$	14	3759	4
$ZIP\text{-}LVG_{BMq}$	$x \sim 1$; $z \sim RO_q + BU_q + DE_q + WS_q + CU_q + CU_q^2 + E_q$	9	3755	0
$P\text{-}GL_q$	$\sim RO_q + PL_q + SL_q + BU_q + DE_q + WS_q + BS_q + PI_q + PI_q^2 + CU_q + CU_q^2 + D\text{-}L_q + F\text{-}L_q + E_q + N_q + TA_q$	17	3776	22
$P\text{-}LVG_{Mq}$	$\sim RO_q + BU_q + DE_q + WS_q + PI_q + CU_q + CU_q^2 + D\text{-}L_q + E_q + N_q + TA_q$	12	3770	15
$P\text{-}LVG_{BMq}$	$\sim RO_q + BU_q + DE_q + WS_q + PI_q + CU_q + CU_q^2 + E_q$	8	3768	13
$NB\text{-}GL_q$	$\sim RO_q + PL_q + SL_q + BU_q + DE_q + WS_q + BS_q + PI_q + PI_q^2 + CU_q + CU_q^2 + D\text{-}L_q + F\text{-}L_q + E_q + N_q + TA_q$; θ	18	3764	9
$NB\text{-}LVG_{Mq}$	$\sim RO_q + BU_q + DE_q + WS_q + PI_q + CU_q + CU_q^2 + D\text{-}L_q + E_q + N_q + TA_q$; θ	13	3757	2
$NB\text{-}LVG_{BMq}$	**$\sim RO_q + BU_q + DE_q + WS_q + CU_q + CU_q^2 + E_q$; θ**	**9**	**3755**	**0**

Note: The best model overall is shown in bold.

[a, b] As in Table 2.2.

Table 2.5: Maximum likelihood estimates, confidence intervals and p-values for each term included in model NB-LVG$_{BMq}$ (Table 2.4).

Term[a]	β	S.E.	p-value	ΔAIC[b]
β_0	-0.05	0.22	0.81	
RO_q	0.47	0.10	<0.01	18.1
BU_q	-1.23	0.26	<0.01	24.7
DE_q	-2.03	0.25	<0.01	68.3
WS_q	2.91	0.54	<0.01	24.3
CU_q	3.33	1.29	0.01	3.1
CU_q^2	-8.71	4.44	0.05	
E_q	-0.20	0.05	<0.01	15.2
Theta (θ)	3.33	1.00		

[a, b] Same as Table 2.3.

Table 2.6: Selected best models of spring, summer and fall fire-day counts per township (≈10 000 ha) over the period 1995-2002, with relative measures of model support. Models were fitted separately for each season. The sample dataset consisted of 598 townships located in northeastern Alberta.

Season	Model	Terms[a]	k[b]	AIC[b]
SP[c]	Null_{SP}	~ 1	1	648
	$\mathrm{ZIP\text{-}LVG}_{BM,SP}$	$x \sim 1,\ z \sim RO_t + BU_t + DE_t + E_t$	6	628
	$\mathrm{P\text{-}LVG}_{BM,SP}$	$\sim RO_t + BU_t + DE_t + E_t$	5	631
	$\mathrm{NB\text{-}LVG}_{BM,SP}$	$\sim BU_t + DE_t + E_t\ ;\ \theta$	5	629
SU[c]	Null_{SU}	~ 1	1	1649
	$\mathrm{ZIP\text{-}LVG}_{BM,SU}$	$x \sim 1,\ z \sim RO_t + BU_t + DE_t + WS_t + \log(F\text{-}L_{t,SU}) + E_t$	8	1575
	$\mathrm{P\text{-}LVG}_{BM,SU}$	$\sim RO_t + BU_t + DE_t + WS_t + \log(F\text{-}L_{t,SU}) + E_t$	7	1584
	$\mathrm{NB\text{-}LVG}_{BM,SU}$	**$\sim RO_t + BU_t + DE_t + WS_t + \log(F\text{-}L_{t,SU}) + E_t\ ;\ \theta$**	**8**	**1579**
FA[c]	Null_{FA}	~ 1	1	921
	$\mathrm{ZIP\text{-}LVG}_{BM,FA}$	$x \sim 1,\ z \sim DE_t + WS_t + BS_t + BS_t^2 + F\text{-}L_{t,FA} + F\text{-}L_{t,FA}^2 + E_t$	9	847
	$\mathrm{P\text{-}LVG}_{BM,FA}$	**$\sim DE_t + WS_t + BS_t + BS_t^2 + F\text{-}L_{t,FA} + F\text{-}L_{t,FA}^2 + E_t$**	**8**	**846**
	$\mathrm{NB\text{-}LVG}_{BM,FA}$	$\sim DE_t + WS_t + BS_t + BS_t^2 + F\text{-}L_{t,FA} + F\text{-}L_{t,FA}^2 + E_t\ ;\ \theta$	9	847

Note: The best model overall for each season is shown in bold, except for SP (see explanation in text).

[a, b] Same as Table 2.2.

[c] SP = spring (30 April to 31 May), SU = summer (1 June to 31 July), FA = fall (1 August to 29 September).

Table 2.7: Final best models selected from Table 2.6 to describe spring, summer, and fall fire-day counts per township (≈10 000 ha) over the period 1995-2002. Models were fitted separately for each season. The sample dataset consisted of 598 townships located in northeastern Alberta.

Season	Model	Parameter estimates and included covariates
SP[a]	P-LVG$_{BM,SP}$	$-2.76 + 0.63\ RO_t - 2.51\ BU_t - 2.0\ DE_t + 0.32\ E_t$
	NB-LVG$_{BM,SP}$	$-2.56 - 2.59\ BU_t - 1.83\ DE_t + 0.33\ E_t$
		$\theta = 1.27$
SU[a]	NB-LVG$_{BM,SU}$	$0.28 + 0.67\ RO_t - 1.07\ BU_t - 2.15\ DE_t + 3.78\ WS_t$
		$+ 0.24 \log(F\text{-}L_{t,SU}) - 0.19\ E_t$
		$\theta = 5.91$
FA[a]	P-LVG$_{BM,FA}$	$1.43 - 1.25\ DE_t + 6.69\ WS_t + 3.37\ BS_t - 3.49\ BS_t^2$
		$- 0.30\ F\text{-}L_{t,FA} + 0.02\ F\text{-}L_{t,FA}^2 - 0.62\ E_t$

[a] SP = spring (30 April to 31 May), SU = summer (1 June to 31 July), FA = fall (1 August to 29 September).

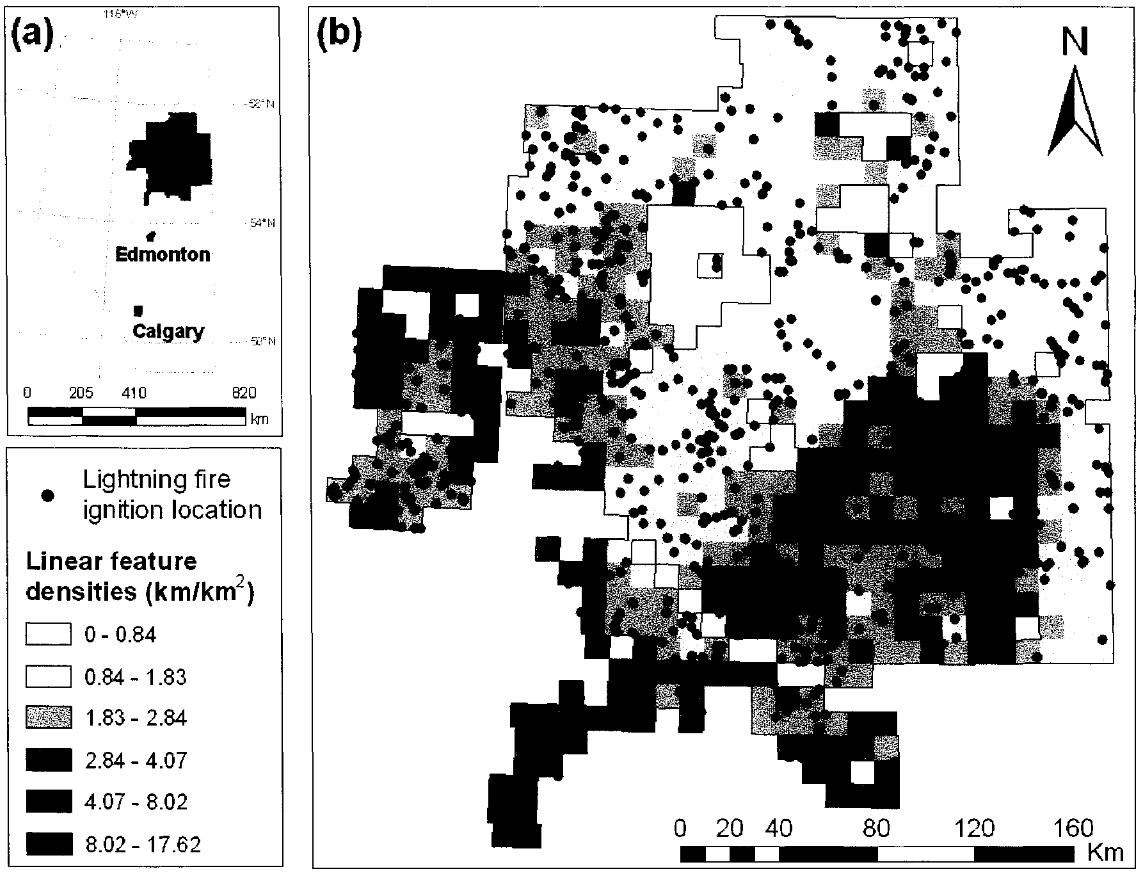

Figure 2.1: (a) The study area is a ≈67 000 km^2 region of boreal mixedwood forest located in northeastern Alberta, Canada. (b) The study area was divided using the provincial township grid, yielding 598 cells of 9 230 ha size on average. Linear feature densities, in km/km^2 of dry land, were calculated for each township and are shown in different shades of green. The red points in the map represent the ignition locations for the 919 lightning-caused fires that were used for analysis.

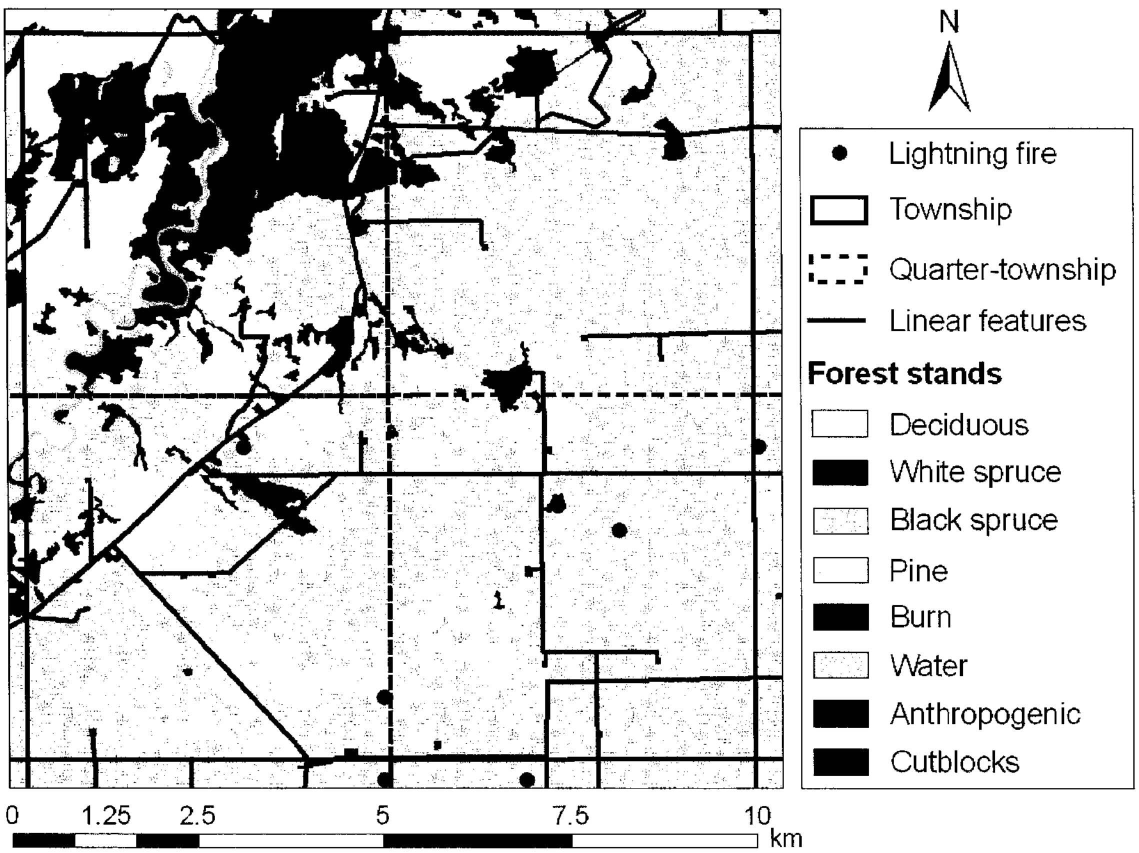

Figure 2.2: An example of variable collection for each township and quarter-township. Linear feature classes included roads, pipelines and seismic lines, and the densities of each class were calculated in km/km^2 of dry land. The proportional areas of each of six forest stand types (deciduous, white spruce, black spruce, pine, recent burns and cutblocks) were also calculated for each cell. Other landscape elements such as water bodies and human infrastructure (well pads, processing facilities) were not considered for analysis. The red points in the map represent lightning-caused fire ignition locations.

