CURRENT AND FUTURE WILDFIRE RISK IN THE PERI-URBAN ACADIAN FOREST REGION

by

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ABSTRACT

The majority of the world's population now lives in urban areas, and the peri-urban has grown simultaneously, creating new Wildland-Urban Interface (WUI) where development comes into contact – and intermingles with – wildlands. WUI has an elevated wildfire risk. This study examines current and future wildfire risk in the Acadian Forest Region, and consists of two papers. The first manuscript of this thesis describes a model to delineate WUI at a site-scale for municipal risk management, using fire behaviour modelling. The second manuscript uses climate and fire behaviour modelling, projecting an increase in fire weather severity in the Acadian Forest Region under climate change, indicating increased future fire susceptibility. Shifts in tree species composition may offset this risk, as tree species become a negative fire risk driver. The relative importance of fire risk drivers was solicited from experts to assess the net impact on fire risk. Together these papers identify an increasing fire risk in the region under climate change, depending on site-level tree species composition dynamics, and an opportunity for municipal management of fire risk. Delineation of WUI and risk management are necessary, given increasing future fire risk and uncertainty under climate change.

LIST OF ABBREVIATIONS USED

AFR – Acadian Forest Region
AHP – Analytical Hierarchy Process
BP – Burn Probability
CI – Consistency Index
CR – Consistency Ratio
CRCM – Canadian Regional Circulation Model
FBP – Forest Fire Behaviour Prediction
GHG – Greenhouse Gas
HFI – Head Fire Intensity
HRM – Halifax Regional Municipality
IPCC – Intergovernmental Panel on Climate Change
DEM – Digital Elevation Model
DOQQ – Digital Orthophoto Quarter Quads
DSM – Digital Surface Model
FFC – Forest Fuel Code
FS – Fire Susceptibility
FWI – Fire Weather Index
LiDAR – Light Detection and Ranging
nDSM – Normalized Digital Surface Model
NDVI – Normalized Difference Vegetation Index
NIR – Near-Infrared
NIST – National Institute of Standards and Technology
NSDNR – Nova Scotia Department of Natural Resources
RI – Random Index
SRES – Special Report on Emissions Scenarios
WUI – Wildland-Urban Interface

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CHAPTER 1 INTRODUCTION

1.1 THE WILDLAND-URBAN INTERFACE OF THE HALIFAX REGIONAL MUNICIPALITY

The majority of the world's population now lives in urban areas, as do a preponderance of Canadians (Bollman & Clemenson, 2006; United Nations, 2011). Urban areas have grown to support the increasing population of urban residents, and suburbs have expanded simultaneously. Suburbs are "residential districts with low densities that are located at, or near, the urban fringe" (Harris, 2004, p. 7). In recent years the majority of new development in Canada was in the form of suburban growth (Turcotte, 2008). This suburban growth is in response to the demand for residences in close proximity to both urban and natural amenities (Hammer, Stewart, & Radeloff, 2009a; Poudyal, Johnson-Gaither, Goodrick, Bowker, & Gan, 2012). As suburban development expands outward from urbanized areas, it comes into contact with wildlands, known as the wildland-urban interface (WUI).

The WUI is the area where urban development meets and intermingles with wildlands (Hirsch, n.d.; McGee, 2007). Wildlands are natural areas that include forests, peatlands, shrubs, grasslands, tundra or alpine tundra, and other ecosystems. Vegetation in wildlands may serve as fuel for wildfires (Canadian Council of Forest Ministers, 2005). This peri-urban fringe area is associated with many environmental and management concerns. Past research about WUI has examined such broad topics as the impact of development on native species (Boren, Engle, & Masters, 1997; Chace, Walsh, Cruz, Prather, & Swanson, 2003), conservation and forest management issues (Vince, Duryea, Macie, & Hermansen, 2005), biodiversity and invasive species in urban fringes (Alston & Richardson, 2006; Alvey, 2006), and social research on the views of WUI residents (Blanchard & Ryan, 2007; Cortner, Gardner, & Taylor, 1990; McGee, 2007).

Of concern to this thesis is WUI and wildfire risk. Fire hazard in this thesis is defined as the potential fire behaviour of a fuel type, regardless of ignition sources and fire weather. Fire risk, by contrast, is the chance that a fire might start as affected by local fuel and weather conditions, as well as causative agents (Hardy, 2005). WUI has an inherently elevated wildfire risk, associated with the intermingling of natural and urban fuels, natural wildfire regimes in wildlands, and human-caused ignitions (Hirsch, n.d.; Radeloff et al., 2005; Theobald & Romme, 2007). The delineation of WUI areas with a focus on fire risk is important for the targeting of wildfire risk management initiatives (Haight, Cleland, Hammer, Radeloff, & Rupp, 2004).

This study examines the WUI of the Halifax Regional Municipality (HRM). The HRM is a large municipality in the centre of the Atlantic coast of Nova Scotia, Canada (Figure 1.1) (Halifax Regional Municipality [HRM], 2013a). Halifax is made up of the central Halifax serviced area, which includes Halifax, Dartmouth, Bedford, and Spryfield. Mainly rural lands border this developed core, and central Halifax is surrounded by stands of natural and managed forest (HRM, 2012). In 2011, central Halifax had a population of 297,943, making the cities of Halifax and Dartmouth the largest Canadian population centre east of Québec City (Statistics Canada, 2012).

The population of the HRM has grown steadily since the 1960s, and projections suggest that population growth will continue in the near future (HRM, 2013b). In recent years, the majority of this growth occurred in suburbs and commutershed areas (Figure 1.2), in keeping with the overall trend in North America (HRM, 2006; Seto, Fragkias, Güneralp, & Reilly, 2011; Turcotte, 2008). Despite setting a goal of slowing the trend of suburban growth, the HRM has failed to meet targets for urban and rural growth set out in the regional municipal planning strategy (HRM, 2006, 2013b; Our HRM Alliance, 2011). Thus, suburban areas of HRM have continued to expand into the wildlands surrounding the city, creating new WUI. Suburban communities in the HRM have experienced wildfires which damaged property, forced residents to evacuate, destroyed houses and cost the Municipality, the Province, homeowners, and insurance companies considerably (CBC News, 2008, 2010a; Fanning, 2010; Shea, 2012; Shiers, 2010). As the Municipality has become increasingly familiar and experienced with urban wildfire, identifying and managing fire risk in the WUI has become a priority.

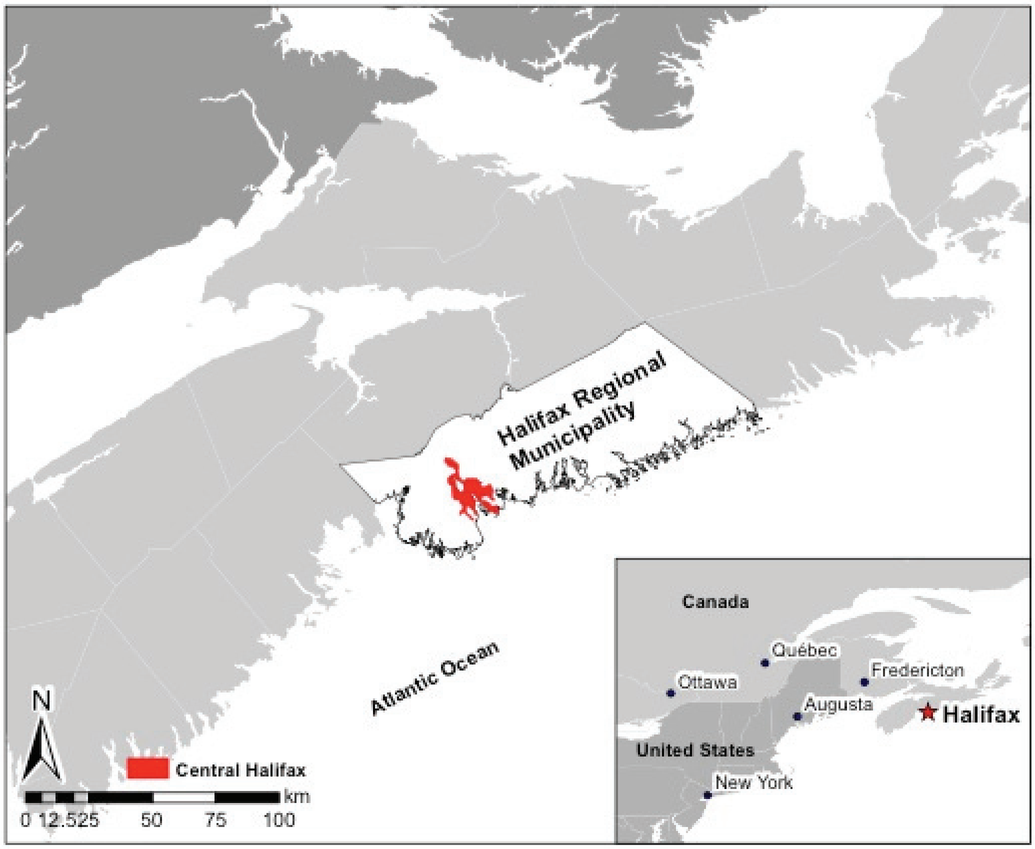

Figure 1.1 Location of Halifax Regional Municipality in North America.

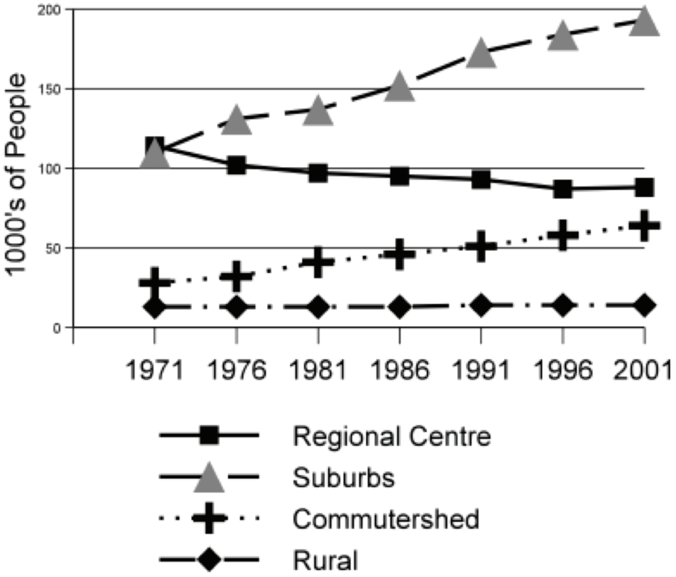

Figure 1.2 Halifax Regional Municipality population by subregion (1971 – 2001). Reproduced from HRM (2006) (copyright release included in Appendix A).

The forest fuels that surround central Halifax are stands of Acadian Forest Region (AFR) tree species. The AFR is a transitional forest zone between the northern boreal forest and the primarily non-coniferous forests of eastern Canada and the United States (Loo & Ives, 2003). The AFR covers the Maritime Provinces, and in places extends into to southern Québec and Maine (Loo & Ives, 2003; Mosseler, Lynds, & Major, 2003; Simpson, 2008). Due to the transitional nature of this forest zone the AFR is characterized by a mixed non-coniferous and coniferous tree community, with coniferous species making up a slight majority (Loo & Ives, 2003; Natural History Museum of Nova Scotia [NSMNH], 1994a). Typical tree species in this forest region are red spruce (*Picea rubens*), eastern hemlock (*Tsuga canadensis*), yellow birch (*Betula alleghaniensis*), sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), balsam fir (*Abies balsamea*), and white pine (*Pinus strobus*). Disturbance and poor site conditions favour black spruce (*Picea mariana*), red maple (*Acer rubrum*), white birch (*Betula papyrifera*) and aspen (*Populus* spp.) (Loo & Ives, 2003; Mosseler et al., 2003; NSMNH, 1994b; Steenberg, Duinker, & Bush, 2011).

Historically, the AFR had a natural regime of disturbance due to wildfire, which is especially severe in areas where coniferous boreal tree species dominate. Some local native species, such as jack pine and pin cherry, are adapted to wildfire and may require fire for regeneration (Hély, Bergeron, & Flannigan, 2000; Johnson, 1992; Neily, Basquill, Quigley, Stewart, & Keys, 2011; Neily, Quigley, Stewart, & Keys, 2007; Wein & Moore, 1977). Today the vast majority of wildfires in the province (97 - 99%) are lit intentionally or accidentally by human activity (Neily et al., 2007; Nova Scotia Department of Natural Resources [NSDNR], 2011, 2013). The HRM's pattern of development at urban fringes alongside stands of AFR fuel is exacerbating WUI wildfire risk.