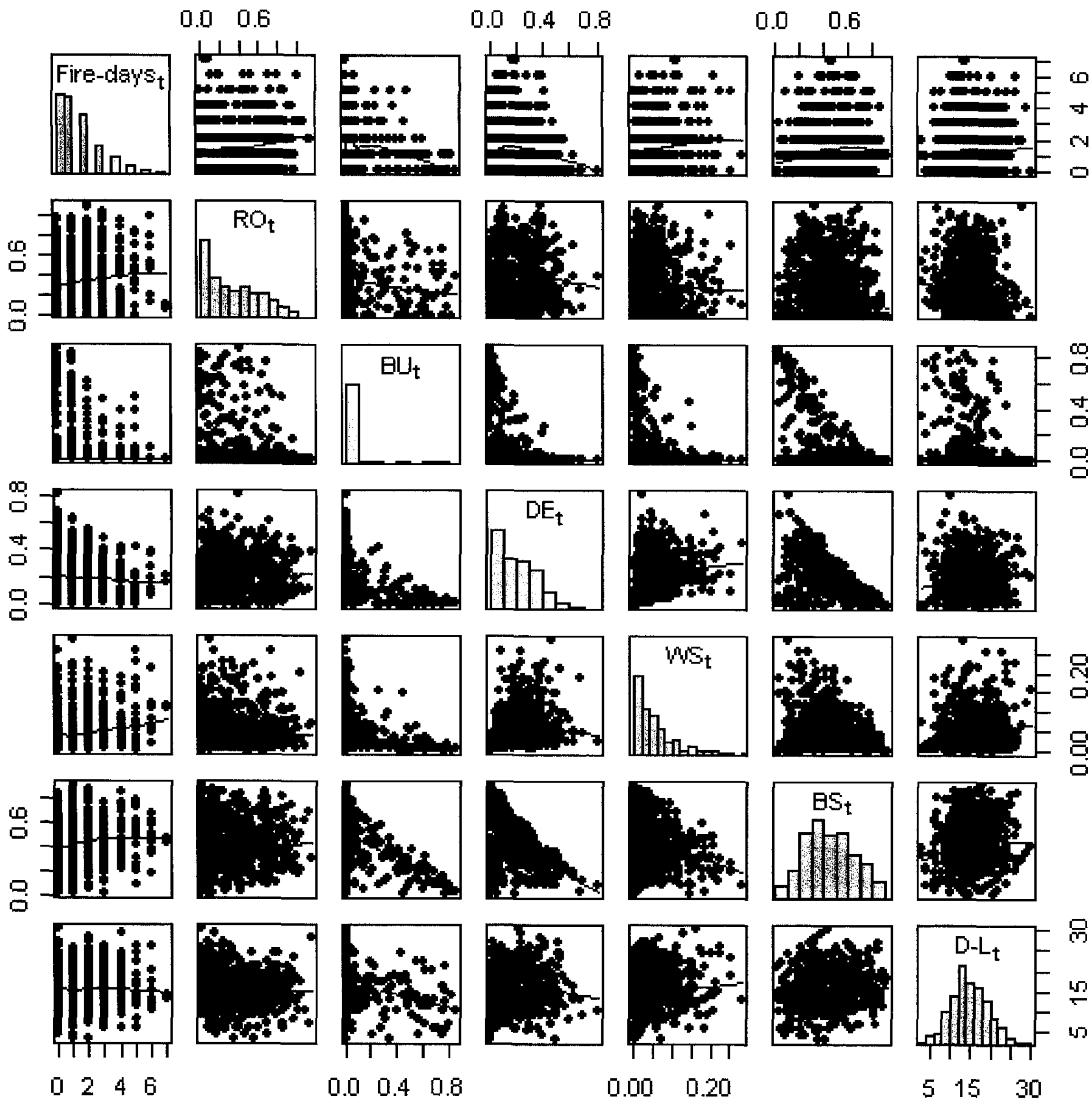

Figure 2.3: The inter-correlation and distribution of seven study variables collected at the township level (≈10 000 ha landscapes). The red line in the panels represents a locally weighted and smoothed line.

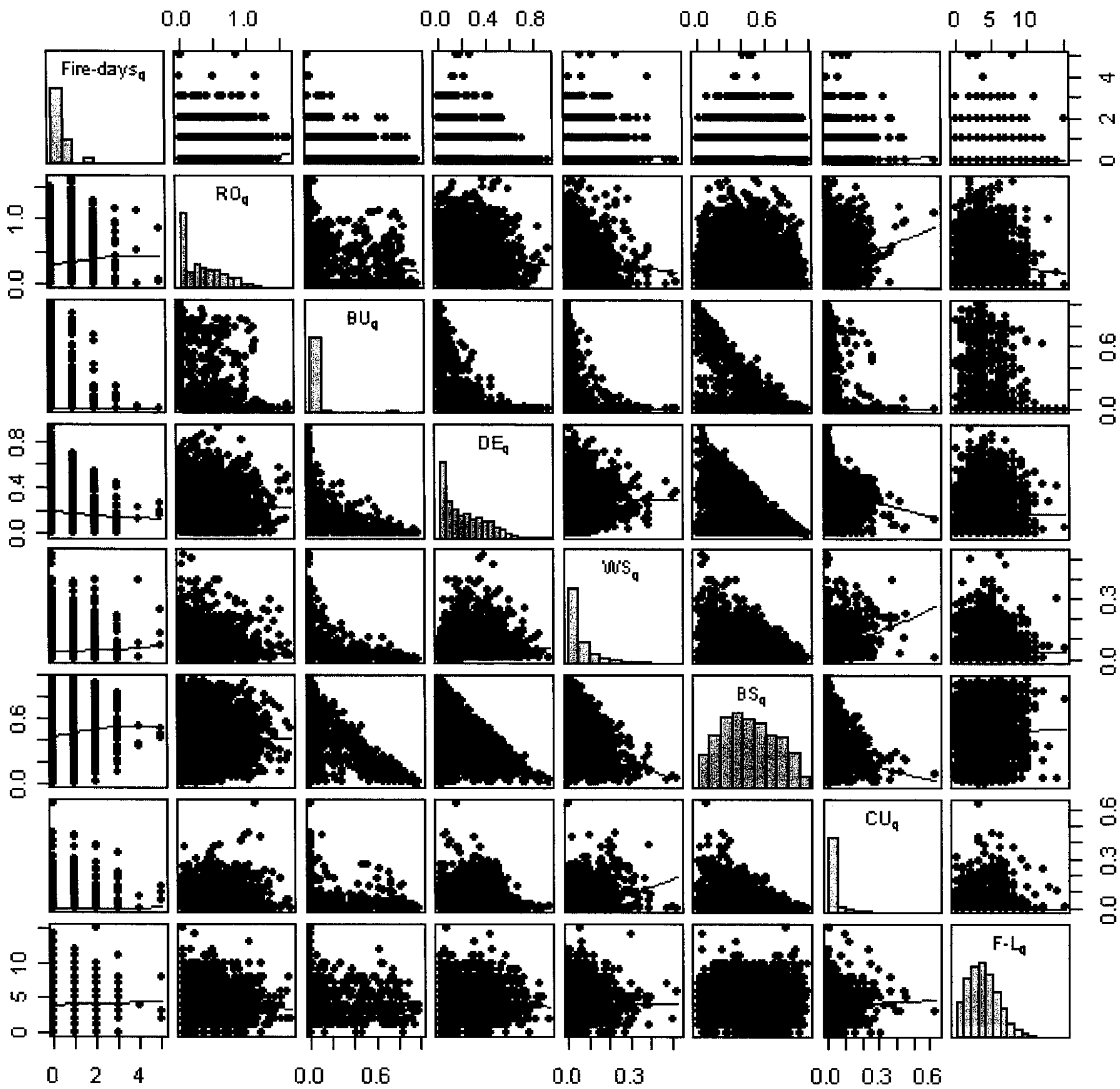

Figure 2.4: The inter-correlation and distribution of eight study variables collected at the quarter-township level (≈2 400 ha landscapes). The red line in the panels represents a locally weighted and smoothed line.

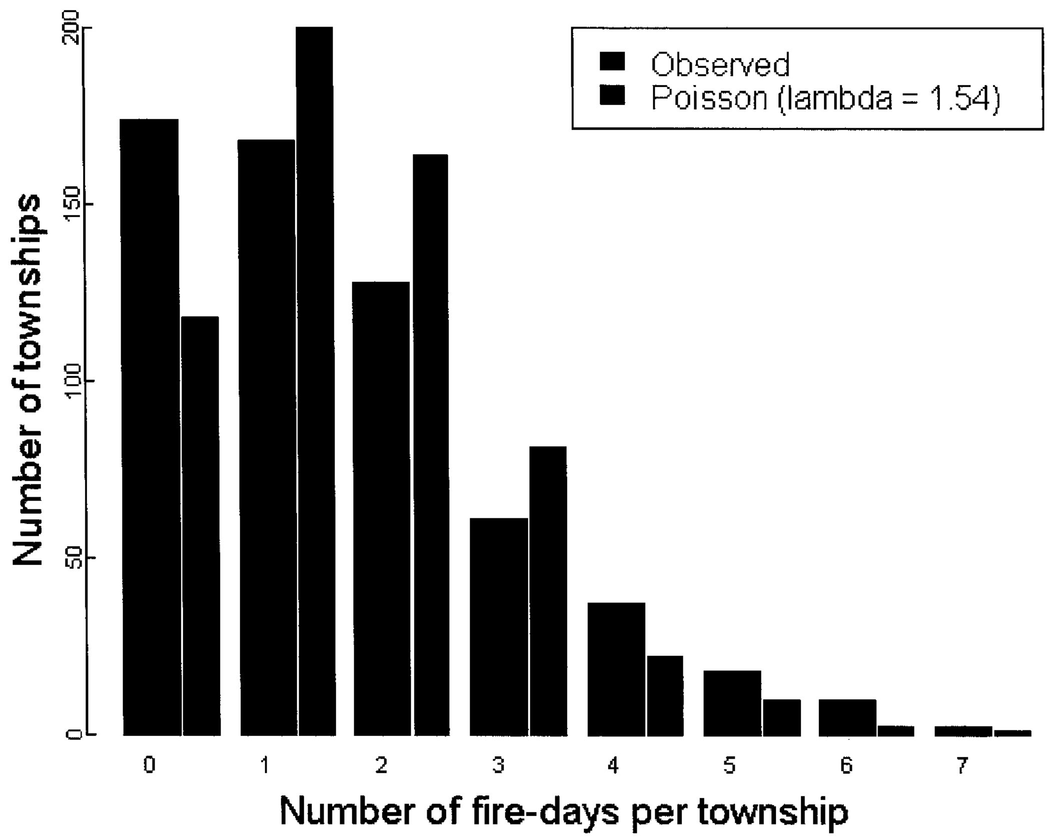

Figure 2.5: Histogram of the number of lightning-caused fire-days per township (≈10 000 ha cell) for all fires combined. The red bars show the expected number of fire-days under a Poisson distribution with parameter $\lambda = 1.54$. The sample consisted of 919 single fire-day events that occurred within 598 townships between 1995 and 2002.

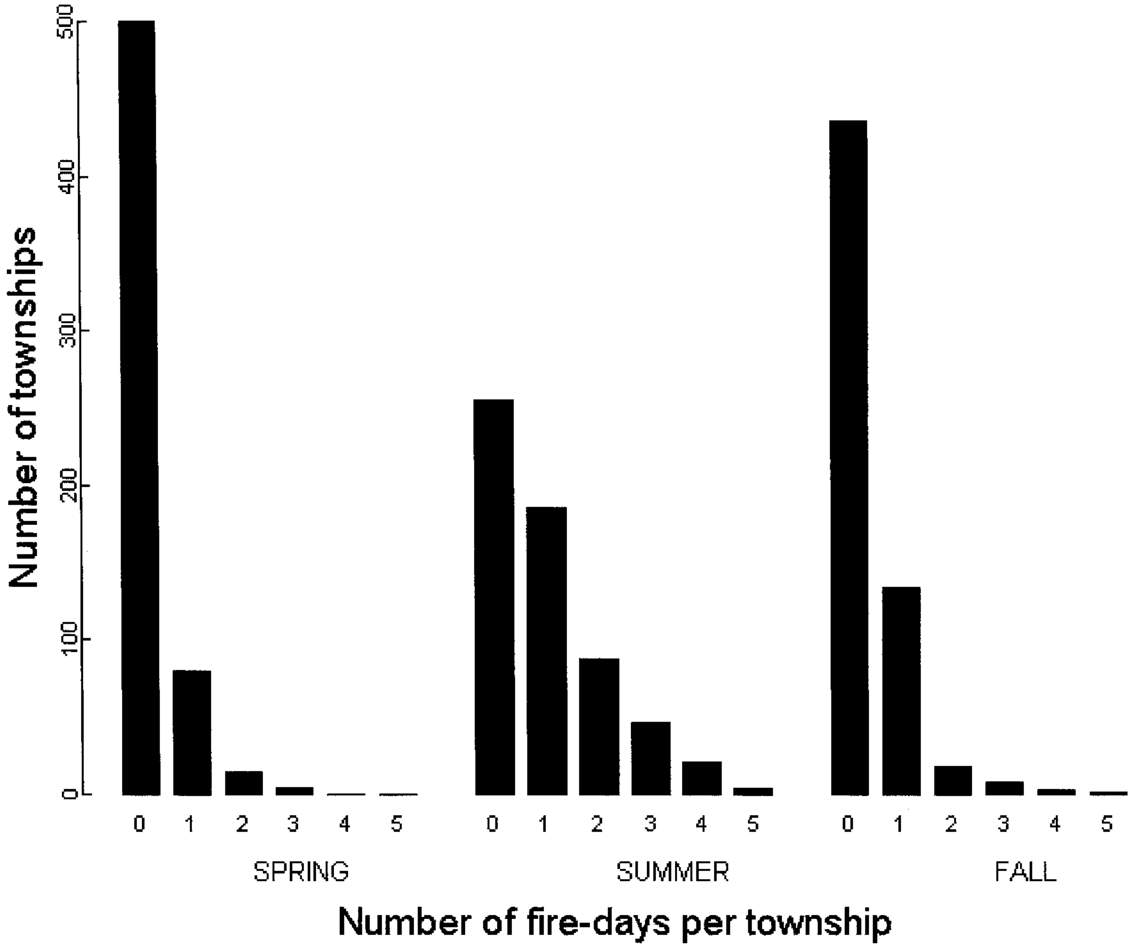

Figure 2.6: Histogram of the number of lightning-caused fire-days per township (≈10 000 ha cell), discriminating between fires that occurred during spring, summer or fall. The sample consisted of 117, 598 and 204 single fire-day events that occurred during spring, summer and fall, respectively, within 598 townships.