1.2 PRESSURES ON THE HALIFAX REGIONAL MUNICIPALITY WILDLAND-URBAN INTERFACE

Development in the HRM has been characterized by suburban and commutershed growth, spreading outward at the fringes of the urban core (HRM, 2006, 2013b; Millward, 2002). As ongoing suburban growth encounters AFR wildlands, the HRM

WUI and associated area at risk from wildfire has increased. Additional external pressures associated with climate change and urban forest management are further complicating WUI management in the HRM.

1.2.1 CLIMATE CHANGE

Climate change due to anthropogenic emissions of greenhouse gasses (GHGs) is expected to produce a future climate in Nova Scotia different from that of today (Richards & Daigle, 2011). Human activities consuming fossil fuels, such as manufacturing, power generation, and transportation result in the production and emission of CO_2 and other GHGs. When the emission of these gasses exceeds the capacity of natural removal processes they accumulate in the atmosphere (Intergovernmental Panel on Climate Change [IPCC], 2007). Emissions of these gasses have grown fairly steadily from 1970 to 2005 (when cumulative global emissions were last examined) (Figure 1.3) (Herzog, 2009; IPCC, 2007).

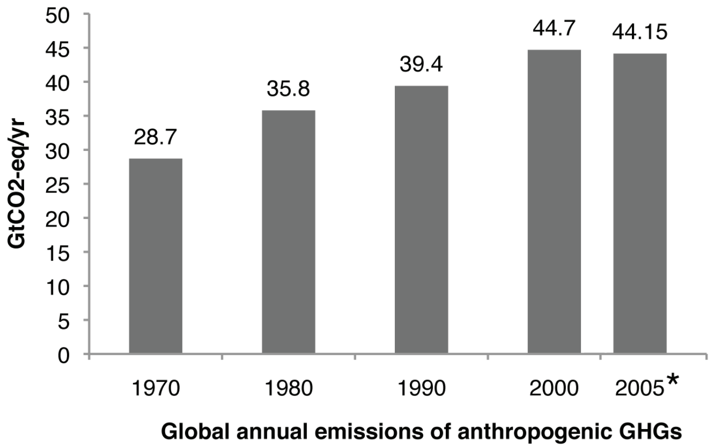

Figure 1.3 Global annual emissions of anthropogenic greenhouse gasses from 1970 to 2004, expressed in CO_2 equivalents. All data from IPCC, 2007, except * from Herzog, 2009.

Concentrations of some GHGs (CO_2 and CH_4) in the atmosphere have far exceeded the known natural range (IPCC, 2007). Recent declines in GHG emissions observed since 2009 are partially attributed to the global economic downturn, due to the relationship between gross domestic product and fossil fuel use (Cambridge Econometrics, 2009; Stecker & ClimateWire, 2011; York, 2012). Overall, anthropogenic emissions of GHGs are gradually warming the surface of the earth due to the dominance of these gasses in the radiative forcing (warming influence) of the global climate system (IPCC, 2007).

Given these continued increases, anthropogenic GHG emissions are expected to cause temperature and precipitation shifts. These changes to the climate will drive changes in the Nova Scotia forest community and landscape, impacting future wildfire risk (IPCC, 2007; Johnston et al., 2009; Millar, Stephenson, & Stephens, 2007). In the AFR the outcomes of future climate change are unclear, due in part to the transitional nature of this forest zone. Cumulative impacts of climate change on future fire risk are uncertain, as some drivers are projected to increase the risk under future climate change while others are expected to decrease the risk of wildfires. Drivers of fire risk in the AFR are: (a) fire weather during the fire season, including temperature and precipitation; (b) the length of the fire season; (c) tree species composition; (d) accumulation of coarse woody debris due to wind throw, pests, and tree species decline; (e) weakened stands of trees; and (f) decomposition of dead trees.

Projections suggest that the HRM will experience increases in the volume and frequency of precipitation, increasing storm event severity, and increases in temperature under future climate change (Figure 1.4) (Jiang & Perrie, 2007; Richards & Daigle, 2011). These changes to the weather pattern should impact the weather during the fire season, and lengthen the fire season, allowing more opportunities for ignitions to occur (Flannigan & Wang, 2012; Hessl, 2011; Wotton & Flannigan, 1993). In Canada climate change-caused warming correlates with increases in the area burned (Gillett, Weaver, Zwiers, & Flannigan, 2004). Conversely, the frequency of rain negatively correlates with area burned by wildfires in Canadian forests (Flannigan & Harrington, 1988). As the weather pattern in Nova Scotia warms and as the frequency and volume of precipitation increases (Figure 1.4), it is unclear how these processes will drive changes in future wildfire risk.

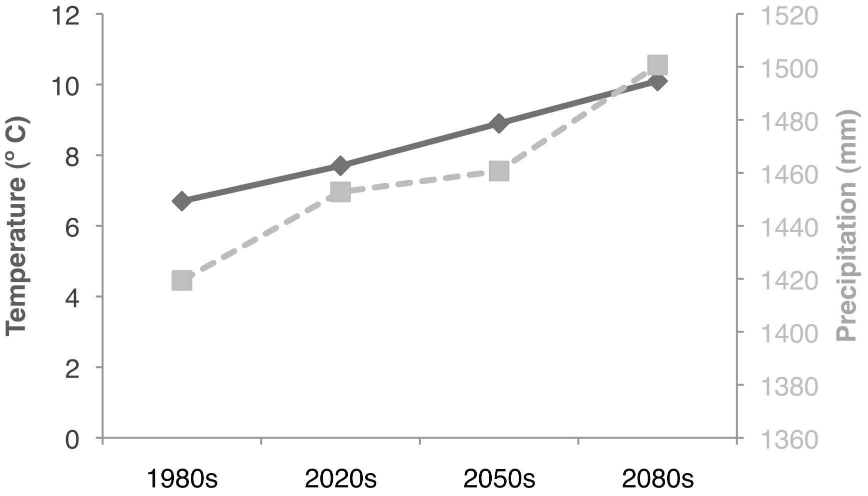

Figure 1.4 Projected average yearly temperature (solid line in dark grey) and annual precipitation (dashed line in light grey) for the Halifax Regional Municipality. Data from Richards & Daigle, 2011.

In Nova Scotia's mixedwood forests wildfire risk is closely related to tree species composition, as conifers are more susceptible to ignition, and are more likely to lead to serious crown fires (Hély et al., 2000; Johnson, 1992). Changes to the future climate should also influence the tree species composition of the forest community in the AFR, creating environmental conditions that favour non-coniferous eastern tree species, and disfavour boreal species, conifers in particular (Bourque, Hassan, & Swift, 2010; Johnston et al., 2009; Steenberg et al., 2011). Non-coniferous species are better adapted to thrive in the warmer, wetter climate expected in Nova Scotia.

A transition to a non-coniferous tree species-dominated forest would reduce wildfire risk in the HRM; however, the research literature suggests that tree species range shifts will lag behind changing climate envelopes. Maladapted species may not be rapidly replaced, potentially leaving mid-term forests of boreal species, increasing wildfire hazard (Iverson & McKenzie, 2013; Iverson, Prasad, Matthews, & Peters, 2008; McKenney, Pedlar, Lawrence, Campbell, & Michael, 2007; Steenberg, Duinker & Bush, 2013). In addition, projected increases in severe hurricane activity would likely increase wind throw in local forests (Foster, 1988a; Jiang & Perrie, 2007). Conifers are particularly susceptible to wind throw. Thus, increasing storm activity may hasten the decline of boreal species, and lead to accumulation of coarse woody debris, further increasing local wildfire hazard in the short term (Brown, Reinhardt, & Kramer, 2003; Foster, 1988a, 1988b; Liu, Lu, & Shen, 2008). Over a longer time span, this accelerated removal of boreal species would allow for a more rapid transition to a non-coniferous forest, reducing wildfire risk.

Climate change in the AFR may also influence the frequency, severity and timing of pest and disease outbreaks. In general, warming temperatures are associated with increased metabolic rates in insects, as well as accelerated development, increased consumption, and spread of populations. Warmer temperatures in the winter could lead to a reduction in insect mortality (Dukes et al., 2009). Mortality of tree stands may also alter local microclimate, leading to further warming in regions affected by insect outbreaks (Maness, Kushner, & Fung, 2012).

Two insects native to the AFR and with spruce tree hosts are projected to increase tree mortality, and thus fire risk under climate change. In the HRM, spruce budworm (*Choristoneura fumiferana)* outbreaks in inland areas are projected to show a slight increase in severity, with a slight decline in coastal regions of the municipality (D. Gray, personal communication, November 19, 2012; Gray, 2007). Although changes in the severity of spruce budworm outbreaks are expected to be minor, outbreaks are projected to increase in duration. Associated with longer outbreaks are increases in tree mortality, which in turn elevates local wildfire risk (Fleming, Candau, & Mcalpine, 2002; Gray, 2007). Spruce budworm-damaged stands are also more susceptible to wind throw, further increasing tree mortality and the accumulation of woody debris and ladder fuels, acting as a positive driver of future fire risk in the HRM (Fleming et al., 2002; Gray, 2007; Taylor & MacLean, 2009). Modelling of the native spruce beetle (*Dendroctonus ruffipenis*) life cycle and outbreak patterns under a warmer future climate projects shorter generation times, and a much higher probability of outbreaks occurring in the boreal forest (Bentz et al., 2010), suggesting that the AFR may also face increased spruce beetle outbreak potential.

In general, the reviewed research literature suggested that invasive species may benefit from future climate change in this region, negatively impacting forests (Dukes et al., 2009), although not all such species will thrive. Brown spruce longhorn beetle (*Tetropium fuscum*) is a significant concern in Halifax's forests but research on this nonnative invasive insect is limited. Over time, the biotic disturbance of Atlantic Canadian forests from insects and pathogens are expected to increase moderately (Johnston et al., 2009).

The altered future climate could also impact the susceptibility of trees to forest pathogens. Weakened, maladapted trees are more susceptible to disease and fungus than those that are healthy (Dukes et al., 2009; Johnston et al., 2009). Direct effects of climate change on tree pathogens may include "(i) increased growth and reproduction; (ii) altered propagule dispersal, transmission rates, and infection phenology; and (iii) changes in overwinter survival" (Dukes et al., 2009, p. 236).

Phenology, or life cycle timing, of insects and insect predators may also be modified by future climate change. The disruption of synchrony between life cycle events of forest pests and plants, or insects and their predators, is a likely pathway for climate change to influence future forest composition and wildfire susceptibility. This is very difficult to model, and presents a challenge to researchers, which prevents its explicit inclusion in this synthesis (Johnston et al., 2009; Logan, Régnière, & Powell, 2003).

Increases in fuel loading under climate change, such as that from maladapted tree species, wind throw, or disease, may be mitigated in part by an increased rate of decomposition, which increases with increased temperature (Anderson, 1991; Chambers, Higuchi, Schimel, Ferreira, & Melack, 2000). Decomposing and decomposed slash fuels burn at less intense levels and more slowly than freshly downed woody debris, driving fire risk down (Brown et al., 2003; Foster, 1988a; Kurt & Scott, 2007; Liu et al., 2008; Zeng et al., 2009). It has already been noted that temperature is expected to increase in the future due to climate change (Richards & Daigle, 2011), which should affect the profile of decomposition in the future.

1.2.2 URBAN FOREST MANAGEMENT

In many parts of North America urban tree cover is declining, making the protection and expansion of urban forests a growing priority for managers (Nowak & Greenfield, 2012). In the HRM this prioritization of urban forests is recognizable through the recent adoption of the municipal Urban Forest Master Plan (UFMP). The UFMP will guide management decisions about the local urban forest, ensuring its long-term sustainability (HRM, 2012). Promoting a healthy urban forest and increasing canopy cover requires the protection of existing urban trees and remnant patches, as well as the planting of new trees to replace those that decline, die, or are removed (HRM, 2012; Nowak & Greenfield, 2012).

Fuel management is the intentional manipulation of fuels and flammable material to adjust fire behaviour, fire effects, and the ease of suppression in the event of a fire (Canadian Council of Forest Ministers, 2005). The use of defensible space fuel management around a home significantly reduces the likelihood that the structure will be destroyed in a fire (Bhandary & Muller, 2009; Gibbons et al., 2012). Implementation of defensible space, and the Canadian *FireSmart* program recommendations for protection of homes from wildfire require the removal of trees and vegetative fuels close to houses (HRM, 2011a; Partners in Protection, 2003). In WUI areas it is difficult to reconcile urban forest promotion with wildfire risk reduction fuel management treatments, as the management responses for the two appear to be in opposition with one another.

1.3 EXISTING MODELS OF WILDLAND-URBAN INTERFACE IDENTIFICATION FOR WILDFIRE RISK MANAGEMENT PURPOSES

Many researchers have previously developed approaches to delineate WUI for wildfire risk management, using proxy-based models and buffer-based models. Despite the availability of existing models, there are few that are ideal for delineation of WUI for municipal purposes and take advantage of detailed data, and advanced fire behaviour and susceptibility modelling software. Existing models of WUI delineation developed for wildfire risk management are discussed in this section.

1.3.1 PROXY-BASED MODELS

Early models of WUI delineation were developed with the purpose of modelling a large extent at a small map scale, from the conterminous United States to all of a sate or county. Where community and site-level planning is of interest, these scales do not provide adequate detail. These models use proxies for risk, typically identifying WUI as areas where low-density development intersects with natural vegetation, representing both ignition potential and fuel (Gering, Chun, & Anderson, 1998; Radeloff et al., 2005; Stewart, Radeloff, Hammer, & Hawbaker, 2007; Theobald & Romme, 2007). These models generally originate from the United States Federal Register definition of WUI, which included thresholds for population density to identify WUI for targeted national funding of fire risk management activities (Federal Register, 2001; Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007).

Many examples of models using proxies for delineation of WUI exist. One such early model is that by Gering, Chun and Anderson (1998), who identified WUI as areas outside of town limits where land cover data indicated wildland fuels, and housing unit density was between one and 15/km^2. This model also included economic variables by identifying thresholds of median household income. Gering et al.'s (1998) model was also employed in student research done for the HRM in the area of Fall River (Gale, 2011). Radeloff et al. (2005) developed a similar model (excluding the use of financial thresholds), and conducted one of the first studies examining the WUI of the conterminous United States. The work of Stewart et al. (2007) consisted largely of the same group of researchers and the same study area. As these two models were developed based on the United States Federal Register definition, the models differentiate WUI based on the categories of interface and intermix WUI, with intermix areas identified as those where natural vegetation dominates the area, rather than development (Radeloff, Hammer, Stewart, et al., 2005; Stewart, Radeloff, Hammer, & Hawbaker, 2007). Both Radeloff et al. (2005) and Stewart et al. (2007) identified WUI as those areas where housing unit density was greater than one per 40 acres. If greater than 50% of the census block used to calculate housing density was made up of wildland vegetation, the area was classified as intermix WUI. Census blocks that were not intermix WUI, and were within 2.4 km of wildlands, were defined as interface WUI (Radeloff, Hammer, Stewart, et al., 2005; Stewart et al., 2007). The two methods of delineation differ in that Radeloff et al. (2005) included an additional parameter of a minimum patch size when assessing exposure to wildland vegetation (Radeloff et al., 2005). Theobald and Romme (2007) used a similar method, but criticized Radeloff et al. for differentiating intermix and interface WUI based on vegetation density. Rather, in Theobald and Romme's (2007) study population and housing unit density differentiate the two classifications of WUI. Theobald and Romme (2007) also removed protected lands from census blocks before calculating housing unit density. They then identified intermix WUI as a census block intersecting wildland vegetation, with a density of 1 to 2.5 housing units per 40 acres. Those census blocks that intersected wildland vegetation with a density of greater than one unit per 0.3 acres were delineated as interface WUI, consistent with the Federal Register definition (Theobald & Romme, 2007). Theobald and Romme (2007) then divided the WUI into hazard classes based on local fire regimes and forest fuels.

These models were not adopted for use in this study primarily because of the small, countrywide map scale for which they were developed, making them reliant on proxies for risk, and low on detailed spatial information. These models defined WUI as areas of a certain dwelling density, in intersection with wildland fuels, and do not account for topographic and site effects on wildfire risk, such as slope and the role of different fuel types in influencing wildfire behaviour (Forestry Canada Fire Danger Group, 1992; Johnson, 1992). Using proxy-based models, census data boundaries are the basic unit for delineating WUI, so WUI edges may represent a political boundary, rather than a shift in actual wildfire risk. Despite these concerns, these models are appropriate for management of a large area, and for the deployment of the Federal Register funding program. In the HRM, more detailed spatial data about the location of houses, housing density, and landscape and fuels are available, allowing us to model WUI within census tract boundaries.

The above models also use thresholds to identify WUI, such as requirements of a certain density of dwelling units, or that a minimum percentage of a census tract be vegetated. These thresholds may arbitrarily exclude some structures or areas from the WUI, ignoring the fire susceptibility of that specific site. The thresholds for unit dwelling density generally originate in the United States Federal Register government-funding program, which is not applicable within Canada. While these models are suitable for WUI delineation at a small scale they can be further refined for municipal management purposes.

1.3.2 BUFFER-BASED MODELS

Identifying WUI using distance buffers to represent the proximity of ignition sources to structures is common in more recently developed WUI delineation models. These models typically identify WUI by buffering dwellings at a series of distances representing the exposure to different ignition sources (Beverly, Bothwell, Conner, & Herd, 2010; National Institute of Standards and Technology [NIST], 2012). Another approach buffers homes at distances in which fuel removal is recommended to reduce wildfire risk and examines the contents of these zones (Lampin-Maillet & Bouillon, 2011).

While these models have the use of buffers in common, they employ them in different ways to identify WUI. Beverly et al. (2010) identified WUI in study communities in Alberta by buffering structures at: a) 0.1 to 30m, representing the distance at which radiant heat may ignite a home; b) 0.1 to 100m, representing exposure to short range wildfire spotting; and c) >100 to 400m representing the distance a firebrand or spark may travel, igniting a roof. WUI was classified as WUI I, II, III and IV depending on how many ignition processes the structure was exposed to, and the severity of this exposure. The National Institute of Standards and Technology (NIST) (2012) developed a WUI hazard rating system, which is similar to the method used by Beverly et al. (2010). The NIST model rates the level of hazard in areas of fringe development based on the level of exposure to ignition by fire or embers, calculated using the proximity of wildland fuels. Introducing scales of topography and wind exposure to determine hazard severity further refines this method (NIST, 2012). This model considers topography and weather, but limited detail is available about the method at present. Lampin-Maillet and Bouillon (2011) also identify WUI by buffering structures in fire sensitive areas at a distance of 100m, according to the prescription of a local brush clearing law in the region analyzed. Within the buffer, housing types are characterized based on the density of development. These areas are then contrasted with three classes of vegetation aggregation, characterizing WUI using the various combinations of development types and vegetation aggregation values (Lampin-Maillet & Bouillon, 2011).

The buffering models were also not adopted for use in this study, in part due to how some generalize complex natural processes using buffer distances. For example, Beverly et al. (2010) based their long-range firebrand spotting buffer value of 500m on data that included distances ranging 400 to 1600m. Despite this concern, these models are better suited for WUI modelling in small regions than proxy-based models. They often considered the local fuel composition and, in some cases, topographical effects on wildfire behaviour. Although the buffer models are more refined than proxy models, they do not use wildfire behaviour and susceptibility simulation tools, which can model fire behaviour using site-level inputs, such as *Burn-P3* (Canadian Forest Service, 2012) and *FARSITE* (USDA Forest Service, 2010b). As is the case with proxy-based models, the highly detailed spatial data available in the HRM is more refined than that used in the buffer models. Further, buffer models may still exclude relevant high-risk areas that fall outside of buffers, much like the exclusionary thresholds used in proxy models.

1.4 PROBLEM STATEMENT

In the context provided above, this study seeks to answer the question: what is the current and future wildfire risk in the AFR WUI?

To answer this question I have set the following goals and objectives:

1) Develop a model to identify WUI for municipal management and delineate WUI in AFR case study communities.
    a) Develop a transferrable model to identify WUI areas at a municipal scale, without using proxies for hazard or arbitrary thresholds.
    b) Delineate WUI in two peri-urban study communities using this model.
    c) Generalize lessons for WUI delineation in other settings.
2) Examine future wildfire risk in the AFR WUI under climate change.
    a) Develop a systems model of the impacts of climate change on wildfire risk drivers in the AFR, and their interactions with one another.
    b) Weight the relative importance of drivers of wildfire risk to determine the net impact of climate change on fire risk.
    c) Examine potential future fire susceptibility under altered landscape and climate conditions using scenario modelling.

1.5 METHODS AND MODELS

To address the problem statement above, this research takes a mixed-methods approach, combining spatial analysis, fire behaviour modelling, climate change projection, and a modified Analytical Hierarchy Process (AHP) expert weighting exercise. Each subsequent manuscript chapter includes its own comprehensive methods section. An overview is provided here, as well as some additional detail not appropriate for the manuscripts those chapters represent.

The majority of this research is spatial in nature, using raster and vector data in *ArcGIS 10.0* (ESRI, 2011). Spatial analysis for this research was conducted in the two HRM communities of Spryfield and Beaver Bank. Spryfield and Beaver Bank were chosen as case study communities due to their location at the urban fringe of Halifax, their proximity to wildlands, the ongoing suburban development in these areas, and their identification and delineation as communities in the HRM UFMP (Greater Halifax Partnership, 2007; HRM, 2011b, 2012). As spatial analysis was not conducted outside of these communities, this research should be considered a case study. The conclusions of this research were drawn from these sites and are thus transferrable to a limited extent. Despite this limitation, the models developed, and results related to climate change projections and the AFR system are more broadly applicable.

The Nova Scotia Department of Natural Resources (NSDNR) and the HRM produced the *ArcGIS* shapefiles used in this research. Remote sensing data were provided by the HRM or purchased from *DigitalGlobe*. Remotely sensed data were used to identify a real forest edge in the study communities. The NSDNR Forest Fuel Code (FFC) layer was then adapted to reflect this edge. The modified FFC layer was used in the *Burn-P3* fire behaviour model to determine current fire susceptibility (FS) by identifying a natural break as a threshold to delineate the WUI. These models and methods were used to meet the first goal of this research, to determine the current wildfire risk in the HRM WUI (Figure 1.5, Goal 1).

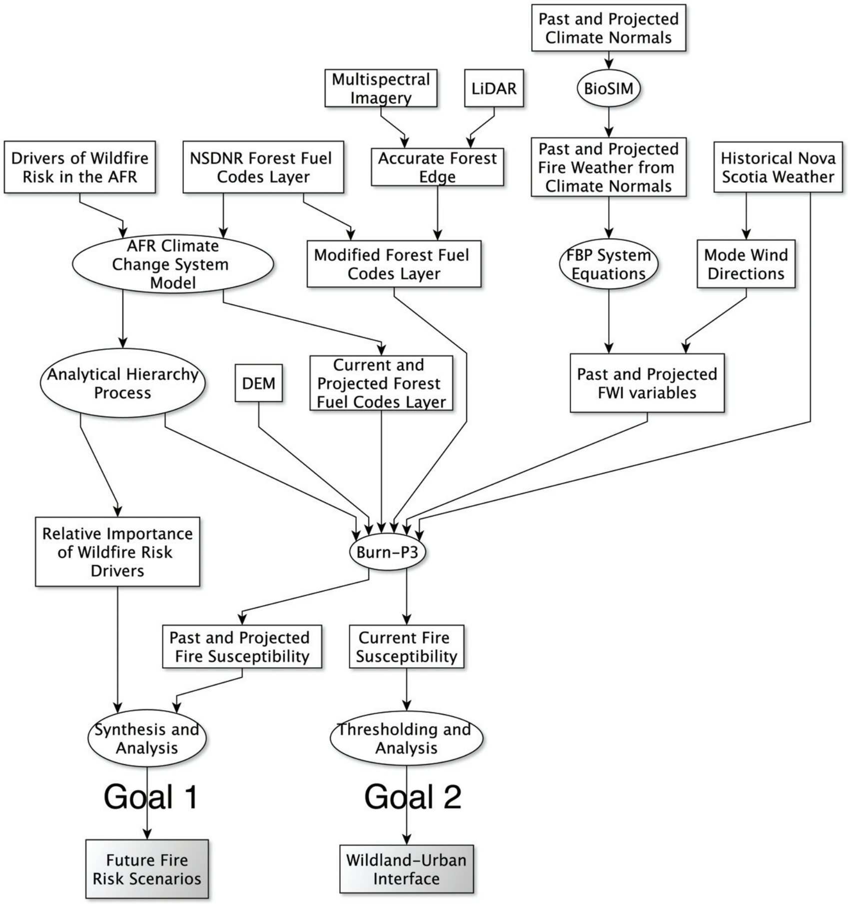

Figure 1.5 Relationship between the goals of this thesis, and models, methods, and data uses. Models and analysis are represented as oval, while data inputs and outputs are rectangular.