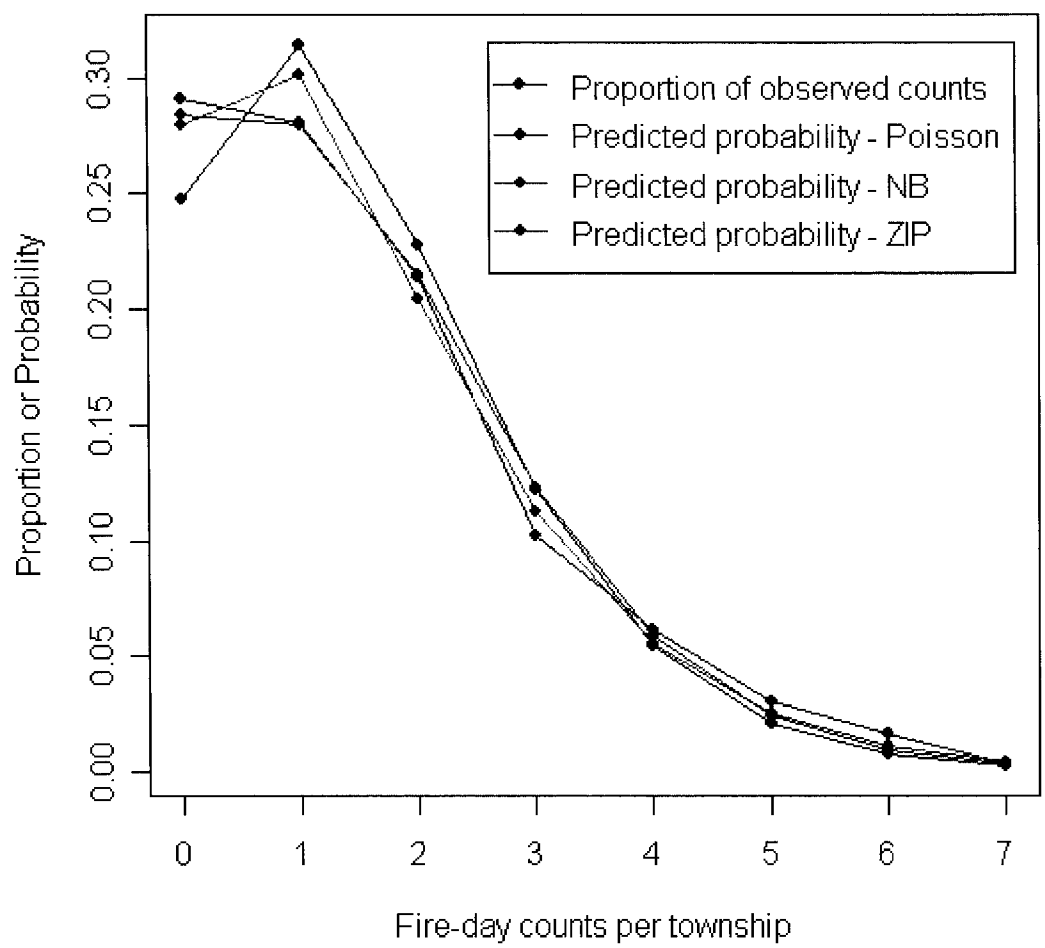

Figure 2.7: Observed distribution of lightning-caused fire-day counts per township (≈10 000 ha cell), compared to the predicted probability for each count category under a Poisson, Negative Binomial (NB) and zero-inflated Poisson (ZIP) regression model. Predicted probabilities were calculated using models P-LVG$_{BMt}$, NB-LVG$_{BMt}$ and ZIP-LVG$_{BMt}$, respectively (Table 2.2).

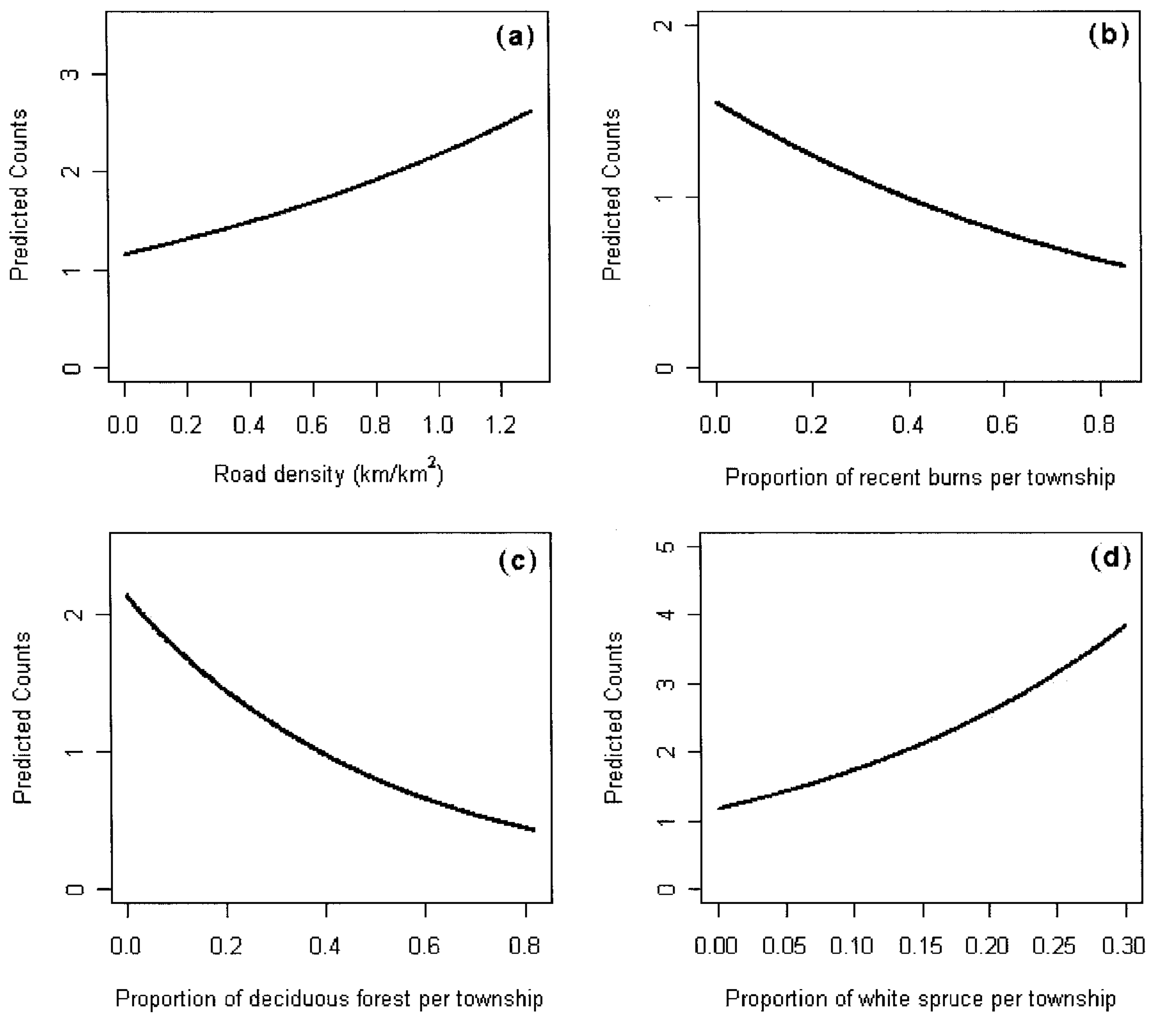

Figure 2.8: Predicted lightning-caused fire-day counts per township (black lines), together with 95% CI (grey envelopes), as a function of (a) road density (km/km^2 of dry land), (b) proportion of recent burns, (c) proportion of deciduous-dominated forest and (d) proportion of white spruce-dominated forest. Fitted frequencies were calculated using model NB-LVG$_{BMt}$ (Table 2.3), and the mean values for the remaining covariates. **Note:** the scale of the y-axis differs between (a), (b), (c) and (d).

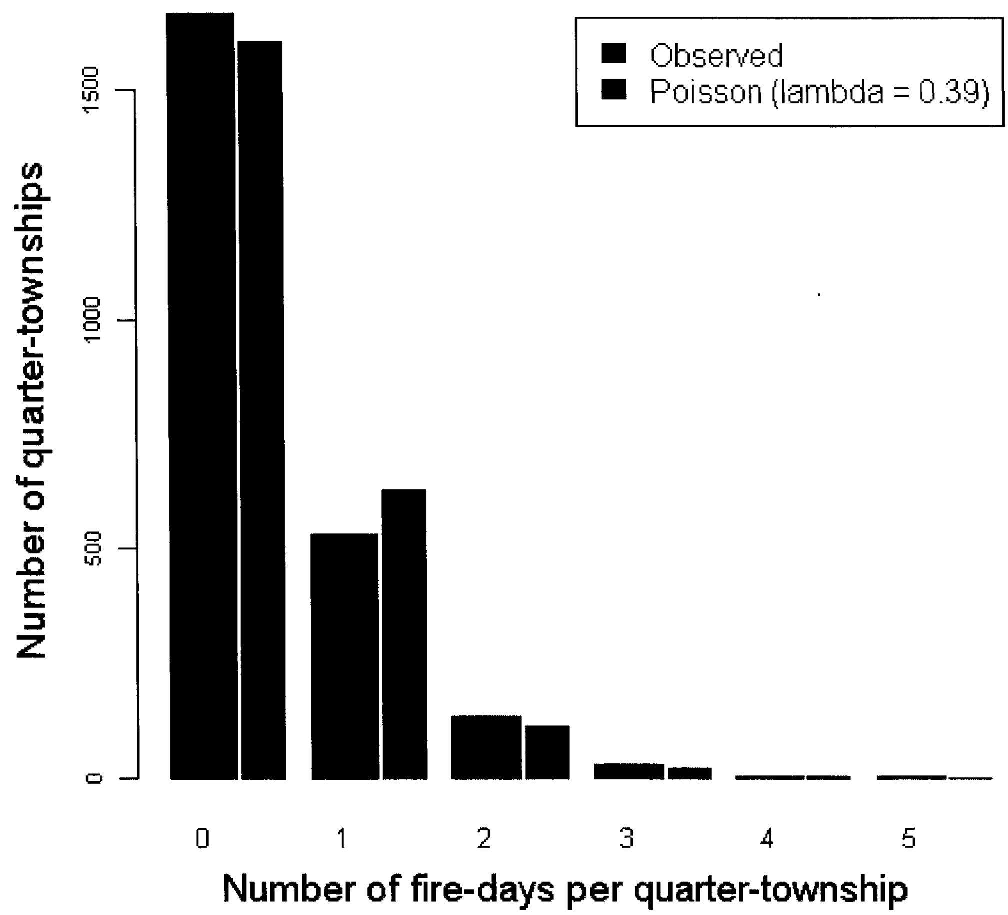

Figure 2.9: Histogram of the number of lightning-caused fire-days per quarter-township (≈2 400 ha cells) for all fires combined. The red bars show the expected number of fire-days under a Poisson distribution with parameter λ = 0.39. The sample consisted of 916 single fire-day events that occurred within 2 369 quarter-townships between 1995 and 2002.

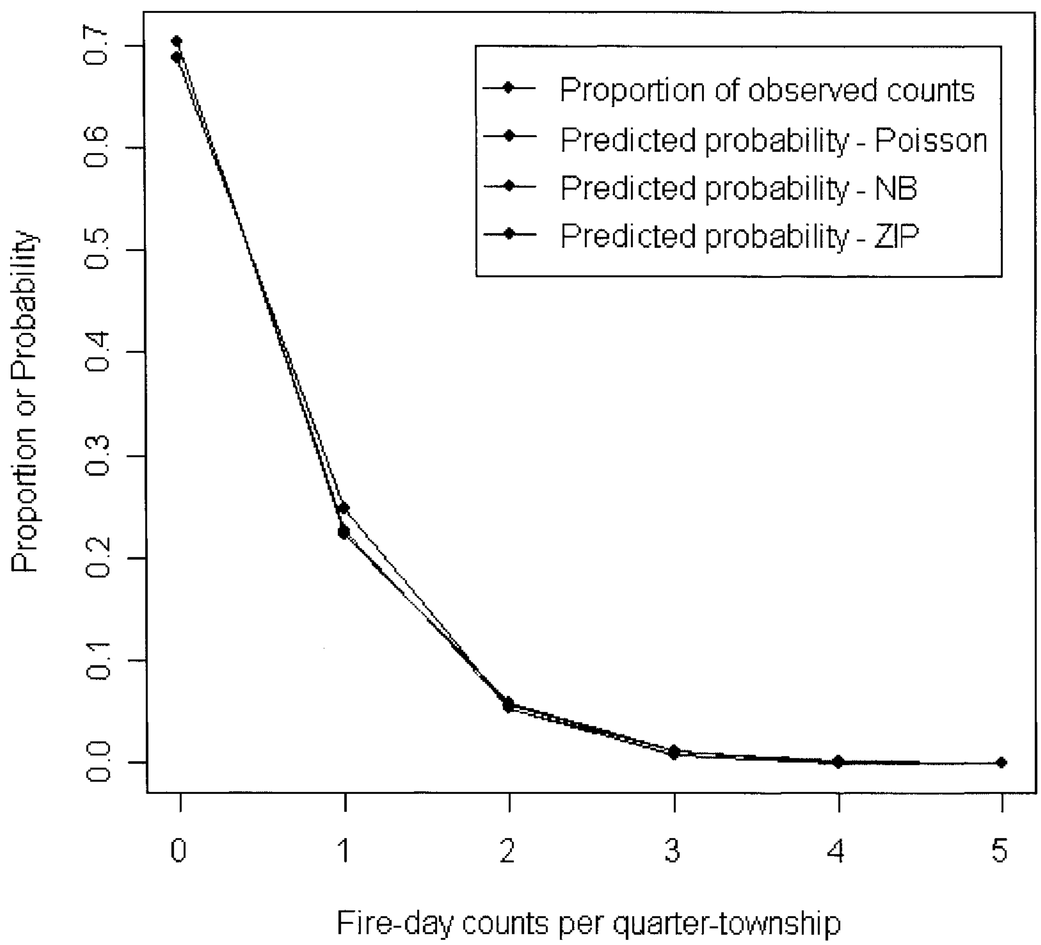

Figure 2.10: Observed distribution of lightning-caused fire-day counts per quarter-township (≈2 400 ha cells), compared to the predicted probability for each count category under a Poisson, Negative Binomial (NB) and zero-inflated Poisson (ZIP) regression model. Predicted probabilities were calculated using models P-LVG$_{BMq}$, NB-LVG$_{BMq}$ and ZIP-LVG$_{BMq}$, respectively (Table 2.4).

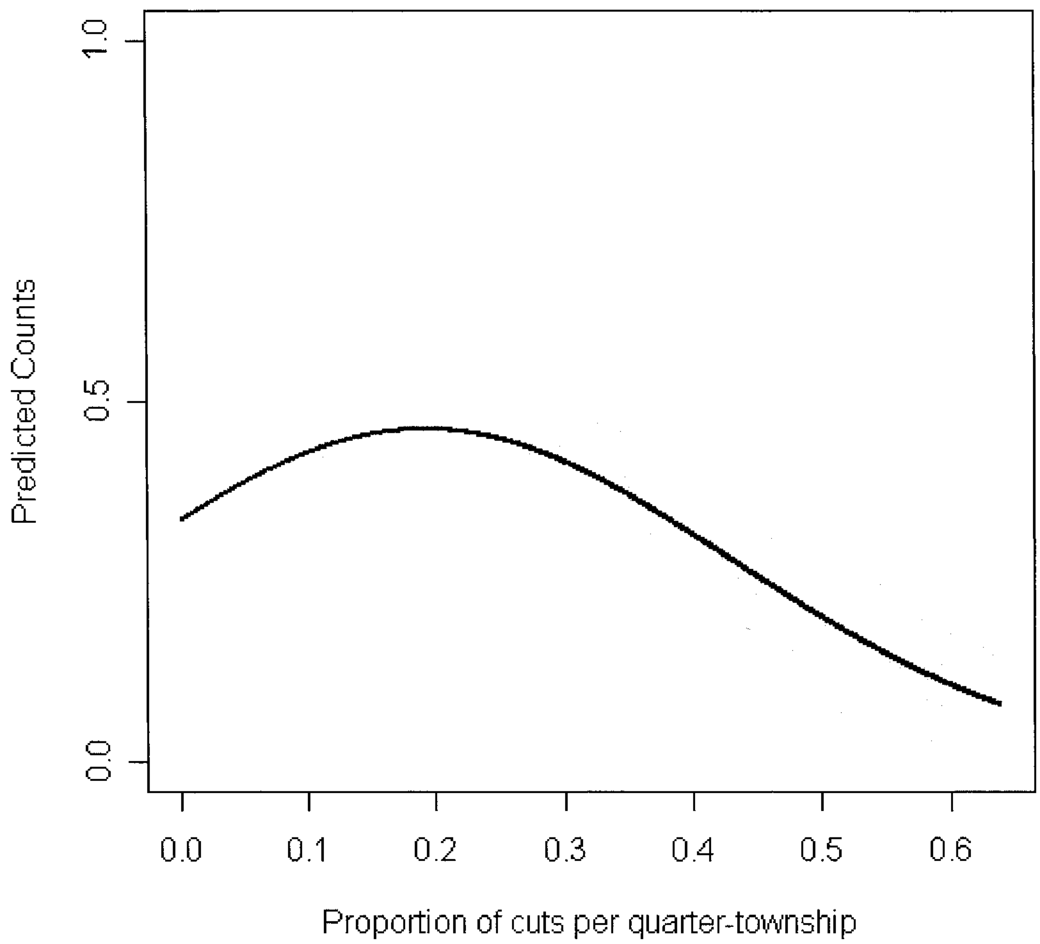

Figure 2.11: Predicted lightning-caused fire-day counts per quarter-township (black line), together with 95%CI (grey envelope), as a function of the proportion of cutblocks within the cell. Fitted frequencies were calculated using model NB-LVG$_{BMq}$ (Table 2.5), and the mean values for the remaining covariates in the model.