To examine future fire risk, downscaled projected climate normals from the Canadian Forest Service were used in the *BioSIM* model (Canadian Forest Service, 2013) to project future fire weather at Shearwater, Nova Scotia. A systems model of the impacts of climate change on wildfire risk drivers was created and used to structure three future forest scenarios. The FFCs reflecting these scenarios were developed in *ArcGIS*, and used in concert with the projected fire weather from *BioSIM* to project future FS in the AFR, using *Burn-P3*. The relative importance of AFR wildfire risk drivers was determined by expert opinion, solicited through the AHP process. These models and methods were used to meet the second goal of this research of examining future wildfire risk under climate change (Figure 1.5, Goal 2).

Through the process described above using these interrelated models, methods, and data I have assessed current and future wildfire risk in the HRM. A detailed technical discussion of methods, modelling, software and the data used to complete this research follows.

1.5.1 ANALYTICAL HIERARCHY PROCESS (AHP)

An AHP was used to determine the relative importance of wildfire risk drivers, by eliciting weights from wildfire experts. Our application of this method was modelled after that detailed in Nyerges and Jankowski (2010). An initial list of Canadian wildfire experts was drafted. These experts were contacted by email using the script included in Appendix B. Those who agreed to participate were asked to recommend other participants, using a snowball sampling method.

In an online *Opinio* (ObjectPlanet, 2012) weighting exercise participants were presented with pairs of seven drivers of wildfire risk, and asked to indicate if the two were equally important, or if one was more important than the other (Appendix B). The fire risk drivers assessed by experts using the AHP were: (a) Precipitation, (b) Coarse woody debris, (c) Temperature, (d) Decomposition rate, (e) Length of fire season, (f) Local tree species composition, and (g) Weakened tree stands on the landscape (full definitions included in Appendix B). If the participant indicated that one driver was more important, he or she was then presented with a verbal scale with eight options ranging from “slightly more important” to “extremely more important” to rate the relative importance of the more important driver (Table 1.1).

Participant responses were recorded and the verbal scale of importance was converted into numerical values ranging from 1 (equal importance) to 9 (extremely more important) (Table 1.1). Responses were entered into a matrix where the importance of driver A to driver B was equal to the inverse of the importance of driver B to driver A (Equation 1.1). Weights of relative importance out of one were calculated for each driver, using the method of weight calculation developed by Thomas Saaty for pairwise comparison, outlined in Nyerges and Jankowski (2010).

Table 1.1 One to nine interval scale of importance used in the Analytical Hierarchy Process weighting exercise (Nyereges & Jankowski, 2010).

1	Same importance
2	Slightly more important
3	Weakly more important
4	Weakly to moderately more important
5	Moderately more important
6	Moderately to strongly more important
7	Strongly more important
8	Greatly more important
9	Absolutely more important

$$\begin{array}{ccccccc}
1 & d_{12} & d_{13} & \dots & \dots & \dots & d_{17} \\
1/d_{12} & 1 & & \dots & \dots & \dots & D_{27} \\
1/d_{13} & & 1 & & & & \\
\dots & & & 1 & & & \\
\dots & & & & 1 & & \\
\dots & & & & & 1 & \\
1/d_{17} & 1/d_{27} & \dots & \dots & \dots & \dots & 1,
\end{array} \tag{1.1}$$

Where D is the reciprocal and square pairwise comparison matrix.

The pairwise comparison matrix columns were normalized. Averages of each row of the normalized matrix represent the eigenvector, or relative weight of that driver. The Consistency Ratio (CR) of the matrix was calculated to examine whether a participant's responses were internally consistent (Equation 1.2) (Nyerges & Jankowski, 2010). CR is a product of the Consistency Index (CI) of the matrix and the standardized Random Index (RI) value for the number of drivers being compared. A CR of <0.1 indicates that the matrix is non-random. CI is equal to the average value of the matrix consistency vector (λ), less the number of drivers being compared, over the number of drivers minus one

(Equation 1.3) (Nyerges & Jankowski, 2010). Finally, driver weights of relative importance were averaged across all participants, generating a single weight out of one for each fire risk driver.

$$\mathrm{CR} = \mathrm{CI}/\mathrm{RI} \tag{1.2}$$

$$\mathrm{CI} = \lambda\text{-}n/n\text{-}1 \tag{1.3}$$

1.5.2 REMOTE SENSING

Remote sensing data were used to identify a detailed local forest edge. Raster analysis of remote sensing data was performed using raster calculator in *ArcGIS*. Remote sensing data used in this study included Light Detection and Ranging (LiDAR) surface height data, as well as multispectral imagery from the *QuickBird* satellite (more detailed information about data used is included in Appendix C, C.1). Some *QuickBird* imagery required further orthorectification using rational polynomial coefficients in the program *ENVI* (Exelis Visual Information Solutions, 2012). This imagery includes multiple spectral bands of information, representing both the visible spectra and portions of the near-infrared (NIR) in the electromagnetic spectrum (Jensen R., Gatrell, & McLean, 2007). Multispectral *QuickBird* satellite imagery was used to calculate a Normalized Difference Vegetation Index (NDVI) to locate vegetated areas (Equation 1.4). NDVI is calculated by dividing the difference in the NIR reflectance band and the red reflectance band by the sum of these two bands (J. Jensen, 2000).

$$\mathrm{NDVI} = (\mathrm{NIR} - \mathrm{Red})/(\mathrm{NIR} + \mathrm{Red}) \tag{1.4}$$

NDVI values range from -1 to 1, with a value of 0 representing no leaf cover, and dense vegetation represented by values of greater than or equal to .5, distinguishing between vegetated and non-vegetated areas (Carlson & Ripley, 1997; Myneni, Hall, Sellers, & Marshak, 1995; Weier & Herring, 2011).

The NDVI was used with LiDAR data to identify vegetation heights. LiDAR is an optical remote sensing system that uses returns of NIR laser light to calculate the distance from the sensor to the earth's surface (Figure 1.6) (R. Jensen et al., 2007, p. 10). To generate LiDAR, pulses of light are emitted from a laser transmitter mounted on a vehicle, such as an airplane. A receiver uses the return time delay of the reflected pulse to measure distance. Each point collected through the LiDAR system is precisely located using a global positioning system and has X,Y, and Z values (Figure 1.6) (Chang, 2006). Multiple return values from a single pulse may be collected as the light encounters, and reflects off of surfaces. For example, a LiDAR pulse may encounter and reflect off of a tree branch (a first return) and continue on to produce a return from the earth's surface (last return). Multi-return LiDAR can then be processed to determine the height of objects on the earth's surface, the internal structure of objects, or the bare earth (Figure 1.6) (Greenfield & Guay, 2000; J. Jensen, 2000). A LiDAR-generated Digital Surface Model (DSM) represents the height of every surface of the area flown for LiDAR, including the heights of trees and buildings as well as the ground. A Digital Elevation Model (DEM) represents elevation of the ground level surface only, in what is sometimes called a bare earth model. By obtaining the difference between the DSM and the DEM I derived a Normalized Digital Surface Model (nDSM) (Equation 1.5) (Demir, Poli, & Baltsavias, 2008; Greenfield & Guay, 2000; Priestnall, Jaafar, & Duncan, 2000).

$$\text{nDSM} = \text{DSM-DEM} \tag{1.5}$$

The nDSM represents heights of objects that are above ground level, such as buildings and trees, and assigns a height value of 0 to ground-level surfaces. (Demir et al., 2008; Priestnall et al., 2000). Tall areas that intersect with NDVI values representing dense vegetation indicate tree heights and locations (Demir et al., 2008; Waser, Eisenbeiss, Kuechler, & Baltsavias, 2008).

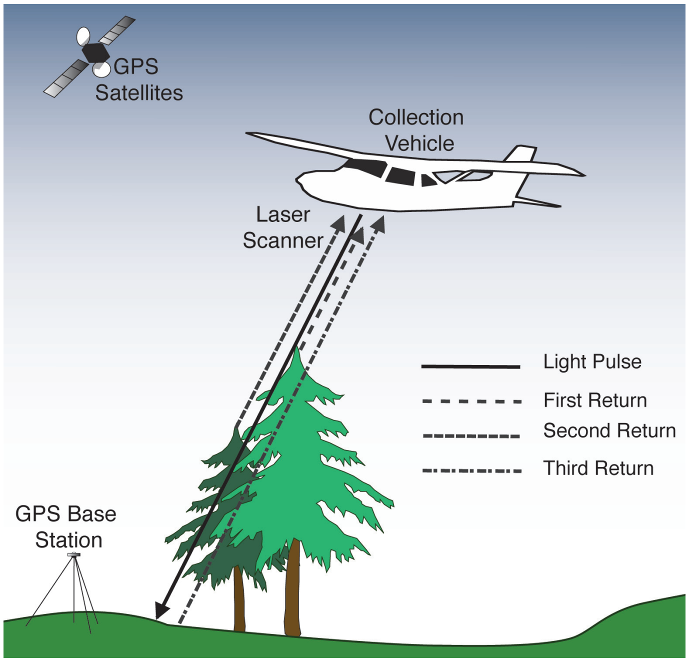

Figure 1.6 Representation of a Light Detection and Ranging (LiDAR) system.

1.5.3 FIRE BEHAVIOUR MODELLING

Fire behaviour modelling software provided simulated fire behaviour, allowing me to examine the relative susceptibility to wildfire across a study site. Fire behaviour modelling in this study was performed using the *Burn-P3* model, developed by the Canadian Forest Service and partners (Canadian Interagency Forest Fire Centre, 2011a). *Burn-P3* iteratively simulates fires over a landscape to estimate fire susceptibility, using the equations of the Canadian Forest Fire Behaviour Prediction (FBP) System to model fire behaviour (Canadian Interagency Forest Fire Centre, 2011a). The FBP System is a subset of the Canadian Forest Fire Danger Rating System, which also encompasses the Canadian Forest Fire Weather Index (FWI) System. The FBP System uses mathematical equations to model fire behaviour and produce quantitative estimates of head fire spread rate, fuel consumption, fire intensity, and fire descriptions, using an elliptical fire growth model (Forestry Canada Fire Danger Group, 1992).

Burn-P3 uses inputs of: (a) real and simulated FWI fire weather variables (e.g. fine fuel moisture code, initial spread index, and buildup index); (b) a DEM representing slope, aspect, and local topography; (c) FFCs representing 16 major fuel complexes and their associated fire behaviours; and, (d) user-determined inputs, such as historical ignition distributions, ignition locations, and numbers of fires to simulate (Canadian Interagency Forest Fire Centre, 2011b; Hirsch, 1996). In this study *Burn-P3* was used to produce raster outputs of fire susceptibility (FS). The FS of a raster cell in the study area modelled is the proportion of modelled burns that ignited a given cell, as a percentage (Equation 1.6). Two thousand ignitions were simulated for each model run in the study communities. This number was selected using preliminary model runs to identify at what point the study area had burned entirely, creating a continuous scale of FS.

$$\text{FS} = \text{Number of times a raster cell burned/ number of total ignitions} \times 100\% \qquad (1.6)$$

FS is similar to the measure of fire hazard called Burn Probability (BP). BP, however, is calculated out of the total number of model iterations, each iteration drawing from a distribution of the potential yearly number of ignitions in the region. Available ignition distribution data encompassed the entire province of Nova Scotia, and could not be modelled meaningfully in the study areas. FS is only comparable within an individual study area, as it is a relative percentage within the modelled region. The numbers, therefore, only indicate relative risk within that area, rather than the overall likelihood comparable between study sites.