Chapter Three

Initial Attack Fire Escape Probabilities and Linear Features

3.1 Introduction

Fire suppression is widely practiced in those countries where wildfires are naturally occurring events. In Canada, suppression efforts are divided into three categories: preventative measures, initial attack, and the management of large fires (Martell 2001). For this work, I focused on initial attack (IA); 'the action taken to halt the spread or potential spread of a fire by the first fire fighting force to arrive at the fire' (Merrill and Alexander 1987). The ultimate aim of IA is to minimize area burned and subsequent suppression costs that result from uncontained large fires (Parks 1964). Its immediate objective, on the other hand, is to minimize the number of large fires, which I here consider are those whose final size is ≥ 3 ha (Cumming 2001a). This value represents a compromise between different target sizes set by fire management agencies across Canada: the observed proportion of fires larger than 2 or 4 ha are used as operational performance measures in Alberta (C. Tymstra, personal communication, 2005) and Ontario (R. Mcalpine, personal communication, 2005), respectively.

An IA operation succeeds if the fire is controlled below some small size, here 3 ha. Otherwise the fire is said to have escaped. There are two ways a fire can escape: (1) its size may already be larger than 3 ha when the fire fighters arrive, which I call a *response failure*, or (2) its size may be smaller than 3 ha, but the IA crew fail to contain it below the target size, which I call a *containment failure*.

IA operations in the Canadian boreal forest are conducted through the construction of a continuous fireline around the fire perimeter or through hot-spotting, a technique that concentrates efforts on the most intense points of the fire front. Crew productivity refers to the rate (in meters per hour) at which fire fighters construct fireline (Hirsch and Martell 1996), and probability of containment (POC) refers to the likelihood of establishing fireline sufficient to halt fire spread (Hirsch *et al.* 1998). Both of these variables are related to IA success probability. Crew productivity has been primarily related to fuel type (Murphy *et al.* 1989, Murphy *et al.* 1991, Barney *et al.* 1992, Hirsch *et al.* 2004), fire behaviour indices (intensity, rate of spread, etc.) and crew characteristics such as size, training and equipment (Quintilio *et al.* 1990, Hirsch and Martell 1996). Hirsch *et al.* (2004) found that crew productivity was inversely related to fire intensity, tree stem density and the volume of dead material on the ground within a stand. Using an expert judgement study, Hirsch *et al.* (1998) found that mean POC was inversely related to fire size at IA and intensity, although their model did not account for variation in fuel and weather conditions. POC has also been related to fire management strategy, i.e. to the decision process by which IA crews are deployed to temporally co-occurring fires (Cumming 2005).

In northeastern Alberta, Canada, the forestry and energy sectors have disturbed the forest by creating, among other structures, a growing network of linear features: roads, pipelines and seismic lines (Schneider *et al.* 2003). Linear features have been hypothesized to have both positive and negative effects on fire behaviour and suppression effectiveness. For example, existing linear features in the vicinity of a fire may facilitate fireline construction (Kiil 1970). Furthermore, linear features disrupt horizontal fuel continuity, which may impede fire spread (Miller and Urban 2000, Duncan and Schmalzer 2004). On the other hand, the herbaceous vegetation on some of these lines may increase fire spread rate, particularly during spring or fall when grasses are dry (Kiil

1970). Nevertheless, none of these effects have been unequivocally demonstrated.

The aim of this study is to describe the influence of biotic and anthropogenic factors on initial fire suppression, with special emphasis placed on assessing the effect of linear features. To do this, I used logistic regression to model the effect of fire cause, season, linear features, fuel types, fire weather, response time and fire size at initial attack on response and containment failure probabilities for fires that occurred over an eight year period (1995-2002) in a large region of boreal forest in northeastern Alberta (Fig. 3.1a). My sample consisted of 1 196 forest fires (Fig. 3.1b), of which 123 were response failures, 82 resulted in containment failures and 991 were successfully suppressed below 3 ha. This study adds to previous knowledge on fire suppression by incorporating factors previously considered (fuel types, fire size at IA) along with other influential variables (fire cause, weather, response time) and new landscape modifiers (linear features).

3.2 Data and methods

3.2.1 Study region

My study region was a large (≈67 000 km^2) forested area in the boreal mixedwood forest (Rowe 1972) of northeastern Alberta, Canada (Fig. 3.1a). The area is characterized by a continental climate (Strong and Leggat 1992) and a mild topography. Mature mixed stands containing either trembling aspen (*Populus tremuloides*) or balsam poplar (*Populus balsamifera*) and white spruce (*Picea glauca*) are characteristic of well drained sites (Strong and Leggat 1992) and cover approximately 25% of the area (Table 3.1). Rapidly draining sites are vegetated principally by jack pine (*Pinus banksiana*), and to a lesser extent by white spruce and aspen. Poorly drained areas are dominated by black spruce (*Picea mariana*) and tamarack (*Larix laricina*). Very poorly drained areas have sparsely treed or open bogs and fens, covering about 50% of the study region (Table 3.1).

In addition to forestry, oil and gas exploration and extraction activities are conducted in the area. The study region was relatively undisturbed by human activity until the 1950s, when forest harvesting and fossil fuel exploration and extraction intensified, resulting in changes in vegetation structure and increased linear feature density (Schneider *et al.* 2003). About 3% of the area has been directly altered by surface developments such as roads, pipelines, seismic lines and permanent clearings (Fig. 3.1b). Furthermore, Schneider *et al.* (2003) estimated that an average of 75 km/yr of permanent roads and 500 km/yr of temporary roads will be built in the area over the next 50 years and, on average, 3 km of seismic lines and 0.1 km of pipelines will be generated for each new conventional oil or natural gas well drilled.

Seismic lines are narrow corridors (≤ 8 m) constructed for the detection of oil and gas reserves and are revegetated using mixes of general agronomic species or allowed to revegetate naturally (Revel *et al.* 1984). Pipelines connect wells with intermediate processing facilities; they are generally buried but their surface is prevented from regenerating to forest in order to facilitate maintenance. Roads include non-permanent industrial roads and all-weather provincial highways, and both have grassy clearings on each side.

3.2.2 Data

The Government of Alberta's Forest Protection Division has on-line provincial-wide databases of fire records from 1961-2003 (ASRD 2004). Despite this large available dataset, various confounding factors and the limited availability of covariates had to be considered when deciding on a sampling time interval: the total annual area burned has varied by four orders of magnitude in the last 40 years (Armstrong 1999), fire suppression policy and strategy have changed (Cumming 2005), detection effort has not been uniform (Murphy 1985), and industrial activity has increased considerably (Schneider 2002). Time series spatial linear feature data do not exist for any part of Alberta, and complete present day maps, which lack most construction dates, are available only for limited areas, notably my study region. I chose 1995 as the lower time limit as a compromise between unmeasurable rates of landscape change resulting from industrial activities in the area and a sample size large enough to allow me to test the desired hypothesis. I set 2002 as the upper time limit because fire weather data for later years could not be generated within the scope of this study.

A fire arrival is a detected and reported fire (Cunningham and Martell 1976). Operationally, an arrival is said to have escaped IA if its final size exceeds 3 ha. As noted, escapes can result from two distinct processes, which I analysed separately. For the first analysis, p_i represents the probability of a response failure, which I modeled using a binary variable describing whether fire size at initial attack was < or ≥ 3 ha. For the second analysis, q_i represents the probability of a containment failure. This analysis used only those fires whose size at IA was strictly smaller than 3 ha. I constructed a binary variable classifying these fires as containment failures if their final size was ≥ 3 ha.

I modelled p_i and q_i as functions of eight data components: year, cause, season, response time, size at IA (for q_i only), linear feature densities, forest cover and fire weather (Tables 3.2 and 3.3). I computed the first five components from information contained in the provincial fire database, while the remaining three were calculated from various available GIS layers using Arc Map 9 (ESRI 2004).

I included fire year (YE) as a factor because annual arrival frequency varies in relation to annual changes in fire weather conditions (Weir *et al.* 2000, Krawchuk *et al.* 2006). Human and lightning-caused fires differ in phenology and spatial distribution (Stocks *et al.* 2002), thus I also included a categorical variable for fire cause (CA). Response time (RT), the time in minutes from a fire being reported to the start of IA, was included because I expected that the probability of occurrence of a response or a containment failure would increase for longer response times.

Fire behaviour varies seasonally as a result of changes in temperature, precipitation and fuel condition (Martell *et al.* 1989). Consequently, I expected that there would be seasonal differences in IA escape probabilities. Season (SE) was computed from the date of detection of a fire as a three-level factor for spring (30 April to 31 May), summer (1 June to 31 July) or fall (1 August to 29 September) fires.

To model containment failure probability, I included fire size at initial attack (SIA < 3 ha). I included this variable through a logit-like transformation $SIA_2 = \log [SIA/(3-SIA)]$ because I believe that q_i should approach 1 as fire size approaches 3 ha, while it must approach zero when the size of the fire is still small. The transformed SIA_2 ranges from $(-\infty, +\infty)$ on the scale of the linear predictor, instead of the (0, 3) range to which SIA is constrained.

If any of the hypotheses on linear features are true, the density of linear features within the vicinity of a fire should be related to its growth prior to and during an IA operation. I considered three classes of linear features: pipelines, roads and seismic lines. Linear feature coverages from the study area, considered valid and complete as of no later than 2003, were intersected with 3 and 12 ha circular buffers centered on the reported fire locations (Fig. 3.2). The total length of each linear feature class within each of the two buffers was calculated; these lengths were later converted to densities, in km/km^2, of pipelines (PL_3, PL_{12}), roads (RO_3, RO_{12}) and seismic lines (SL_3, SL_{12}). I used two different buffer sizes in order to control for potential spatial errors in the reported fire locations.

Some forest types in the boreal forest have been associated with higher fire incidence (Cumming 2001b, Krawchuk *et al.* 2006), faster rate of spread (Hély *et al.* 2001) and difficulty of containment (Hirsch *et al.* 2004). Thus, I expected that spatial variation in forest fuels would affect both modelled probabilities. Using digital forest inventory data (Alberta Vegetation Inventory (AVI) (Nesby 1997)), I calculated the proportional area of six forest cover classes (Table 3.1) within a 3 ha buffer centered at the fire ignition location (Fig. 3.2). Non-forested classes, including water bodies, agricultural land and permanent human infrastructure, accounted for 14.8% of the study region and were omitted from analyses.

Daily fire weather conditions are good predictors of ignition probability (Martell *et al.* 1989, Flannigan and Wotton 1991, Krawchuk *et al.* 2006), fire spread rate and intensity (Hély *et al.* 2001). I used two components of the Canadian Forest Fire Weather Index System (Van Wagner 1987) to measure daily fire weather conditions. Initial Spread Index (ISI) combines wind speed, and litter and fine fuel moisture content to represent the initial spread rate, independent of the amount of fuel burned (Van Wagner 1987). Fire Weather Index (FWI) combines wind speed and moisture content of litter, cured fine fuels and organic matter in the soil to represent the intensity of the spreading fire as energy emitted per unit length of fire front (Van Wagner 1987). Both these indices quantify the potential for fire spread and intensity. Daily weather indices for the study region and time interval were interpolated to the township grid (10×10 km^2 cells) by M. Krawchuk (see Krawchuk *et al.* 2006, for details on methodology).

Recorded fires with inconsistencies between size at IA and final size, vegetation buffers with incomplete data due to the limited spatial extent of the available AVI coverage, and those that had more than 30% water within the 3 ha buffer were not considered for analyses. Of the total 1 696 arrivals that occurred within the study region and time interval, 500 did not fulfil these selection criteria and were dropped from analysis. Of the remaining 1 196 fire arrivals (Fig. 3.1b), 205 (17%) escaped IA (Table 3.4). Of these, 123 (60%) were response failures and 82 (40%) were containment failures.

3.2.3 Model formulation

I adopted a multiple working hypothesis approach, where a number of alternate *a priori* models were compared (Burnham and Anderson 1998), as opposed to stepwise variable selection procedures. Two main issues had to be decided upon: the number of models to be contrasted and the maximum number of parameters to be included in any one model. Burnham and Anderson (1998) advocate carefully developing a set of four to 20 alternative models; I formulated 12 for each analysis. To guard against overfitting, Harrel (2001) recommends that the number of model parameters do not exceed *m/10*, where *m* is the 'limiting sample size'; in my case, this constrained the response and containment failure models to have no more than 12 or 8 parameters, respectively. The global models, containing all covariates, were allowed to exceed this limit as they were constructed only for purposes of comparison, rather than for inference or prediction (Burnham and Anderson 1998).

I formulated 12 competing models to assess the effects of the covariates on response failure probability (p_i) (Table 3.5). The global model (G_L) included all the covariates. The weather model (W_E), which included ISI and FWI, assessed the independent influence of fire weather on p_i. The vegetation model (V_{EG}) included the proportions of six forest cover classes and was built to assess the independent influence of fuel type. The influence of linear features was tested by three alternate models: LF_1 included the presence or absence of each linear feature class within the 3 ha buffer, LF_2 included the densities of each linear feature class within the 3 ha buffer, and LF_3 included densities within the 12 ha buffer. Model B_SP_I was designed to test for the effects of black spruce and jack pine fuel types, two of the most flammable conifers found in the boreal mixedwood forest (Cumming 2001b), combined with four other variables related to fire behaviour and management: cause, season, ISI and response time. These four variables were also included in each of the remaining five models. Model LF_{SE} tested the hypothesis that linear features totally vegetated with grasses (pipelines and many seismic lines) have a season-dependent effect on p_i. Model LF_{WE} tested whether linear features with grasses interact with daily weather conditions (ISI) by enhancing fire spread and consequently increasing p_i. Model CO_{WE} combined the effects of black spruce, pine, season and ISI. Model RT_{WE} tested if p_i is significantly influenced by RT and its interaction with ISI. Finally, model W_{SP} was constructed to test for an effect of white spruce, an ignition-prone stand type (Krawchuk *et al.* 2006) on which I expected suppression actions would be prioritized, given its high economic value.