1.5.4 CLIMATE MODELLING

Climate normals and modelling of FWI values under climate change were used to examine the influence of climate change on future fire weather over time. Climate normals "... summarize or describe average climatic conditions of a particular location" (Environment Canada, 2013). Past and projected climate normals were obtained from the Canadian Forest Service. Projected normals were developed from climate normals generated and supplied by Ouranos (R. St-Amant, Personal communication, July 19, 2012), a Canadian consortium of climate scientists and professionals (Ouranos, 2013). These projected climate normals under future climate change were developed by Ouranos using the Canadian Regional Circulation Model (CRCM) 4.2.0 (Music & Caya, 2007) at a 45 km grid resolution. The runs of the CRCM were driven by the 3rd generation coupled global climate model, using the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios A2 high GHG and aerosol emissions scenario (Nakicenovic et al., 2000; R. St-Amant, Personal communication, July 19, 2012). Downscaling was performed using the delta method and averaging of CRCM 45 km grid cells for the nearest weather station (R. St-Amant, Personal communication, July 19, 2012).

Modelling of FWI values under future climate conditions was performed in the *BioSIM* model. The Canadian Forest Service developed *BioSIM* primarily for modelling of forest pest outbreaks, however, recent updates added an FWI module, expanding the model to simulate current and future FWI using climate normals (Régnière, Saint-Amant, & Béchard, 2012). *BioSIM* modelling of future fire weather used inputs of climate normals ranging from 1961 - 1990 to 2071 – 2100, each normals data set spanning 30 years. The FWI module in *BioSIM* was set to begin producing fire weather data three days after snowmelt if >75% of the days in January and February had snow, and the maximum snow depth was at least 10 cm. Otherwise, the fire season modelling began following three consecutive days where noon temperature was >12ºC. The model was run for 100 replications for each set of climate normals. This generated daily precipitation, temperature, FWI values and other associated fire behaviour outputs from the FBP System for the study area. Microsoft Excel was used to refine the data produced in *BioSIM*, removing null values and performing statistical analysis. Equations of the FBP System were then used with this data to produce additional fire weather variables required for consistent fire behaviour modelling in *Burn-P3* and selection of daily data for modelling, limiting the future fire weather modelled to high fire-intensity days (Canadian Forest Service, 2012; Forestry Canada Fire Danger Group, 1992).

Daily wind direction cannot be modelled by *BioSIM*, and is a necessary input to model fire behaviour in *Burn-P3*. As wind direction could not be derived from *BioSIM* outputs, and is not correlated to other variables produced by *BioSIM* (Ahrens, 2012) historical weather data from 1990 to 2012 provided by the NSDNR was used to calculate the mode (most common) wind direction on each day of the fire season. Mode wind directions were matched to *BioSIM* data by corresponding month and date during the Nova Scotia fire season.

1.6 STRUCTURE OF THE THESIS

This thesis was completed in a manuscript format with intention to publish. Because of this, each manuscript chapter stands alone and includes background and methods sections, in addition to those included here. Specifically, Chapter 2 Modeling fire susceptibility to delineate wildland-urban interface for municipal-scale fire risk management; and, Chapter 3 Projecting wildfire risk in the Acadian Forest Region under climate change, are stand-alone manuscripts. The manuscript format is also reflected in the language used in these chapters. For example, often the term “we” is used. While this thesis has consisted of my original research, the involvement, support, and co-authorship of the supervisory committee is acknowledged in this term. The prefaces to each manuscript chapter discuss my role, and acknowledge co-authors, as well as the status of each manuscript in publication or process.

CHAPTER 2 MODELING FIRE SUSCEPTIBILITY TO DELINEATE WILDLAND-URBAN INTERFACE FOR MUNICIPAL-SCALE FIRE RISK MANAGEMENT

Ellen Whitman was responsible for the writing and research of this manuscript. Eric Rapaport and Kate Sherren were thesis supervisors, and co-authors. This manuscript has been revised, re-submitted and is under review in the journal Environmental Management. Environmental Management requires US American spelling. Because of this requirement the language in this chapter differs from the rest of this thesis. Footnotes are included for the ease of the reader of this thesis, and are not part of the manuscript.

2.1 INTRODUCTION

Urbanization of the earth's surface is occurring at a rate equal to, or greater than that of population growth, with implications for human wellbeing and adjacent ecosystems (Seto et al., 2011). In 2011, for the first time, a majority of the global population lived in urban areas, reflecting the ongoing re-distribution of population from rural areas to denser population centers (United Nations, 2011). Development patterns in Canada have followed this trend of urbanization. By 1931, the majority of Canada's residents lived in urban centers with populations of 1,000 people or more, and by 2006 80% lived in urban areas (Bollman & Clemenson, 2006; Statistics Canada, 2012). Much of the urbanization in Canada is low-density suburbanization (Bollman & Clemenson, 2006; Seto et al., 2011; Turcotte, 2008). Suburbs are "residential districts with low densities that are located at, or near, the urban fringe," (Harris, 2004, p. 7). Supporting the demand for suburban housing and access to natural amenities in rural areas, expanding suburban developments are encroaching on surrounding rural lands and natural ecosystems (Radeloff, Hammer, Stewart, et al., 2005; Stewart et al., 2007; Vince et al., 2005).

Associated with suburban and rural development is an expansion of the Wildland-Urban Interface (WUI). Wildlands are natural areas consisting of forests, peatlands, shrubs, grasslands, tundra, or alpine tundra (Canadian Council of Forest Ministers, 2005).

WUI describes areas where urban development occurs adjacent to – and intermingles with – wildlands (Federal Register, 2001; Hirsch, n.d.; Radeloff et al., 2005; Theobald & Romme, 2007). Development in WUI areas is the cause of many environmental management concerns, such as fragmentation of natural landscapes, habitat loss, and biodiversity decline (Alvey, 2006; McKinney, 2002; Vince et al., 2005). In this research we examine WUI delineation for forest fuel and wildfire hazard management, rather than for general WUI management and identification.

Fire risk is the chance of a fire occurring, including the role of causative agents; while fire hazard is potential fire behavior, as determined by the fuel type and physical characteristics of an area (Hardy, 2005). Delineation of WUI is important for fire hazard management, as it describes, in part, areas where urban development and residents are at risk from wildfire. Delineating the WUI is essential for targeting fire hazard management activities, such as the modification of wildland fuels to reduce the likelihood of crown fires (Federal Register, 2001; Haight, Cleland, Hammer, Radeloff, & Rupp, 2004; Hirsch & Fuglem, 2006; Partners in Protection, 2011). The scale of such analysis determines its effectiveness. Typically, WUI is delineated at small map scales for federal, provincial or county-level risk management. An extent often modeled in WUI studies is the conterminous United States. This is such a large area that the level of detail examined is limited (Gering, Chun, & Anderson, 1998; Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007). Modeling fire risk over such large areas requires the use of proxies for risk, such as the assumption that the intersection of a specific density of human population or housing density, and natural vegetation, presents a fire risk. Areas with such an intersection are delineated as WUI, allowing managers to prioritize these settlements when allocating resources and making decisions. Many of these models stem from the United States Federal Register definition of WUI, which was created in part to help distribute funding (Federal Register, 2001). These models of WUI are appropriate for risk management planning and funding allocation over a large area and for the identification of WUI in general. These models can be used by local or municipal fire risk managers, in spite of the generalization of natural and built environment. Additional detail information regarding natural landscape, however, can improve the understanding of site-specific drivers of fire hazard, such as types of vegetation and fuels found in the transition zone between forest edge and the built environment.

The small-scale models described above often benefit from matching spatial scales, whereas the scales of municipal data do not match with regional forest cover databases or census data. Census data, used in many WUI models to identify populated urban fringes, typically consists of polygons that represent a dissemination area or census tract, with population and development data generalized within this area (Statistics Canada, 2007). Natural landscape features are rarely catalogued at anything finer than a regional, 1:10,000 scale. This includes forest and land cover mapping done at the provincial level, in the Canadian context (Nova Scotia Department of Natural Resources [NSDNR], 2012). Forest cover data do not identify buildings in areas where sparse development is intermixed with forests, and they are often years out of date. While census and land cover data are useful for analysis, housing, population, and remotely sensed vegetation data may be available at the municipal level at a finer level of detail, allowing for improved delineation of WUI. Our study is based in the Halifax Regional Municipality (HRM) of Nova Scotia, Canada (Figure 2.1). In this region, building footprints or civic address center-points are available at a large map scale (1:1000 to as small as land survey data (Halifax Regional Municipality [HRM], 2011a)). In addition, HRM invests in high-resolution remote sensing coverage, and carefully maintains and updates its planning databases (HRM, 2011c).

Scale mismatches are not the only challenges faced by municipal fire risk managers. Another challenge is the Boolean delineation of forest edge in regional forest cover layers, which assumes no forest in areas identified as urban, regardless of their tree cover. The boundary between forest and city is fuzzy, with trees, decorative shrubs, grasses, and outlying structures blending together in a complex gradient of fuels. It is important to reflect this transition zone in model inputs to generate a realistic estimation of fire risk.

Municipalities are increasingly responsible for risk management, however, few models for WUI delineation have been developed for use at a municipal level (Cooper, 2010; Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007; Winter, McCaffrey, & Vogt, 2009). Those models for small areas that do exist typically identify

WUI by creating buffers from forest fuels at distances typical of fire convection and spotting ignition processes, delineating WUI as developed areas within those buffers and thus at risk of ignition from surrounding fuels (Beverly et al., 2010; National Institute of Standards and Technology, 2012). In this paper we describe a method for delineating WUI for municipal fire risk management, using local fire susceptibility (FS) modeled from projected fire behavior, instead of proxies for fire risk at scales limited by census and land cover databases, or a buffer model.

2.2 CONTEXT

Suburbanization is the driving developmental force in Canada (Bollman & Clemenson, 2006; HRM, 2006; Turcotte, 2008), as well as in the HRM, which has been experiencing steady population growth since the 1970s (HRM, 2006; Statistics Canada, 2011). Recently, the HRM has failed to meet densification targets set out in the 2006 Regional Municipal Plan, with only 16% of growth taking place in urban areas, while suburban development made up 56% of growth in the municipality in 2011 (HRM, 2006; Our HRM Alliance, 2012). Low-density suburban development is pushing into wildlands and the Acadian Forest stands that surround urban Halifax, expanding the WUI and increasing the complexity of local fuels as development and natural vegetation intermingle (Hammer, Stewart, & Radeloff, 2009b; HRM, 2012; Our HRM Alliance, 2012; Theobald & Romme, 2007). Fires are a naturally occurring disturbance in Nova Scotia, due to occurrence of coniferous forest stands. Urban development in proximity to this forest thus carries an inherent risk.

The Acadian Forest comprises non-coniferous, coniferous, and mixed tree stands, including climax species of eastern hemlock (*Tsuga canadensis*), red spruce (*Picea rubens*), sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*) and yellow birch (*Betula allehaniensis*) (Nova Scotia Museum of Natural History [NSMNH], 1994a). Some of these are fire-selected species, such as pin cherry (*Prunus pensylvanica*) and jack pine (*Pinus Banksiana*) (Neily et al., 2011, 2007; NSMNH, 1994b, 1994c). Because coniferous species typically burn much more readily than non-coniferous species, susceptibility to wildfire and fire hazard in Nova Scotia is typically dependant on the species composition of local forests (Johnson, 1992; NSDNR, 2010). Researchers have modeled return periods for fires in Nova Scotian forests from 200 to 2000 years, depending on suppression activity (HRM, 2012; Lauzon, Bergeron, Gauthier, & Kneeshaw, 2006; Neily et al., 2007). Today the vast majority of wildfires in Nova Scotian forests are caused by humans, however, who are responsible for 97% of ignitions in the province on average, while natural ignitions due to lightning make up only 3% of the provincial total (NSDNR, 2011).

WUI areas in the HRM have a history of wildfire. Fires in the past often burned over large areas, associated with human activities in semi-rural areas (Loo & Ives, 2003; Shea, 2012; Wein & Moore, 1979). In recent years large human-caused fires in forest areas have evacuated thousands of residents, closed roads, destroyed property and twelve homes, and burned thousands of hectares in the municipality, costing provincial and municipal governments, homeowners, and insurance companies hundreds of thousands of dollars (CBC News, 2012; Fanning, 2010; Shiers, 2010; The Canadian Press, 2008). The impacts of these large and destructive fires on residents of the Municipality have raised awareness about fire risk in the local WUI, but little practical action has resulted, such as planting policies or locally relevant education campaigns.

2.3 METHOD

We took a case-study approach to identifying WUI for municipal applications, using two suburban HRM communities as cases (Figure 2.1). The case study communities were chosen because: (a) They exist in close proximity to Acadian Forest wildlands; (b) They are experiencing ongoing suburban development, suggesting WUI is actively expanding in these areas (HRM, 2006, 2011b); and, (c) They have well-defined community boundaries, recently developed for the HRM Urban Forest Master Plan (HRM, 2012).