A second suite of 12 models assessed the effects of covariates on containment failure probability (q_i) (Table 3.6). The logic behind the construction of the first six models was as above. Model H_{98} represents the relation between containment probability and size at IA (SIA_2) and ISI described by Hirsh *et al.* (1998). Model H_{04} represents findings of Hirsch *et al.* (2004), and so includes covariates for mixedwood forest, black spruce and pine forest types and fire intensity (ISI). Model B_SP_I was designed to test for the effects of black spruce and jack pine, combined with ISI and SIA_2. These last two variables were also included in the remaining three models. Model LF_{SE} tested the hypothesis that linear features with a grass cover (pipelines and many seismic lines) have a seasonal effect on q_i. Model B_SS_{IA} tested the hypothesis that q_i is influenced by the proportion of black spruce in the fire vicinity and fire size at IA. Finally, model LF_{WE} included an interaction term between grassy linear features and ISI to test the hypothesis that these features enhance fire spread under high wind conditions (a main component of ISI; see ISI definition in section 3.2.2); making containment more difficult, and consequently increase q_i.

3.2.4 Statistical analysis

I used logistic regression (Collet 1991) to model response and containment failure probabilities as a function of the covariates. In this type of regression, the response (r_i) is related to the vector of covariates (x_i) and parameters (β) through a linear predictor ($\eta_i = x_i^T \beta$) and a link function ($g(r_i) = \eta_i$). The logit link ($g(r_i) = log(r_i/(1-r_i))$) is recommended for retrospective studies of binomial data (McCullagh and Nelder 1989). I implemented logistic regression within a generalized linear model framework using R (R Development Core Team 2005).

I verified the assumption of linearity between the log-odds of the response and each candidate covariate using smoothed scatterplots and Generalized Additive Models (GAMs) (Hosmer and Lemeshow 2000). These relationships were linear with the exception of FWI, ISI, RT and four of the linear feature covariates. Log transformations of fire weather indices and response time induced linearity (0.5 was added to each covariate to accommodate zeros). I added quadratic terms to account for the non-linear relationship between the log-odds of a response failure and road (RO_3, RO_{12}) and pipeline (PL_3, PL_{12}) densities. Finally, quadratic terms in pipeline densities (PL_3, PL_{12}) induced a linear relationship with the log-odds of a containment failure. I assessed pair-wise collinearity between covariates using Spearman correlation coefficient. ISI and FWI were highly correlated ($r_s = 0.91$), so only ISI was retained. The remaining coefficients were $\leq |0.5|$, indicating multi-collinearity in terms should not affect the analyses (Bonate 1999).

I compared the candidate models using Akaike's Information Criteria (AIC) to determine which one composed the most parsimonious model while at the same time explaining the greatest amount of variation in the data (Burnham and Anderson 1998). I also computed Akaike weights (w_i), which are calculated as a ratio of likelihoods multiplied by prior probabilities and are interpreted as the weight of evidence in favour of model *i* relative to the rest of the models in the set; the smaller the w_i, the less plausible model *i* is the actual best model in the candidate set (Burnham and Anderson 1998).

Model predictive accuracy was assessed using measures that quantify the overall accuracy of predictions, such as the Nagelkerke R^2 index and the le Cessie - van Houwelingen - Copas - Hosmer unweighted sum of squares test for global goodness of fit (Harrell 2001). Model discrimination was assessed using the area under the Receiver Operating Characteristic Curve (AUC) (Hosmer and Lemeshow 2000), which quantifies the ability of a model to accurately classify sample units into those that experience the outcome of interest versus those that do not. AUC values between 0.7 and 0.8 are considered acceptable discrimination; values > 0.8 are considered excellent discrimination and indicate the model has high predictive power.

I performed internal validation by bootstrapping in order to assess how well the selected best models would perform when predicting outcomes on future samples (Harrell 2001). I generated 200 random samples with replacement from the original data, calculated new model parameters for each bootstrap sample and applied this 'newly derived' model to the original sample. The difference in the indices of predictive accuracy (R^2 and AUC) between the bootstrap and the original sample provides an estimate of the bias or 'optimism' due to the use of the development dataset at the validation stage (Harrel 2001). This approach provided me with bias-corrected R^2 and AUC estimates, which I compared to the original values in order to assess the reliability of the models in terms of future predictive performance.

I examined the signs of the estimated parameters in the best models to verify agreement with expectations based on common subject-matter knowledge. I checked graphs of partial residuals against each explanatory variable included in the final models to assess if the linear component was appropriate or whether particular covariates needed to be (further) transformed (Collet 1991). Finally, I checked for outliers and influential observations in the selected best models using graphs of standardized deviance residuals, likelihood residuals and leverage values (Collet 1991). I assessed the magnitude of the effect of each covariate on the response variable by computing odds ratios ($exp(\beta_i)$). This value represents the proportional change in the probability of occurrence of a response or a containment failure per unit change in the independent variable (Hosmer and Lemeshow 2000). For continuous variables, odds ratios are calculated for a change in *c* units in the value of the covariate, with *c* being a meaningful and realistic change.

Supplementary analysis included t-tests to assess differences in mean RT and ISI between response and containment failures, and a χ^2-test to assess seasonal differences in their frequency; I used a Kolmogorov-Smirnov test to compare final fire size distribution of the two types of failures. Additionally, I used the selected best model from Table 3.6 to estimate the number of recorded response failures that would have resulted in containment failures if they had been first actioned when their size was 0.5, 1 or 2 ha.

3.3 Results

The total area burned by the 1 196 sample fires was 4 219 km^2, of which 99.9% resulted from escaped fires. Ten percent of all fires were response failures while 7% of all fires were containment failures. Response failures represented 60% of escaped fires and were responsible for 85% of the area burned, while containment failures constituted the remaining 40% of escaped fires and accounted for 15% of the area burned (Table 3.4).

Mean response time was longer for response ($\bar{x} = 252.9\,\text{min}$) than for containment failures ($\bar{x} = 128.5\,\text{min}$) (t = 2.14, d.f. = 187.3, p = 0.03), but mean ISI did not differ (t = -0.28, d.f. = 174.7, p = 0.77). The frequencies of response and containment failures did not differ between seasons ($\chi^2 = 0.86$, d.f. = 2, p = 0.65). The distribution of final sizes differed between response ($\bar{x} = 2905.6\,ha$) and containment failures ($\bar{x} = 782.2\,ha$) (D = 0.32, p = 0.00009).

Densities of the three classes of linear features varied markedly between fire locations (Table 3.3). Seismic lines were the most abundant class, with some buffers having densities as high as 55.54 km/km^2 (Table 3.3). Black spruce was the most abundant forest class; the median proportional area of this class was 0.53, and at least some of it was present in 72% of the buffers. The proportional area of the different forest types ranged from 0 to 100% cover, while the median proportional areas were all approximately zero. I concluded from Table 3.3 that there was sufficient spatial contrast in forest composition and linear feature densities to support my analyses.

3.3.1 Response failure probability (p_i)

From the 1 196 arrivals used to model p_i, 123 were response failures ($\bar{p} = 0.103$). The 11-parameter model CO_{WE} had the lowest AIC value relative to the global model (Table 3.5), and the highest w_i. This model related p_i to fire cause, season, proportion of black spruce and pine, ISI, response time, and the interactions between season and black spruce, and pine abundance and ISI (Table 3.7). Signs of estimated coefficients for fuel types, ISI and RT were consistent with subject matter expectations (Table 3.7). Graphical assessment of model residuals indicated that there were no outliers or influential observations. The goodness of fit test indicated that model CO_{WE} was a well-fit model (p =

0.65). The R^2 was 0.11, while the bias-corrected R^2 was 0.09, indicating that there was a small amount of overfitting in the original model. The AUC score was 0.72 (Table 3.5), indicating acceptable classification power; while the bias corrected AUC was 0.70.

Other factors being equal, the odds of a response failure for lightning fires were more than twice that of human-caused fires (OR = 0.46; 95%CI: 0.22, 0.98). Increasing RT from 15 (lower quartile) to 65 (higher quartile) minutes increased the odds of a response failure by a factor of 1.25 (95%CI: 1.05, 1.46). Response failure probability was greater overall during spring (Fig. 3.3). Black spruce abundance significantly affected p_i during summer, when the odds of a response failure was 3.37 (95%CI: 1.64, 6.91) times higher for fires with 100% black spruce cover compared to fires with 0% cover (Fig. 3.3). However, this effect was not observed in spring or fall. The proportion of pine in the vicinity of a fire affected response failure probability, though this effect was conditional on fire weather conditions (Fig. 3.4). Under mild weather conditions (ISI = 10), p_i increased steadily with the abundance of pine. Under severe fire weather conditions (ISI ≥ 15), on the other hand, p_i rapidly approached 1, even for fires that had less than 50% of pine within their immediate vicinity (Fig. 3.4).

Response failure probability was not related to the proportions of deciduous, white spruce, burned or cut forest cover classes within the fire vicinity. There was no evidence for an effect of linear feature densities, even after accounting for the possibility of seasonally-dependent responses or interactions with fire weather conditions (models LF_{SE} and LF_{WE}, Table 3.5).

3.3.2 Containment failure probability (q_i)

To model q_i, I left aside 123 fires with size at IA ≥ 3 ha. From the remaining 1 073 arrivals, 82 were containment failures ($\overline{q} = 0.076$). The selected best model was the two-parameter model H_{98} (Table 3.6), which related q_i to ISI and SIA_2 through the equation

$$logit(q_i) = -2.66 + 0.88 \times \ln(ISI) + 0.73 \times SIA_2$$

The 95% confidence intervals for the coefficients of ln(ISI) and SIA_2 were (0.36, 1.40) and (0.57, 0.88), respectively. Both coefficients were significant ($p < 0.01$) and their signs were positive, as expected. There were no outliers or influential observations. The R^2 for model H_{98} was 0.31, and the corresponding bias-corrected R^2 was 0.30, indicating there was no overfitting in the original model. The model's AUC score was 0.85 (Table 3.6), indicating excellent classification power. Furthermore, bootstrap validation showed no bias in the original AUC estimate. I found no significant interaction between ISI and SIA_2 (alternate model not shown). Comparison of AIC and w_i showed little or no support for the alternate models relating containment failure probability to local vegetation, year, cause, season or response time. Finally, none of the linear features models showed any improvement with respect to the null model.

Under model H_{98}, the odds ratio in containment failure probabilities per five-unit increase in ISI was 2.18 (95%CI: 1.38, 3.45). Thus, q_i increased with high wind speeds and/or low fuel moisture, as indirectly measured by ISI. The size of a fire at the onset of IA had a significant positive effect on q_i. The odds ratio for a change in SIA from 1 to 1.5 ha was 1.66 (95%CI: 1.49, 1.84), while for a change from 2 to 2.5 ha was 1.95 (95%CI: 1.69, 2.25). The effect of increasing ISI and SIA on q_i can be observed in Figure 3.5. On the left side of the graph, mild weather conditions result in a low probability of a containment failure for small fires, but high q_i for fires that are already close to the 3 ha threshold. Under extreme weather conditions (ISI > 25), the probability of a containment failure is high, especially for fires that are already larger than 0.5 ha when IA starts.

Using model H_{98}, I estimated that if the 123 recorded response failures had been first actioned when they were 0.5, 1 or 2 ha, the number of fires resulting in containment failures would have been 15, 25 and 50, respectively. This would have reduced the total number of escaped fires (205) to 97, 107 and 132, respectively.

3.4 Discussion

A fire can escape initial suppression via two processes: its size may already be larger than the target size (here, 3 ha) when the IA crew arrives, which I call a response failure; or its size might be smaller, but the fire fighters fail to contain it below 3 ha, which I call a containment failure.

Response failure probability was correlated with fire cause, season, response time (the time elapsed between fire detection and first suppression actions), forest cover type within ≈100 m of the reported fire location, and the fire weather index ISI. The odds of a response failure for a lightning fire were more than twice that of human-caused fires. Lightning and human-caused fires differ in phenology and spatial distribution (Stocks *et al.* 2002), with the latter generally occurring in areas located close to human centers or road access (Vega Garcia *et al.* 1995, Cardille *et al.* 2001). Thus, they are more likely to be detected and actioned soon after ignition; indeed, I found that mean response time was significantly higher for lightning fires than for human-caused fires ($t = 3.19$, d.f. $= 186.8$, $p = 0.0016$). Furthermore, human-caused fires occur primarily in spring, and within non-coniferous fuel types. Both factors may contribute to the reduced probability of a response failure resulting from a human-caused fire, independent of the effect of response time.

Under sufficiently extreme conditions, limited fire suppression resources must be allocated amongst multiple co-occurring fires, such that some fires will be left unattended. Such fires will be able to grow freely, and will be more likely to exceed a given threshold size (e.g. 3 ha) before first action is attempted. In my study region, Cumming (2005) provided indirect evidence that high daily arrival counts significantly limit initial attack effectiveness. I incorporated a proxy for this effect by including response time as a covariate; as expected, RT and the probability of a response failure were positively correlated. However, the effect magnitude of this covariate was lower than might have been expected. When fire managers are deciding the order of priority in which fires should be battled, they base their decisions on information on fire growth rates, fire weather and fuel type maps. It is possible then that those fires that are expected to grow slower are left unattended for some time. If this was the case, a longer response time would not necessarily always result in a response failure. Information on the way this decision process is made could potentially lead to more informative models regarding the true magnitude of the effect of this covariate.

Other factors being equal, response failure probability was highest in spring, coincident with a period of minimal foliar moisture in boreal forests when respiration occurs while the rooting zone remains frozen. This phenomenon, referred to as 'spring dip', has been observed for various types of vegetation present in the boreal forest (C. Tymstra, personal communication, 2005). The probability of a response failure was positively correlated with local black spruce abundance only during summer, when the odds of a response failure were more than three times higher for fires with 100% black spruce cover compared to fires with 0% cover. Black spruce stands tend to contain high levels of flammable fuels, provided by basal branches and abundant downed dead material (Hély *et al.* 2001). These fuels would typically be driest, and most conducive to fire spread, during summer.