Spryfield is a large, older community with a densely developed urban core surrounded by linear developments that spread outward, following roads (Figure 2.1; Table 2.1). Spryfield began to develop into subdivisions after WWII, and typical housing in Spryfield reflects this, consisting of single-detached houses on lots of a mean size of

.12 ha (Figure 2.2 a; Table 2.1). Suburban development continues today, but at a more moderate pace (Greater Halifax Partnership, 2007; HRM, 2012; Nova Scotia Community Counts & Statistics Canada, 2006; Teplitsky, LeClair, & Willison, 2006).

Beaver Bank is a newer and smaller suburban community than Spryfield, centered around a main road with suburban development spreading outward on either side (Figure 2.1; Table 2.1). Beaver Bank was established at the turn of the 19th century, with development escalating in the late 1970s. Rapid suburbanization in this community is ongoing (HRM, 2011b). Housing in Beaver Bank is typically large, single-detached homes on large lots of a mean size of .38 ha (Figure 2.2 b; Table 2.1). The contrasts between the two communities make Spryfield and Beaver Bank excellent cases for studying WUI and developing a transferrable model for delineating WUI in suburban North America.

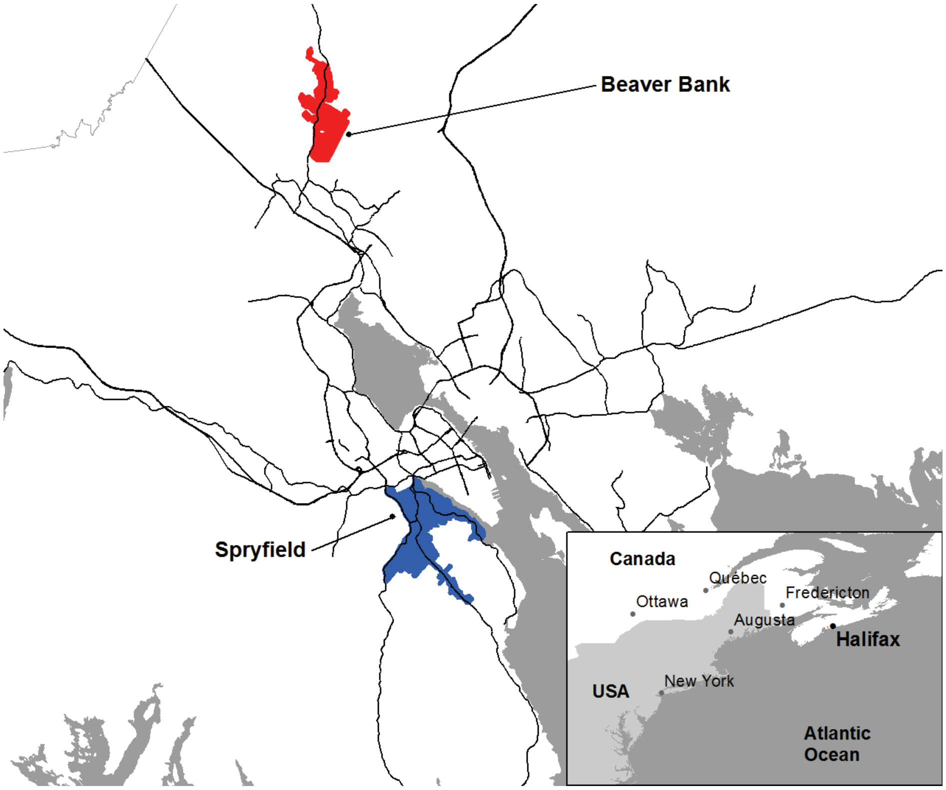

Figure 2.1 Beaver Bank and Spryfield communities in local Nova Scotian and North American context.

Table 2.1 Comparison of Beaver Bank and Spryfield study areas.

	Spryfield	**Beaver Bank**
Community study area size	1073.8 ha	613.6 ha
Number of residences	6653	1905
Number of parcels	5808	1232
Mean parcel size	0.12 ha	0.38 ha
% canopy cover within community boundary	87.9%	52.7%

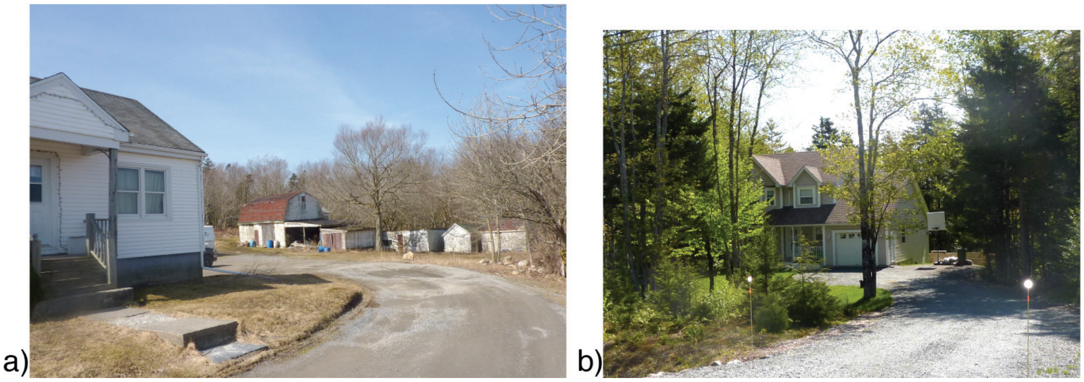

Figure 2.2 Typical wildland-urban interface housing patterns in a) Spryfield and b) Beaver Bank[1].

Within these study areas we used *Burn-P3* (Canadian Forest Service, 2012) to model FS, and thereby to identify and delineate the WUI. *Burn-P3* is a deterministic model endorsed by the Canadian Interagency Forest Fire Centre and developed by its members. *Burn-P3* uses the *Prometheus* common object model to simulate fire behavior based on the predictions of the Canadian Forest Fire Behaviour Prediction (FBP) System (Canadian Interagency Forest Fire Centre, 2011a). *Prometheus* is similar in design to the American fire growth model *FARSITE*. Both are deterministic, wave-propagation models that allow transitions from fire types (e.g. from ground to crown fires), and require similar inputs (Canadian Interagency Forest Fire Centre, 2011b; Mandel, Beezley, Coen, & Kim, 2009; USDA Forest Service, 2010a). *Burn-P3* iteratively simulates wildfire ignitions using: (a) Forest Fuel Code (FFC) maps; (b) a Digital Elevation Model (DEM);

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[1] For additional images see Appendix C, C.2.

(c) ignition grids; (d) weather station locations; (e) Fire Weather Index (FWI) values generated from those stations; (f) historical weather including the number of fire spread event days; (g) the number of total ignitions to be modeled; and, (h) fuel lookup tables that link FFCs with fire behavior (Canadian Interagency Forest Fire Centre, 2011a). FS is the number of times a raster cell burned during modeling, as a percentage of the number of total ignitions modeled.

FS = Number of times cell burned / Number of fires ignited in model x 100%

Briefly, we modeled the WUI by: 1. Modifying the Nova Scotia Department of Natural Resources (NSDNR) FFC layer to accurately represent urban fringes; 2. Using this updated fuel layer in *Burn-P3* to model FS; and, 3. Identified a threshold value of FS to delineate WUI[2]. Details of each step follow.

2.3.1 MODIFICATION OF THE FUEL LAYER

To create a detailed, spatially accurate FFC layer for use in *Burn-P3* we altered the provincial-scale NSDNR FFC layer. This GIS layer excludes urban fuels and generalizes forest edges (Figure 2.3 a). We refined this layer in *ESRI ArcGIS* to be accurate at a local level using Light Detection and Ranging (LiDAR) height data, and *QuickBird* multispectral satellite imagery obtained from the Municipality and *DigitalGlobe*. Briefly we: 1. Identified trees using LiDAR and multispectral imagery; 2. Expanded existing fuels into these treed areas using a logical model; and, 3. Added additional fuels, such as buildings and vegetation shorter than tree height that may conduct fire into the tree canopy (ladder fuels).

2.3.1.1 IDENTIFICATION OF TREES

QuickBird multispectral imagery was used to calculate a Normalized Difference Vegetation Index (NDVI), which identified vegetated regions of the study areas. This

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[2] A more detailed description of the data and GIS processes used in these methods is included in Appendix C, Section C.1.

index takes advantage of the characteristic “red edge” demonstrated by vegetation, between the red and near-infrared (NIR) wavelengths, the latter of which are invisible to the human eye. The NDVI indicates photosynthetic activity of vegetation (Defries & Townshend, 1994; J. Jensen, 2000).

$$\text{NDVI} = (\text{NIR} - \text{Red})/(\text{NIR} + \text{Red})$$

NDVI values range from -1 to 1, with NDVI values of ≥ 0.5 representing dense vegetation (Carlson & Ripley, 1997; J. Jensen, 2000; Myneni et al., 1995; Weier & Herring, 2011). Densely vegetated areas were further limited to trees, using the difference between first and second return LiDAR height data. We generated a Normalized Digital Surface Model (nDSM) by subtracting the DEM height values representing the earth’s surface, from the digital surface model (DSM) of all heights above and at ground level.

$$\text{nDSM} = \text{DSM} - \text{DEM}$$

The nDSM represents heights of features above ground level, with anything at ground level, such as grass, receiving a height value of 0m (Demir et al., 2008; Priestnall et al., 2000). Any surface feature height greater than or equal to 5m that intersects with dense vegetation NDVI values is considered a tree, a standard we applied according to the NSDNR Forest Ecosystem Classification (Neily et al., 2011).

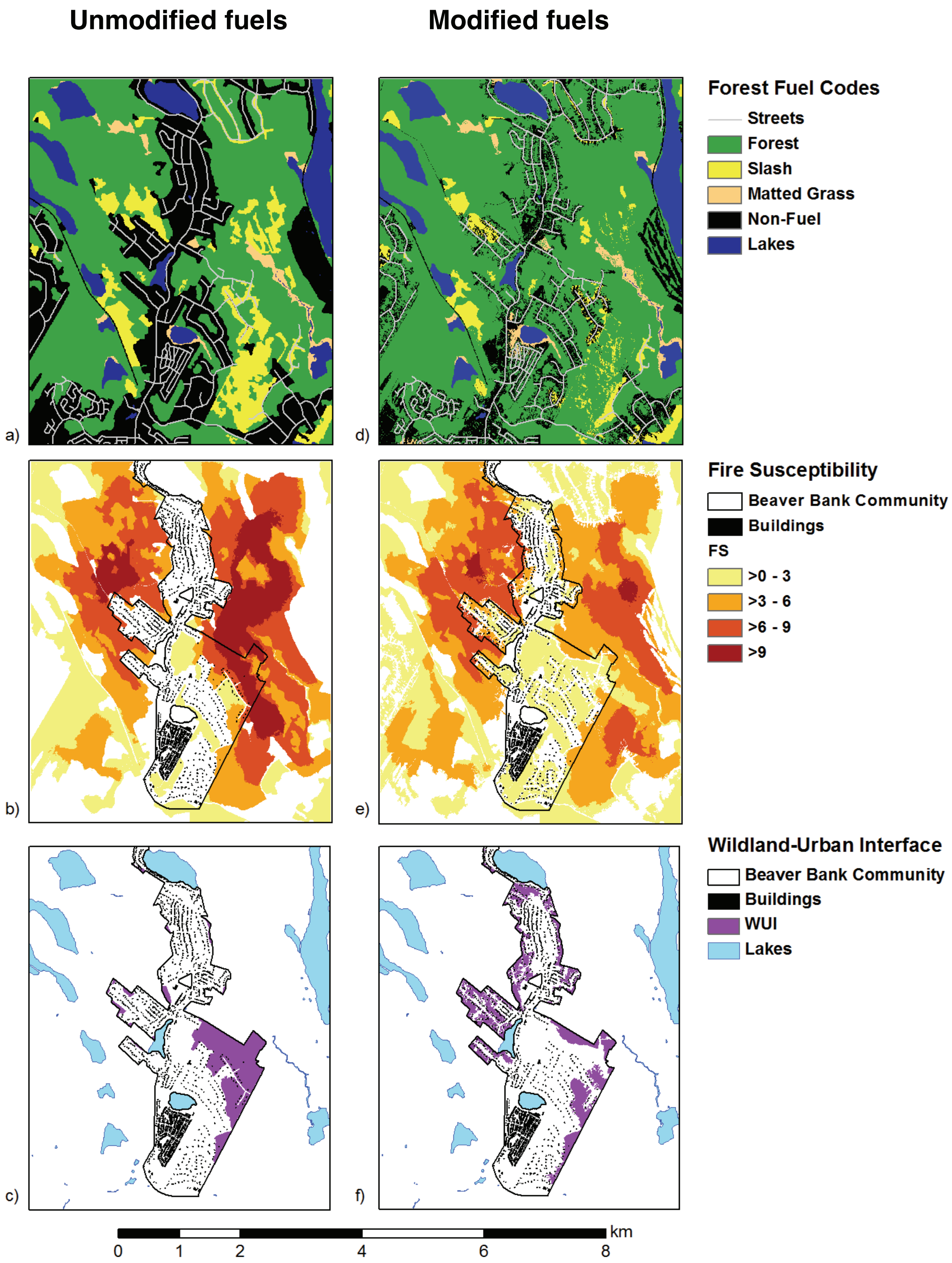

Figure 2.3 Beaver Bank study area; a) generalized unmodified, and d) generalized modified forest fuel codes; fire susceptibility distribution generated using b) unmodified, and e) modified fuel codes; and, wildland-urban interface delineation using fire susceptibility from c) unmodified, and f) modified fuel codes.