Response failure probability was significantly related to the abundance of pine-dominated forest cover through an interaction with ISI. The relationship was strongly positive under extreme weather conditions (ISI $\geq$ 15). Pine is a fire-adapted species that grows in rapidly draining soils and is capable of carrying very intense crown fires (Hély *et al.* 2001). However, in Alberta, jack pine is considered a fuel type where fire behaviour is low except under strong winds (C. Tymstra, personal communication, 2005). The forest floor in these stands tends to be quite clean, yet a fire will rapidly grow if it manages to ignite the canopy, which can easily occur under strong winds. My results are consistent with this phenomenon.

I detected no relation between p_i and the abundance of the other fuel types analyzed. In particular, aspen dominated stands (Van Wagner 1977, Hirsch *et al.* 2001) and recently disturbed areas (Cumming 2001a) have been described as fire breaks, limiting potential fire size. I did not detect such effect; probably because my sample did not provide a high enough contrast in the distribution of aspen and disturbed areas within the 3 ha buffers. Most of the analyzed fires completely lacked deciduous dominated forest within their vicinity, which is consistent with the findings by Krawchuk *et al.* (2006) of low fire arrival probabilities in aspen stands.

Linear features have been pointed as potential natural fire breaks, limiting fire spread (Miller and Urban 2000, Duncan and Schmalzer 2004); in contrast, it has also been suggested that herbaceous vegetation on some of these lines may increase fire spread rate (Kiil 1970). I found no clear evidence to support either hypothesis. Thus, even though linear features may have positive or negative effects on the behaviour of particular fires; I found no evidence for a net effect amongst my sample, even after including interactions accounting for possible differences in effect magnitude or direction between seasons and different weather conditions.

Containment failure probability was influenced by the size of the fire at initial attack (SIA) and the fire weather index ISI. As SIA increases, so does the fire perimeter, making it more difficult for fire fighters to establish an effective fireline and thus contain the fire. In addition, SIA may have also captured part of the effect of response time, with longer response times resulting in larger fire sizes at the onset of IA. I modelled the effect of this covariate using a logit-like transformation to represent the hypothesis of a sigmoidal relationship between q_i and SIA, i.e. $q_i \to 0$ as SIA $\to 0$ while, by definition, $q_i \to 1$ as SIA $\to 3$. Additionally, it is important to note that my results are sensitive to measurement errors in the recorded sizes at IA.

The influence of ISI on the probability of a containment failure can be related to fire intensity and the amount of water needed to extinguish flaming combustion. Laboratory studies have shown that more than three times as much water is needed to extinguish a fire that has an intensity of 1500 kW/m compared to a 500 kW/m fire (Stechishen 1970). Further, fire intensity increases the potential for short range spotting, which refers to jumping burning flames starting small fires neighbouring the main fire. If this occurs, the fire fighters have to divert their efforts from the main fire to attend these spot fires (Hirsch *et al.* 2004). Finally, it has been suggested that the heat generated by high intensity fires can influence the psychology and behaviour of the fire fighters, affecting their ability to efficiently fight a fire (Hirsch *et al.* 2004). I approximated fire intensity using a fire weather index (ISI) that quantifies the potential for fire spread, a main component of fire intensity (Merrill and Alexander 1987), given the wind conditions and the litter and fine fuel moisture contents of the day the fire occurred. Fire weather indices have been shown to be good predictors of fire intensity (Hély *et al.* 2001); however, direct measures of fire intensity, if available, could be easily incorporated in future models.

I did not find a significant effect of fuel type on q_i. Previous studies have shown that forest types affect the probability of containment, primarily by impeding fire fighters to walk through the terrain or by increasing the volume of water needed to be applied. Hirsch *et al.* (2004) identified pine and black spruce as fuel types which impose difficulties to fire fighting, given the large amounts of dead material and/or high stand densities. However, this effect was not detected. Additionally, it is worth mentioning that containment failure probability was not dependent on fire cause, suggesting that lightning and human-caused fires behave alike at the containment stage.

Containment failure probability was not related to the presence of linear features within 3 ha buffers or their densities within 3 and 12 ha buffers surrounding the recorded fire locations. This result suggests that at their current density, the use of existing linear features for fireline construction does not facilitate containment. Previous studies have found that road density is positively related to daily fire occurrence (Vega Garcia *et al.* 1995, Cardille *et al.* 2001), though ignition represents a different stage which could be influenced by a different set of variables than the escape stage analyzed here. It would be interesting to determine whether linear features have an effect on any other component of fire behaviour, given that their density and related disturbances are projected to increase rapidly throughout the boreal forest (Schneider *et al.* 2003).

Crew size and experience have been shown to influence IA success (Hirsch *et al.* 2004, Quintilio *et al.* 1990). The effect of IA crew size is greater in high density fuel types and/or high intensity fires, while fire fighter experience can influence the success or failure of an IA attempt (Hirsch *et al.* 2004). I was not able to assess these effects using the data available to this study; nevertheless, both factors could be incorporated into a similar analysis if the appropriate covariates were available.

Limited availability of spatial data, and the absence of any reliable time-series data on linear feature densities in Alberta, constrained the time interval for this study, and made external model validation unfeasible. I therefore conducted an internal validation by bootstrapping to ensure that obtained estimates of model performance were not overly optimistic. Results from this procedure indicated that the selected models of response and containment failure probabilities both performed reasonably well when fitted to random sub-samples of the original dataset. In particular, the bias-corrected R^2 and AUC score for the response failure model CO_{WE} were both somewhat smaller than the indices calculated on the original sample, indicating that there was a small amount of overfitting. The discrimination ability of this model was just above the conventional minimum for a suitable model, indicating that its predictive accuracy is not high. Model performance could likely be improved by incorporating additional covariates.

Nevertheless, I believe model CO_{WE} provides valuable insight into some of the mechanisms underlying the occurrence of response failures, which seem to be mostly driven by weather, fuels and fire management. Moreover, the analysis also revealed no support for the alternative hypotheses that this study was primarily designed to formulate and test, namely that linear feature densities had a consistent effect on response failure probability.

Containment failure model H_{98}, on the other hand, performed better. The consistency in and magnitude of the original and bias-corrected AUC estimates for model H_{98} indicate this model has good predictive ability, appropriate for management purposes in this region of northeastern Alberta. If the model was to be used for prediction outside of the area where it was developed, external validation would be advisable, in order to provide a more stringent assessment of the model's predictive accuracy (Harrell 2001). However, as I noted before, assembling the necessary data to perform such task would be challenging.

Response failures accounted for 85% of the area burned by large fires ($\geq$ 3 ha). Using model H_{98}, I estimated that if the 123 recorded response failures had been actioned when they were 0.5, 1 or 2 ha in size, only 15 (12%), 25 (20%) and 50 (41%), respectively, would have resulted in containment failures. This would presumably reduce the area burned; nevertheless, I make no claim as to what the magnitude of this reduction could have been. For one thing, the distribution of final sizes differs between response and containment failures, so it is unclear which mean final size to assume for such proposed estimation. However, I believe that some reduction in the number of response failures is feasible, particularly given that mean RT, mean ISI and seasonal frequency differed slightly or not at all between response and containment failures.

Minimizing the economic impacts of wildfires is an important goal of sustainable forest management in fire-driven ecosystems (Bergeron *et al.* 1998). To achieve this goal, quantitative information specifying the IA containment capability under differing fire environments and suppression situations is essential for several forest fire management decisions (Hirsch and Martell 1996). As described before, the models developed in this study can be incorporated into strategic fire management planning in the forests of northern Alberta. They also provide parameter estimates that could be incorporated into simulation models of fire and ecosystem dynamics, increasing our understanding of fire behaviour and the effects of fire management in the boreal forest.

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Table 3.1: Description and abundance of the forest classes used as covariates in logistic regression models. Classification of stands into each forest class was based on species composition and stand density.

Forest class	Forest classification	%[a]
Deciduous (DE)	Aspen or poplar dominated stands	20
White spruce (WS)	White spruce dominated stands	5
Black spruce (BS)	Black spruce or larch dominated stands, and some birch/black spruce mixes	43
Pine (PI)	Jack pine dominated stands, and deciduous stands where pine is the dominant conifer	7.9
Burned (BU)	Old and recent burns	6.7
Cut (CU)	Harvested blocks, partial cuts and identifiable salvage-logged areas.	2.6

[a] Percentage of each forest class calculated over the entire study region.

Table 3.2: Description of the explanatory variables used for analyses. Linear features densities were calculated within 3 and 12 ha buffers around the fire ignition point; proportion of six forest classes were calculated only within the 3 ha buffer.

Variable	Description and units
YE	Year the fire occurred.
CA	Cause of the fire; 0 = lightning, 1 = human.
SE	Season the fire occurred; SP = spring (30 April to 31 May), SU = summer (1 June to 31 July), FA = fall (1 August to 29 September).
PL	Presence or absence of pipelines within a 3 ha buffer.
RO	Presence or absence of roads within a 3 ha buffer.
SL	Presence or absence of seismic lines within a 3 ha buffer.
PL_3	Pipeline density within a 3 ha buffer, in km/km^2.
RO_3	Road density within a 3 ha buffer, in km/km^2.
SL_3	Seismic line density within a 3 ha buffer, in km/km^2.
PL_{12}	Pipeline density within a 12 ha buffer, in km/km^2.
RO_{12}	Road density within a 12 ha buffer, in km/km^2.
SL_{12}	Seismic line density within a 12 ha buffer, in km/km^2.
DE	Proportion of deciduous forest within a 3 ha buffer.
WS	Proportion of white spruce forest within a 3 ha buffer.
BS	Proportion of black spruce forest within a 3 ha buffer.
PI	Proportion of pine dominated forest within a 3 ha buffer.
BU	Proportion of recent burns within a 3 ha buffer.
CU	Proportion of cutblocks within a 3 ha buffer.
RT	Response time, in minutes.
FWI	Fire weather index.
ISI	Initial spread index.
SIA	Fire size at initial attack, in ha.

Table 3.3: Basic descriptive statistics for the continuous explanatory variables used in logistic regression models. Values were calculated from a sample of 1 196 fires that occurred in northeastern Alberta between 1995 and 2002.

Variable[a]	Mean	SD[b]	Median	Max[c]
PL_3	0.41	2.25	0	40.34
RO_3	0.44	1.62	0	12.86
SL_3	2.14	4.72	0	55.54
PL_{12}	0.47	1.82	0	22.58
RO_{12}	0.47	1.17	0	8.75
SL_{12}	2.13	4.30	0	55.48
DE	0.17	0.30	0	1
WS	0.06	0.18	0	1
BS	0.51	0.43	0.53	1
PI	0.06	0.19	0	1
BU	0.04	0.18	0	1
CU	0.04	0.16	0	1
RT	288	2950	29	89530
FWI	15.84	8.91	16	50
ISI	6.08	3.70	6	33
SIA	2.72	15.31	0.2	300

[a] Variable names are as described in Table 3.2.

[b] SD = standard deviation.

[c] Max = maximum value recorded.

Table 3.4: Number of and area burned by non-escaped fires, response and containment failures, and all fires from a sample of 1 196 fires that occurred between 1995 and 2002 in northeastern Alberta.

	Number	Area burned (km^2)
Non-escaped fires[a]	991	4
Response failures[b]	123	3 574
Containment failures[c]	82	641
Total	1 196	4 219

[a] Fires with final size < 3 ha.

[b] Fires with size at IA ≥ 3 ha.

[c] Fires with size at IA < 3 ha, and final size ≥ 3 ha.

Table 3.5: Logistic regression models of response failure probability (p_i), with relative measures of model support and performance. Models were fitted to a sample of 1 196 fires that occurred in northeastern Alberta from 1995 to 2002.

Model	Model covariates[a]	k[b]	AIC[b]	w_i[b]	AUC[c]
Null	~ 1	1	794		
G_L	$YE+CA+SE+PL_3+PL_3^2+RO_3+RO_3^2+SL_3+PL_{12}+PL_{12}^2+RO_{12}+RO_{12}^2+SL_{12}+DE+WS+BS+PI+BU+CU+\ln(RT)+\ln(ISI)+\ln(FWI)$	24	776	0	0.71
W_E	$\ln(ISI)+\ln(FWI)$	3	770	0	0.65
V_{EG}	DE+WS+BS+PI+BU+CU	7	794	0	0.58
LF_1	PL+RO+SL	4	799	0	0.53
LF_2	$PL_3+PL_3^2+RO_3+RO_3^2+SL_3$	6	799	0	0.54
LF_3	$PL_{12}+PL_{12}^2+RO_{12}+RO_{12}^2+SL_{12}$	6	802	0	0.53
B_SP_I	$CA+SE+BS+PI+\ln(ISI)+\ln(RT)$	8	754	0.03	0.70
LF_{SE}	$CA+SE+(PL_3+SL_3)+\ln(ISI)+\ln(RT)+SE\times(PL_3+SL_3)$	9	760	0	0.69
LF_{WE}	$CA+SE+(PL_3+SL_3)+\ln(ISI)+\ln(RT)+\ln(ISI)\times(PL_3+SL_3)$	8	759	0	0.69
CO_{WE}	**$CA+SE+BS+PI+\ln(ISI)+\ln(RT)+SE\times BS+\ln(ISI)\times PI$**	**11**	**747**	**0.96**	**0.72**
RT_{WE}	$CA+\ln(ISI)+\ln(RT)+\ln(ISI)\times\ln(RT)$	5	768	0	0.67
W_{SP}	$CA+SE+WS+\ln(ISI)+\ln(RT)$	7	757	0.01	0.69

Note: The best model overall is shown in bold.

[a] Variable names are as described in Table 3.2.

[b] k = number of parameters in the model, AIC = Model Akaike Information Criteria, w_i = Akaike weights.

[c] AUC = Area Under the Receiver Operating Characteristic Curve.

Table 3.6: Logistic regression models of containment failure probability (q_i), with relative measures of model support and performance. Models were fitted to a sample of 1 073 fires that occurred in northeastern Alberta from 1995 to 2002.

Model	Model covariates[a]	k[b]	AIC[b]	w_i[b]	AUC[c]
NULL	~ 1	1	581		
G_L	$YE+CA+SE+PL_3+PL_3^2+RO_3+SL_3+PL_{12}+PL_{12}^2+RO_{12}+SL_{12}+DE+WS+BS+PI+BU+CU+\ln(RT)+\ln(ISI)+\ln(FWI)+SIA_2$	23	469	0	0.86
W_E	$\ln(ISI)+\ln(FWI)$	3	556	0	0.68
V_{EG}	$DE+WS+BS+PI+BU+CU$	7	590	0	0.55
LF_1	$PL+RO+SL$	4	586	0	0.54
LF_2	$PL_3+PL_3^2+RO_3+SL_3$	5	587	0	0.52
LF_3	$PL_{12}+PL_{12}^2+RO_{12}+SL_{12}$	5	587	0	0.52
H_{98}	**$\ln(ISI)+SIA_2$**	**3**	**439**	**0.78**	**0.85**
H_{04}	$(DE+WS)+BS+PI+\ln(ISI)$	5	558	0	0.68
B_SP_I	$SE+BS+PI+\ln(ISI)+SIA_2$	7	445	0.04	0.85
LF_{SE}	$SE+(PL_3+SL_3)+\ln(ISI)+SIA_2+SE\times(PL_3+SL_3)$	8	447	0.02	0.85
B_SS_{IA}	$SE+BS+\ln(ISI)+SIA_2+SIA_2\times BS$	7	442	0.14	0.86
LF_{WE}	$SE+(PL_3+SL_3)+\ln(ISI)+SIA_2+(PL_3+SL_3)\times\ln(ISI)$	7	446	0.03	0.85

Note: The best model overall is shown in bold.