2.3.1.2 EXPANSION OF FOREST FUEL CODES

FFC data were obtained from the NSDNR. FFCs were expanded into the actual forest edges and other treed areas to assign fire behaviors to the trees identified in step 1. The provincial fuel layer was rasterized and its burnable fuel codes were expanded into the "urban" fuel code, which has no FBP System behavior associated with it, and could not have burned in *Burn-P3*. Cells identified as trees in the former urban fuel code were assigned that of the nearest tree fuel code. This expanded the forest edge inward to the actual edge, which was typically closer to the urban area. Conversely, cells that had received a tree fuel code in the provincial fuel layer, and that were not identified as trees using LiDAR and NDVI, were assigned a non-burnable fuel code, moving the forest edge away from the urban area to the actual forest edge. The remaining urban fuel code cells were assigned a non-burnable fuel code to reflect fire behavior on manicured urban lawns and impermeable surfaces. Treed areas that had previously had urban fuel codes were examined to determine if they were remnant stands. Remnant stands retained the FFC expanded from the nearest forest area. Urban treed areas that were not remnants and that fell within city blocks were assumed to be street trees and were assigned a non-coniferous fuel code, reflecting local municipal tree-planting practices. Areas of new development and clear cuts that had been harvested since the digitization of the FFC layer were included using actual edges identified using LiDAR and *QuickBird* Imagery, and assigned an appropriate fuel code, confirmed in part through field visits to selected sites[3]. Streets, parking lots, sidewalks and all other paved surfaces were obtained from the HRM geodatabase in polygon form and were added as non-fuel codes (Table 2.2).

2.3.1.3 ADDITION OF LADDER FUELS AND BUILDINGS

For both case study communities the final step in modifying the FFC layer was the addition of ladder fuels and buildings. Ladder fuels are fuels low in the forest structure, which allow fire to climb up from the surface into the tree canopy (Kurt & Scott, 2007). We identified ladder fuels as all raster cells with 'densely vegetated' NDVI values (Carlson & Ripley, 1997; J. Jensen, 2000; Myneni et al., 1995; Weier & Herring,

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[3] For additional photography and description of field visits see Appendix C, Section C.2.

2011), intersecting with nDSM LiDAR heights of ≥0.1 and ≤4.999m, i.e. shorter than tree height and taller than grasses (Neily et al., 2011). Taking a fuzzy logic approach (Hajek, 2010), the remaining ladder fuels were compared to a random grid of the same distribution of values. If a cell's ladder fuel height was greater than or equal to the corresponding random grid cell, the ladder fuels were then added into FFC areas that had been identified as non-fuel or grass. The fuzzy logic approach meant that taller fuels were more likely to be considered fuel. For instance, a quarter meter shrub has only a 5% chance of becoming a ladder fuel cell (.25/5), but a 2.5-meter shrub has a 50% chance (2.5/5). Ladder fuels were assigned a mixedwood 50% fuel code (Table 2.2), chosen to represent the mix of ladder fuels in natural and developed areas.

Buildings in the study area were also assigned a burnable fuel code to represent the fire risk associated with the presence of a structure. Buildings were given a fuel code of mixedwood 50% (Table 2.2), allowing buildings to ignite but giving them limited flammability. This code was assigned to represent fire behavior on the advice of wildfire and forest experts in attendance at several presentations of this work in spring 2012; no primary research was conducted to confirm this choice of fuel code.

2.3.2 FIRE SUSCEPTIBILITY MODELING

To examine the impact of the modification of the FFC layer and for sensitivity testing we modeled several FFC layers in *Burn-P3*. These layers were: 1. The "unmodified" provincial FFC layer; 2. An FFC layer with detailed forest edges, without the inclusion of ladder fuels; and 3. The "modified" FFC layer with detailed forest edges and ladder fuels added. The FFC layers, ignition grids, weather station grids, and DEMs for the study areas were converted into 5m x 5m text-based ASCII grids for use in *Burn-P3*[4]. Historical fire and weather station data were obtained from the NSDNR forest protection branch, while the DEM was provided by the HRM. Historical weather used in modeling was selected from previous high fire intensity days. The *Burn-P3* model was used to determine the number of times each cell ignited after simulating a total of 2000 ignitions in the study communities. Text-based ASCII grids of ignition totals for each

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[4] For a discussion about generalization of GIS data see Appendix C, Section C.1.4.

study area were generated by *Burn-P3* and converted into raster layers in *ArcGIS.* FS was calculated in *ArcGIS*, producing a continuous FS raster for each study area, beginning at >0. Field visits to high FS homes in both communities were completed to verify modeling and to confirm the presence of expected forest fuel codes in the area[5].

Table 2.2 Fuel codes of the Canadian Forest Fire Behaviour Prediction System.

FFC	Land cover
C-1	Spruce-lichen woodland
C-2	Boreal spruce
C-3	Mature jack or lodgepole pine
C-4	Immature jack or lodgepole pine
C-5	Red and white pine
D-1	Leafless deciduous
D-2	Green deciduous
D-1/D-2	Deciduous
SF	Nova Scotia seasonal fuel
S-2	White spruce/ balsam slash
S-3	Coastal cedar/ hemlock/ Douglas fir
O-1a	Matted grass
O-1b	Standing grass
NS1	Nova Scotia special
M-1	Boreal mixedwood – Leafless; Urban structure
M-2	Boreal mixedwood – Green
M-1/M2	Boreal mixedwood; ladder fuel; clearcut regrowth
101	Non-fuel; Urban areas; Roads
102	Water
104	Unknown

2.3.3 DELINEATION OF THE WUI

The FS of different study areas cannot be compared; the numbers are a function of the region's size and fuel mix. As such, it is impossible to select a threshold FS value that identifies WUI for all sites. The similarity of the histograms of FS values for the study areas, however, revealed a transferrable approach to identifying WUI. Histograms of FS for each community were bimodal (Figure 2.4). The trough between the two modes of the distributions fell at -1 standard deviation from the mean. We thus considered a cell WUI if it fell: (a) above that value, including only the higher mode, and (b) within the

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[5] An overview of field sampling is included in Appendix C, C.2.

community boundaries defined in the HRM Urban Forest Master Plan (2012) to exclude surrounding wildlands.

$$\text{WUI} = \text{FS} \geq (\bar{x} - \sigma)$$

This process delineates a discrete WUI area for use in municipal risk management.

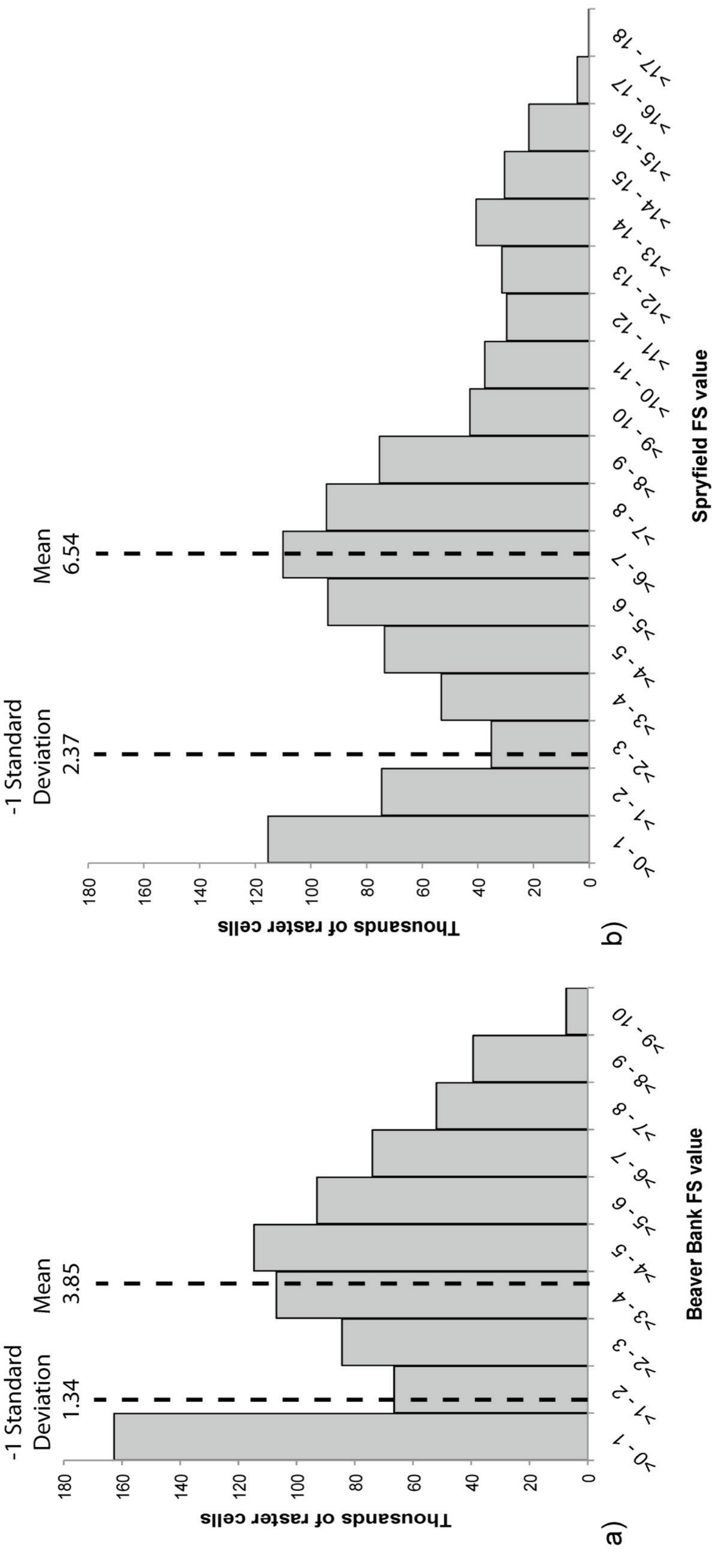

Figure 2.4 Bimodal fire susceptibility distributions, mean fire susceptibility values and standard deviations modeled from modified forest fuel code layers for a) Beaver Bank and b) Spryfield.

2.4 RESULTS

FS was modeled for each community using readily available layers and models, and intuitive pre-processing steps; and WUI was delineated using a consistent principle based on each community's FS values. The results were discussed with, and presented to, municipal risk managers, forest researchers and municipal environmental planners who assessed them as broadly accurate, reflecting expected fire behavior and fire risk.

Both the unmodified FFC layer provided by the province and the modified FFC layer with a forest edge generated using remote sensing data were used in *Burn-P3* to model FS[6]. After the modification of the fuel layer, more of each study area burned, indicating larger, and often very different, areas at risk from fire than would have otherwise been modeled, using provincial data. The majority of cells did not burn in either model (51.1%, and 67.2% respectively for Spryfield and Beaver Bank using the modified layer (Table 2.3)) indicating a low susceptibility to wildfire in the majority of urban areas studied, confirmed by the historical record.