[a, b, c] As in Table 3.5.

Table 3.7: Maximum likelihood estimates, confidence intervals and p-values for each term included in model CO_{WE} (Table 3.5). Significant terms ($p < 0.05$) are shown in bold.

Term[a]	β	95% CI	p-value
Intercept	**-3.83**	**1.06**	**<0.01**
CA	**-0.77**	**0.75**	**0.04**
SP	0.71	0.94	0.14
SU	**-1.06**	**0.87**	**0.02**
BS	-0.31	1.05	0.57
PI	-8.70	8.81	0.05
ln(ISI)	**0.68**	**0.37**	**<0.01**
ln(RT)	**0.15**	**0.12**	**0.01**
SP×BS	0.37	1.38	0.60
SU×BS	**1.52**	**1.27**	**0.02**
PI×ln(ISI)	**4.19**	**3.89**	**0.03**

[a] Variable names are as described in Table 3.2.

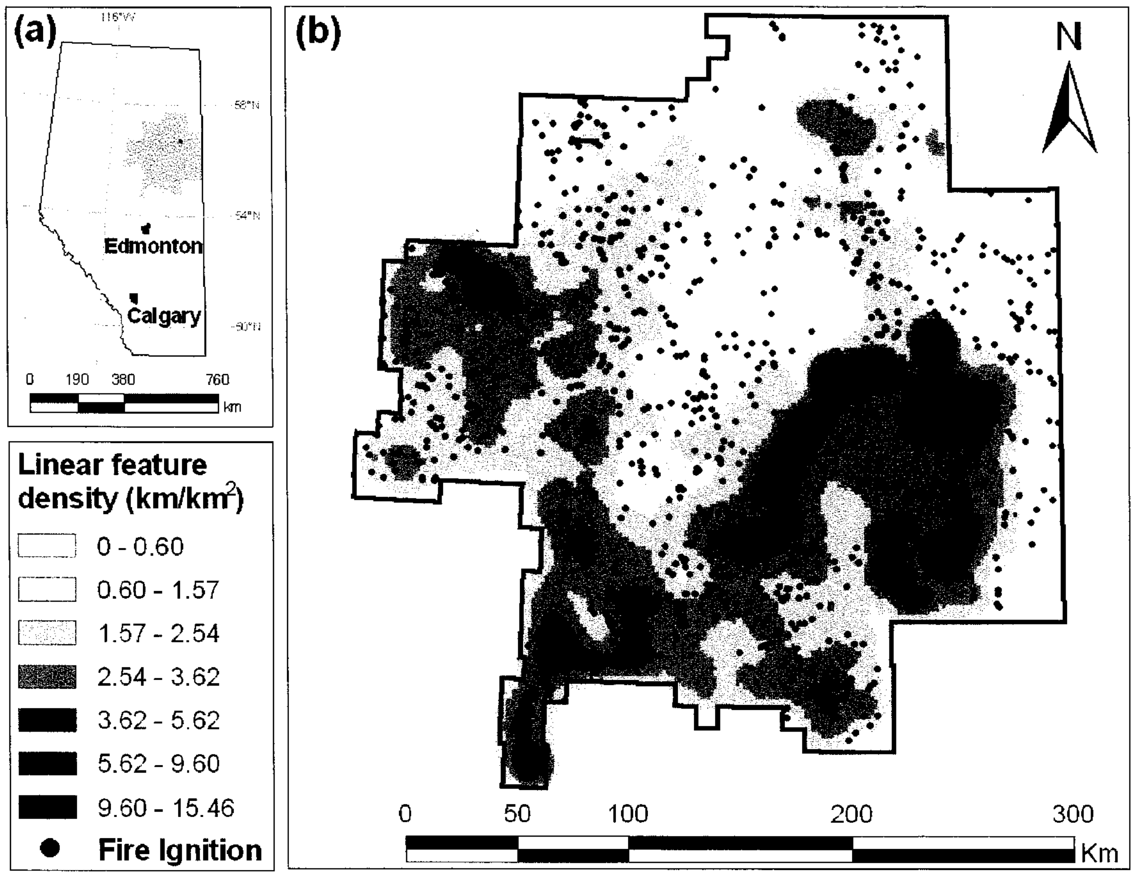

Figure 3.1: (a) The study region covers ≈67 000 km² of boreal mixedwood forest in northeastern Alberta, Canada. (b) The area has been disturbed by human activities since the 1950's, resulting in varying levels of linear feature (roads, pipelines and seismic lines) densities. The map also shows the ignition locations for the 1 196 fires that were used in the analyses.

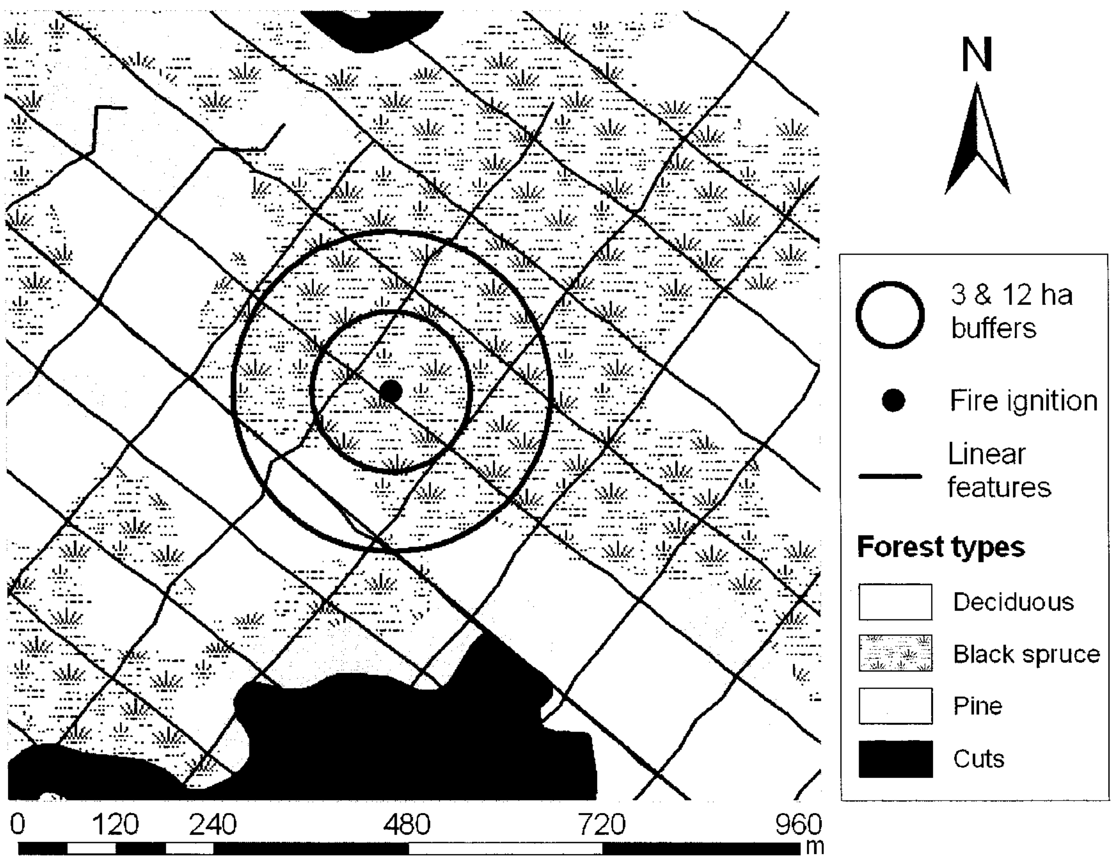

Figure 3.2: An example of variable collection for each fire. Linear feature densities were calculated within a 3 and 12 ha buffer around the fire ignition location; proportion of six types of forest fuels (see section 3.2.2 for details) were calculated only within the 3 ha buffer.

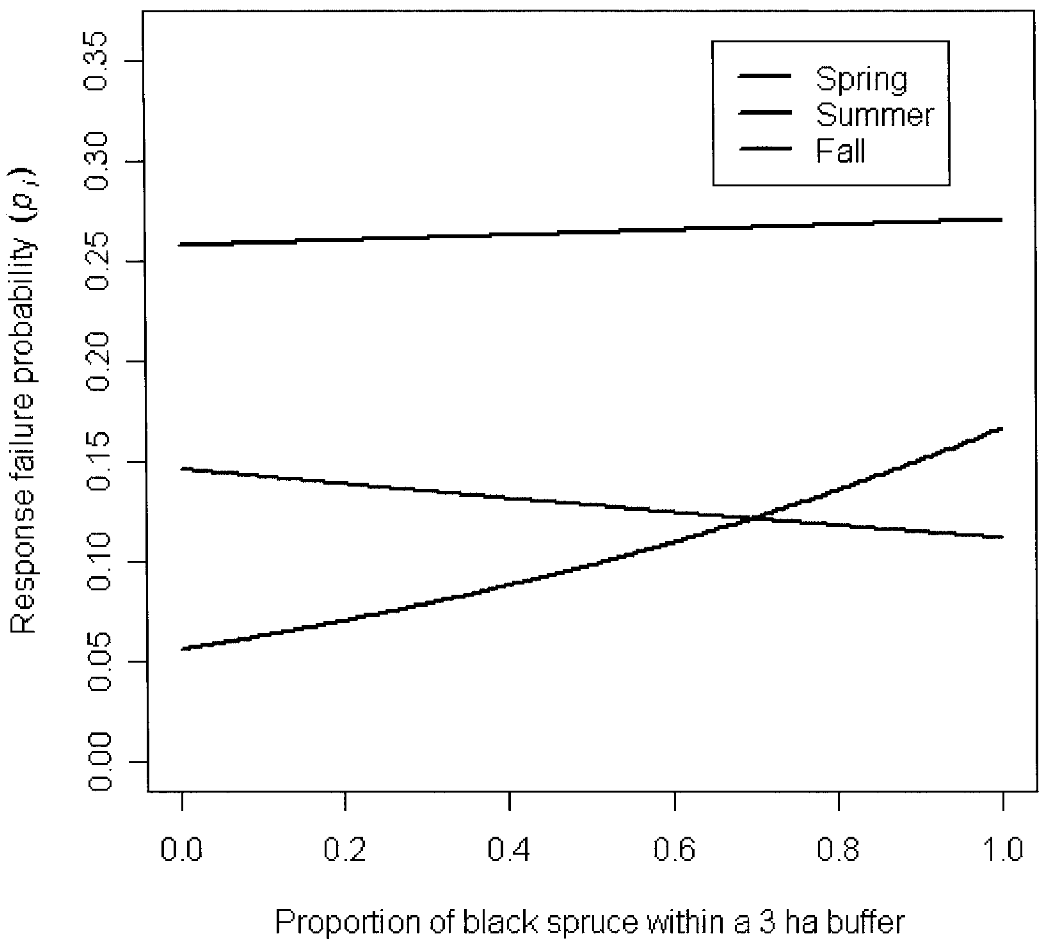

Figure 3.3: Response failure probability (p_i) as a function of the proportion of black spruce within a 3 ha buffer surrounding the fire ignition location. Fitted logistic probabilities were estimated using model CO_{WE} (Table 3.7). Fitted probabilities are graphed separately for spring, summer and fall, and were calculated using cause = lightning, and the mean values for the rest of the covariates in the model.

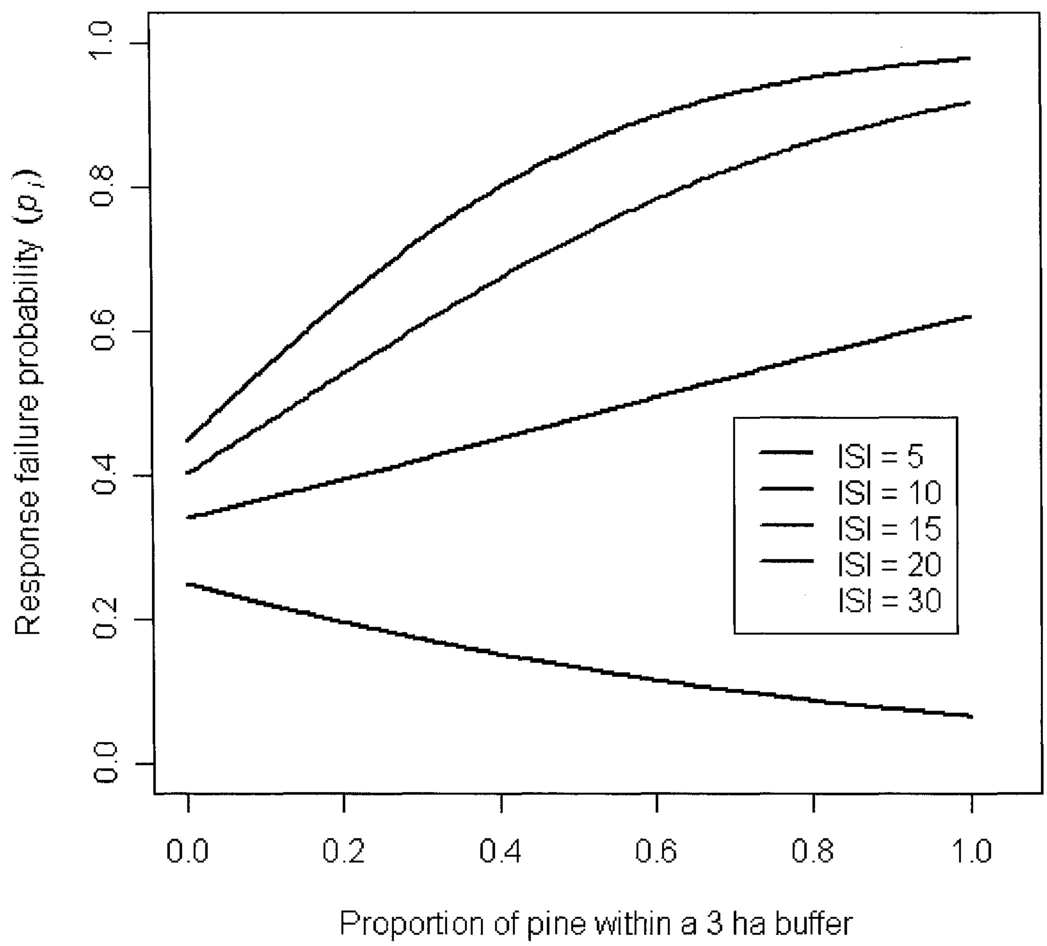

Figure 3.4: Response failure probability (p_i) as a function of the proportion of pine within a 3 ha buffer surrounding the fire ignition location. Fitted logistic probabilities were estimated using model CO_{WE} (Table 3.7). Fitted probabilities are separately graphed for 5, 10, 15, 20 and 30 ISI values, and were calculated using cause = lightning, season = spring, and the mean values for the rest of the covariates in the model.

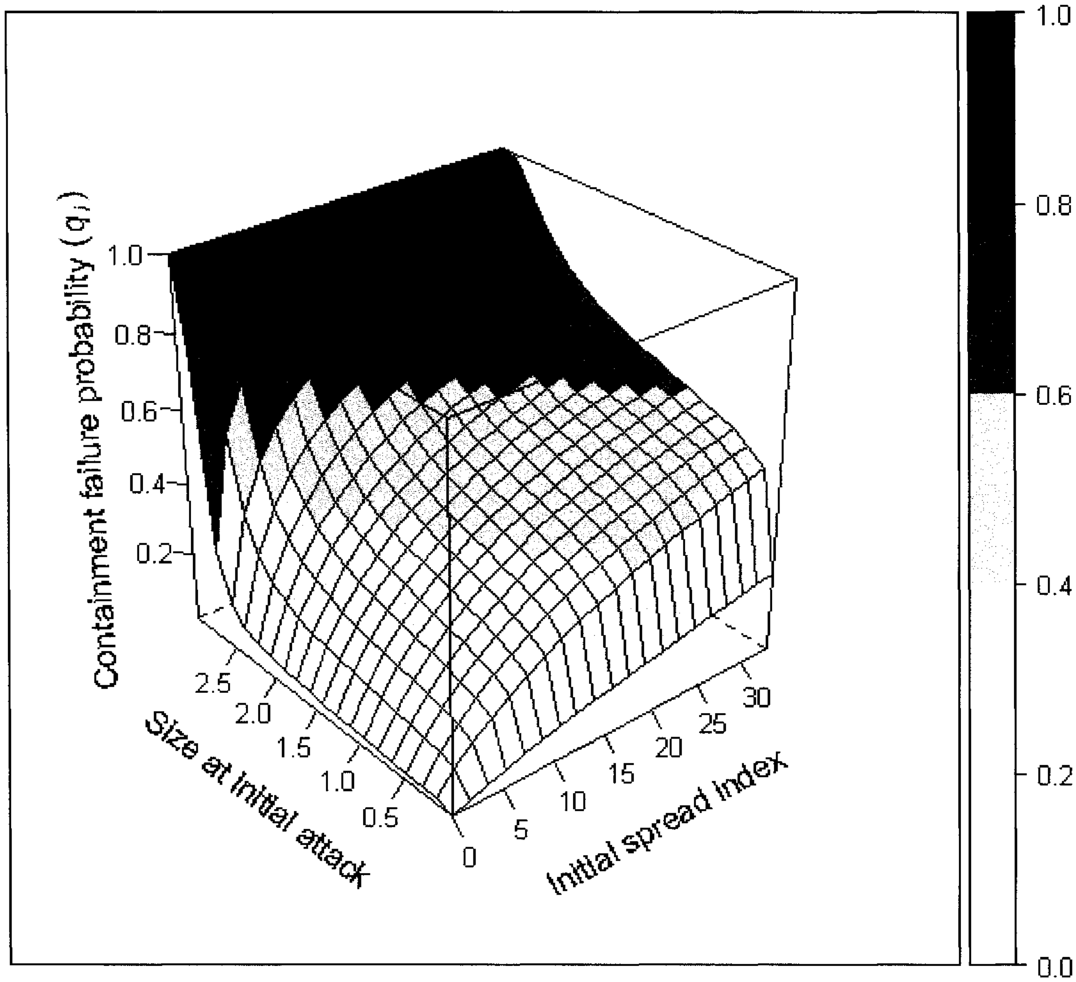

Figure 3.5: Fitted containment failure probability (q_i) estimated using model H_{98} (Table 3.6). The fitted model uses the transformed variables ln(ISI) and SIA_2 = ln[SIA/(3-SIA)] as covariates. However, the x and y axes are displayed on a natural scale for ease of interpretation.

Chapter Four

General Conclusions

4.1 Research summary and conclusions

Linear features in the boreal mixedwood forest of northeastern Alberta have been hypothesized to exert both positive and negative effects on various aspects of fire behaviour. In Chapter Two, I tested for evidence of a positive relationship between lightning-caused fire ignition frequency and linear feature density, expected to derive from the increased availability of fine fuels which are highly flammable at least during certain times of the year. I found a strong positive association between the frequency of days when lightning-caused fires occur and the density of roads in a landscape; this was consistent between two scales, ≈10 000 and ≈2 400 ha, and strongest during summer and spring. Additionally, I found that the effect of pipelines was confounded by the presence of roads. Contrary to what might have been expected, I found no evidence for an influence of seismic lines, even after accounting for seasonality and scale dependency. I also tested for the effects of forest type, geographic location, and fire weather and lightning conditions; I found that ignition frequency was strongly influenced by forest composition, supporting findings by Krawchuk *et al.* (2006) for the same study region.

In Chapter Three, I tested the contrasting hypotheses that linear features can act as fire breaks or fire spread corridors, or improve access for fire fighting activities and the rate of fireline construction. From these, one would predict either positive or negative effects on fire suppression effectiveness, as measured by the proportion of fires that escape IA (final size ≥ 3 ha). I found no evidence for a net effect of road, pipeline or seismic line densities within ≈100 m of the ignition point on the probability of a fire escaping initial attack. Rather, I found this process to be significantly influenced by forest composition, fire weather conditions, and indicators of fire management decisions, namely response time and size at initial attack. These findings are consistent with previous studies (Hirsch *et al.* 1998, Hély *et al.* 2001).

Fires cannot start or spread on paved or gravel roads; thus, the correlation between roads and increased lightning-caused fire frequency must derive from some other attribute or process associated with roadways. In my view, the obvious candidates are roadside clearings, which are vegetated mostly by grasses. The high surface area to volume ratio of grass leaves and the accumulation of dead biomass increase the probability of fire ignition, relative to the original forest cover. If increased ignitions result from the higher flammability of the roadside grass cover, such easily combustible fuels would also be expected to enhance fire spread, making effective suppression more difficult. Indeed, field observations by fire suppression personnel on the Freeman River fire in northern Alberta report that heavier fuel loads on linear disturbances within the fire perimeter resulted in a more intense fire that was harder to contain (FERIC 2006a). It is possible that linear features may have important effects on the behaviour of particular fires, inhibiting or enhancing fire suppression; however, I found no evidence for any net effect within my study region and time interval. It is unclear how roadsides can increase the frequency of lightning-caused fires, yet have no detectable net effect on suppression effectiveness. The density of linear features is projected to increase rapidly throughout the boreal forest over the next 50 years or so (Schneider *et al.* 2003). Thus, it is plausible that at some stage they will begin to exert a significant net influence, yet the magnitude and direction of any such effect cannot be derived or estimated from the present analysis.

In Chapter Two, I predicted that a 'road-saturated' landscape would experience an increase of 37% in the frequency of days where lightning-caused fires occur. Of course, this hypothetical landscape full of roads might never materialize; however, this result suggests that projected road developments in the region will bring associated increases in fire frequency, which has the potential to alter the structure and composition of boreal mixedwood landscapes in the long run. Weir and Johnson (1998) describe an analogous situation in Saskatchewan's mixedwood region, where forests were subjected to short interval fires for agricultural clearance. This resulted in a forest depleted of species that mature at older ages and have sexual reproduction (white spruce), while at the same time promoted the growth of trees with vegetative reproduction (aspen) or serotinous cones (jack pine). Native and non-native grass invasions along roadsides can also create positive feedback cycles, where an increase in the number of fires enhances the invasion of these plant species given their rapid post-disturbance recovery rates relative to forest tree species (D'Antonio and Vitousek 1992). In summary, increasing road densities could potentially drive a mixedwood-dominated forest towards a landscape with progressively higher deciduous and grass components.

Such described changes in landscape structure and composition would translate into economic impacts on the public, by increasing fire suppression costs, and on the forestry industry, by a decrease in the coniferous timber supply. However, it is possible that these effects could be mitigated by active forest management or other compensatory feedbacks (e.g. aspen stands having fewer fires (Krawchuk *et al.* 2006)). In my study region, certain silvicultural practices might be able to hinder the invasion of disturbance-adapted grass species. It is unknown though whether the extent of the change in the fire regime resulting from the grass/fire feedback could be offset by these silvicultural activities, particularly in the face of potential interactions with processes which can further alter fire regimes and forest dynamics such as climate change (Amiro *et al.* 2001, Flannigan *et al.* 2001, Chapin *et al.* 2004) and forest harvesting (M. Krawchuk, personal communication, 2006).

4.2 Limitations and future research

In general, statistical modelling of long-term datasets is considered an excellent tool for describing patterns in the occurrence of large scale processes such as fires. The recognition and description of current general patterns in fire occurrence and spread can help us predict future patterns, with the ultimate aim of improving fire planning and management. This can be successfully done within a modelling framework; however, it is important to recognize the limitations inherent to such approach.

A common limitation in this kind of research is the lack and quality of available data. Since good data are expensive and hard to obtain, the data that is collected and incorporated into these large scale - long term databases is seldom ideal. In the case of my project, I used Alberta's provincial fire database. I found that several fire records within this database had inconsistencies, such as for example bigger fire sizes at initial attack than final fire size, or even some fires that had final sizes of less than zero. Thus, I had to scan the database for this type of errors and correct them or discard some of the fires. Also, it is evident that the reported ignition locations for each fire have some associated error, which arises from the instrument used to take the measurement or even from the difficulty of identifying the exact point on the ground where some fires started. In general, fire personnel state that this error lies between 50 and 80 m; unfortunately, there is no mention as to what the error associated to each individual recorded fire is. Finally, there is no estimate of the number of fires that go out before being detected, since these are missing from the fire record.

The linear feature data layers that I used also had various shortcomings. From the short amount of time I spent in the field, it was evident that these layers were somewhat incomplete. Also, many of the seismic lines had been upgraded to roads. I think an intensive ground proofing of these layers is necessary, though it would represent a large amount of work. Also, as mentioned in Chapter Two, the seismic line layer includes features that are highly diverse in terms of their width, the state of plant regeneration and the amount of water on the line itself. However, the layer does not include any information with regards to these characteristics, so I had to consider all seismic lines the same. More detailed information on seismic line characteristics, or even categorizing each section of seismic line based on the forest stand adjacent to it might prove a better approach to assess the real effect of this landscape element.

A major limitation I was faced with was the lack of time series data on linear feature developments. Thus, I had to use a fairly recent and short time interval over which I conducted my analyses: 1995 to 2002. The main assumption I had to make was that linear features and vegetation structure had remained constant over that period; however, this might not have been the case given the large amount of industrial development in the area. Furthermore, the chosen time interval included several high fire years, particularly 1998, which might have influenced some of the positive results I obtained. If time series linear feature data was available (for example, from analysis of aerial photography), a better approach could be to analyse annual fire patterns with respect to annual changes in landscape characteristics, as well as including more years of fire data.

My thesis focused on the effects of linear features on lightning-caused fire ignition and escape from initial suppression actions. However, these represent only two aspects of fire behaviour; as described in Chapter One, linear features in the boreal forest have the potential to influence other aspects as well. An interesting study would be to consider the increase in human-caused fire ignitions associated with improved access to previously remote areas provided by linear corridors. Human-caused fires are a source of increasing concern because they generally produce more damage, since they tend to burn in different stand types than lightning-caused fires and in closer proximity to human centers. Also, it would be interesting to assess the change in the distribution of final fire sizes resulting from the discontinuity of forest fuels caused by linear disturbances, since the role of these elements as fire breaks might prove of importance for the natural growth of large escaped fires.

My research showed that landscapes with more roads experienced an increased frequency of days where lightning-caused fires occurred. From this, I inferred that native and non-native species of grasses, thriving along roads right-of-ways, augmented the flammability of the fuel bed. Ideally, such effect should be demonstrated through an experimental study, wherein the precise identity and flammability of the plant community growing on roadsides should be assessed and compared to other forest fuel types and linear feature classes. However, experiments of this kind are logistically difficult; particularly given the hazard associated with recreating fire under natural conditions, and the questionable realism of recreating fire within a laboratory setting.

Potential changes in the fire regime and in forest structure and composition deriving from the hypothesized roadside vegetation – fire feedback cycle could be assessed through a simulation modelling study. Such exercise should also incorporate possible roadside management strategies, in order to assess their economic costs and long-term effectiveness in offsetting such predicted changes.

4.3 Management recommendations

Linear features in the boreal mixedwood forest should be designed and managed to minimize the potential for increased lightning-caused fire ignition. If indeed roadside clearings are found to be responsible for this effect, government and industry should focus on planting less flammable species along right-of-ways. Revegetation and restoration standards should be based on empirical studies of the ignition potential of different plant species and fuel types (Latham and Schlieter 1989; Hogenbirk and Sarrazin-Delay 1995). In Alberta, some research is already underway as to what the characteristics and identity of suitable grass and forb species should be (FERIC 2006b). As an interim measure, it is advisable to plant native species instead of general agronomic seed mixes, since these mixes tend to be contaminated with exotic plants. As noted by Stoner *et al.* (2004), the addition of non-native grasses to the native pool already thriving along roadsides represents a new source of combustible fuel which can further enhance ignition.

Fire managers should attempt to systematically monitor and report any apparent effects of linear feature on fire spread and suppression effectiveness. As noted, the evidence for such effect is currently weak, and mostly based on anecdotal reports by individual fire fighters from specific fires, which may have occurred under particular or unusual circumstances. Studies like the one in Chapter Three could be repeated in the future using more refined fire-specific data and up-to-date linear features coverages, in order to verify if my results remain valid as the density of linear disturbances increases in this region.

Finally, the top containment failure model from Chapter Three suggested that a reduction in area burned might be possible if additional fire-specific factors affecting response failure probability could be incorporated into operational decisions. Burning pine stands under high winds and low moisture, and burning black spruce stands in summer represent two conditions where the hazard of fire escape is high. This information should be incorporated into the decision process by which suppression resources are deployed to co-occurring fires, since it could help reduce the probability of fire escape before any actions are attempted, and thus the size and area burned by these fires.

Fire risk is a major concern in the Canadian boreal mixedwood forest, particularly where humans live in close proximity to forests; in consequence, fire management agencies expend great effort on preventive measures as well as in suppression (Kourtz and Todd 1991, Martell 2001). In order to quantify and control for these risks, resource managers need to know when and where fires are most likely to occur, and how biological, physical and anthropogenic factors are related to each component of fire behaviour (Hirsch and Martell 1996, DellaSala *et al.* 2004). This requires empirical models with explanatory and predictive power, such as the ones developed in this thesis. These models contribute to the understanding of fire dynamics in boreal forests, and can be used as inputs for operational fire management as well as for models of integrated ecosystem management.

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