2.4.1 SPRYFIELD RESULTS

Spryfield had a small increase in FS after modification of the FFC layer, and most of this was at low FSs, from >0 - 3. Highly susceptible areas decreased with improved fuel mapping (7.5% to 2.5%), although more structures were modeled as generally at risk, increasing from 14 to 43 (Table 2.3). The unmodified FFC layer produced different FS distributions than those of the modified layer (Figure 2.5). Both the modified and original FFCs left more than half of the community unburned, however more of the community showed some level of susceptibility after the improved fuel mapping (an increase of 4.2%). This was likely due to the improved forest edge and the inclusion of remnant stands, which allowed more fires to penetrate urban areas that had originally been digitized as devoid of any fuel (Figure 2.5).

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[6] For modified and unmodified FFC maps of the entire extent of the two study areas see Appendix D, Figures D.1 and D.2.

Table 2.3 Fire susceptibility distribution results for unmodified and modified forest fuel code layers.

		% area of community by FS value					
	FFC treatment	**0**	**>0 - 3**	**>3 - 6**	**>6**	**Percentage area WUI**	**Number of structures in WUI**
Spryfield area: 1703.8 ha	**Unmodified FFC**	71.4%	10.3%	10.8%	7.5%	11.3%	14
	Modified FFC	67.2%	22.2%	8.1%	2.5%	11.8%	43
Beaver Bank area: 613.6 ha	**Unmodified FFC**	72.1%	18.6%	7.3%	1.9%	17.9%	33
	Modified FFC	51.1%	36%	10.2%	2.8%	22.1%	34

Areas of the Spryfield community where FS was greater than or equal to 2.4 (-1 standard deviation from the mean FS value of 6.5 (Figure 2.4 b)) were classified as WUI (Figure 2.5 d). This represented 11.8% of the area within the Spryfield community boundary and encompassed 43 structures. WUI areas were located at fringes of the community, often where linear development had followed roads. Modeled fires did not penetrate the urban core. In contrast, the unmodified FFC Layer FS classified 11.3% of the community as WUI, and only 14 structures fell into this area (Figure 2.5 b). Fire risk does not appear to be a major consideration in development planning in Spryfield: new communities are being built, and development permits issued, in WUI areas of Spryfield where FS is elevated.

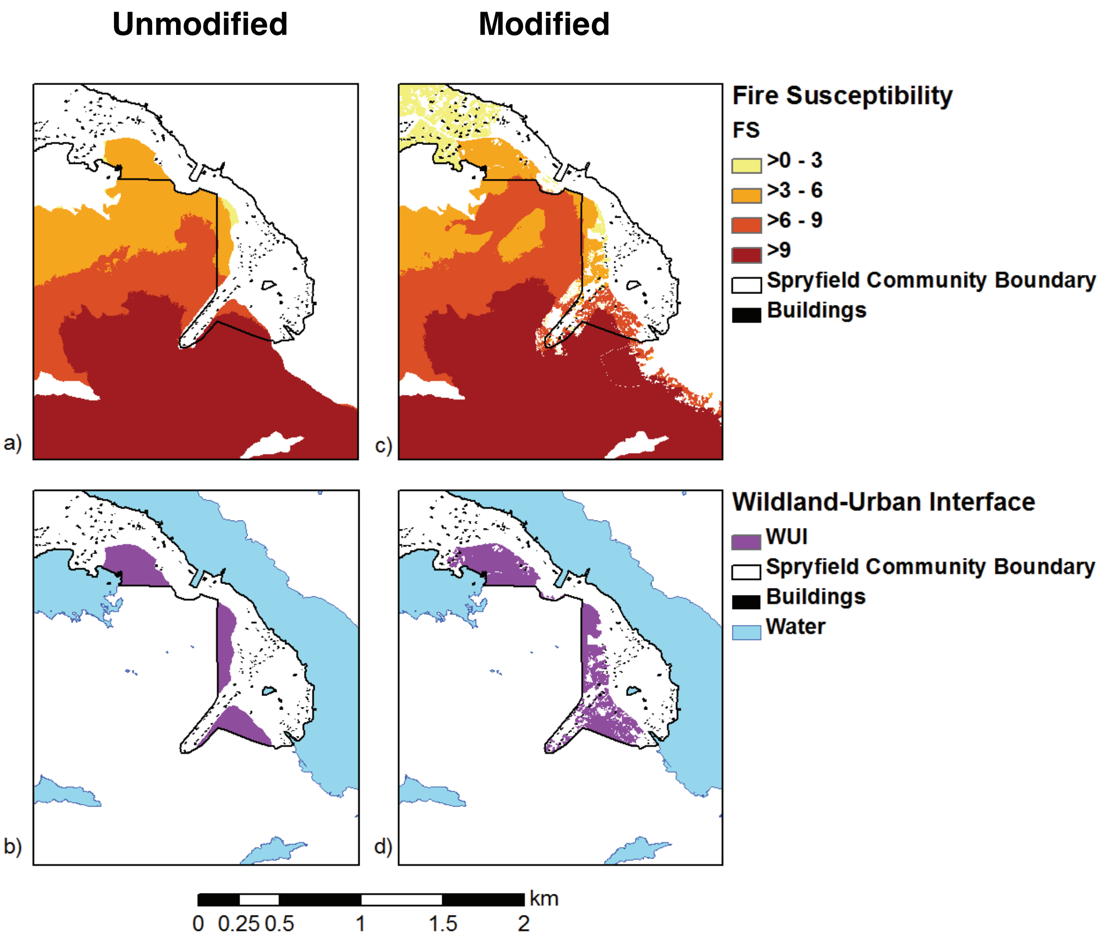

Figure 2.5 Subset of Spryfield results; fire susceptibility distribution generated using a) unmodified forest fuel codes, and c) modified fuel codes; and, b) wildland-urban interface delineation using fire susceptibility from b) unmodified, and d) modified fuel codes.

2.4.2 BEAVER BANK RESULTS

In the Beaver Bank study area the modified FFC layer produced a FS distribution for the entire study area with a mean FS cell value of 3.8. As in Spryfield, the majority of the Beaver Bank community area was not burned during the 2000 iterations. The modification of the FFC layer resulted in an increase of 128.6 ha in the area burned (Figure 2.3 b and e; Table 2.3). The generalized provincial delineation of the urban edge in Beaver Bank (Figure 2.3 a) in the unmodified FFC data made the majority of the community boundary unburnable. The modification of the FFC to represent the real forest edge and to add new development thus resulted in a much larger area of the community being burnable, albeit with a low fire risk (Table 2.3). Most of the cells with some degree of FS (36%) fell into the lowest FS category of >0-3% (Table 2.3). This low FS category also absorbed the majority of the increase in burned area, rising from 18.6% of the study area to 36%. Other FS categories also showed a moderate increase in area with the modification of the FFC layer.

Those areas of Beaver Bank where FS was greater than or equal to 1.3 (minus one standard deviation from the mean (Figure 2.4 a)) were classified as WUI (Figure 2.3 f). Using that threshold, 22.1% of the Beaver Bank study community was identified as WUI using the modified FFC layer. In contrast, only 17.9% of the study area was defined as WUI using the unmodified FFC layer to model FS (Figure 2.3 c). The WUI identified using the unmodified layer contained 33 buildings, and – despite the increase in WUI area - the modified FFC layer added only one additional building. Although these numbers are similar, most of the buildings in the two WUI delineations were entirely different structures; those from the unmodified layer were located primarily in new developments that had not been built at the time the provincial data were digitized. Because the provincial fuel data is not up-to-date, these buildings were modeled as entirely surrounded by forest with no defensible space, and this produced falsely high FS values. The 34 buildings identified in the modified FFC layer WUI were distributed in other forest edge areas that were generalized as urban in the provincial layer, which in fact were near to, and intermixed with forest edges. These two sets of WUI delineations would lead to significantly different management strategies and targeting.

2.5 DISCUSSION

The model described above identified fire risk under moderate fire weather conditions and WUI areas, without using assumptions or proxies for fire risk. The modified FFC layer produced results that more accurately reflected local fuel conditions in the peri-urban than the provincially delineated FFC layer, allowing municipal risk managers to examine fire risk at a fine scale. The modification of the FFC layer allowed for the incorporation of locally relevant data, such as forest stand composition, landscape effects on fire behavior, and weather.

2.5.1 MODELING

The modified FFC layer produced different FS distributions and levels of fire risk from those generated using the unmodified provincial layer. FS results using unmodified FFC layers improperly identified areas of new development as WUI, due to the age of the forest data. Because the unmodified FFC did not include new fuel breaks in the form of roads and lawns, the houses in developments that had been built since the digitization of the provincial FFC layer were modeled as having a high FS, as they were apparently entirely surrounded by forest with no defensible space. Further, the WUI identified using the modified FFC layer included burnable fuels, such as houses, that were previously considered urban and unburnable in the original provincial layer. The modification of the FFC layer altered the number of buildings that were located in the WUI, increasing the number of structures identified as at risk in both communities. By modifying the FFC layer for use in *Burn-P3* we created a more specific representation of urban wildfire susceptibility in the peri-urban area and a more meaningful delineation of community-scale WUI using FS.

Other models of WUI delineation do not incorporate ladder fuels on private property. Typical models rely on broad-scale land cover data, and do not fully account for the role of ladder fuels in fire hazard, despite their contribution to fire spread and crowning (Sturtevant, Miranda, Shinneman, Gustafson, & Wolter, 2012). By using a fuzzy logic approach, we allowed ladder fuels to be probabilistically incorporated into the fuel layer and to contribute to fire hazard without imposing a false dichotomy of fuel or non-fuel, based on vegetation height. Rather, ladder fuels were given an increasing chance of being considered a fuel with their height, reflecting the potentially varying contributions to fire hazard from ladder fuels (Hajek, 2010; Sturtevant et al., 2012). A comparison of the modified FFC layer and the modeled modified FFC layer without added ladder fuels indicated that the inclusion of ladder fuels in our model altered the FS distributions in both communities, and also increased the number of structures that were at risk from wildfire. For example, in Beaver Bank the addition of ladder fuels led to six additional buildings receiving an FS >0.

Areas that were identified as WUI using this model were: (a) Urban fringes where development spread outward into wildlands along roads in a linear fashion; and, (b)

Areas where an outer edge of development abutted wildlands, but shielded a denser urban core. These identified areas match the most common definition of WUI (areas where urban development occurs adjacent to – and intermingles with – wildlands (Federal Register, 2001; Hirsch, n.d.; Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007)), indicating that this model successfully identified WUI.

The modification of the FFC layer generated FS distributions indicating that fires did not penetrate into the urban core in either study area. Such behavior is expected when wildland fires encounter well-maintained lawns (Randall, Hermansen-Báez, & Acomb, 2009). In the Spryfield study area, regions delineated as WUI included roads and communities that were evacuated during recent wildfires (CBC News, 2012; Fanning, 2010), suggesting that the model has correctly identified those developed areas that are at risk from wildfire. Beaver Bank does not have a recent history of wildfire and contains many non-coniferous forest stands. Our model indicated that this community had a lower average FS than Spryfield and fewer homes were identified in WUI, reflecting historic fire trends and expected fire behavior based on local forest composition and development.

2.5.2 APPLICATIONS OF THE MODEL

Due to the fine-grained level of detail and large scale of this model, it is not well suited for federal or provincial risk management initiatives. Existing models that use proxies for risk to delineate WUI are appropriate and adequate for fire risk management and planning at a small scale, where a large area is analyzed and managed. Our model also does not examine the role of settlement characteristics (e.g. dense urban development or scattered rural homes) in WUI. The model described here is focused only on relative fire risk, limiting its applicability for examining other WUI management problems. Rather, this model is best suited for municipal risk management, as WUI and fire risk outputs are community-specific. FS values produced in this model are a percentage determined within the study area and excluding regions outside, so levels of risk within an area are defined relative to one another. The consistency of the FS distributions (Figure 2.4), however, provides a simple means to identify a meaningful threshold for these relative metrics for each site. This allows risk managers to prioritize specific areas that are at greater risk as compared to the rest of the community of interest, rather than using a fire risk layer intended for use over a large region, which generalizes and smoothes the appearance of FS. Given the building-level detail our process involves, however, the resulting maps may have to be carefully controlled to avoid harms, such as reduced property values.

The model also uses local fuel codes and topography (represented by a DEM) to identify WUI. This further refines fire hazard at a local scale by accounting for landscape effects on fire risk (for example, the role of slope in fire behavior) and stand-level detail in wildfire modeling. Using this model, risk is not generalized within political boundaries or identified using housing densities, and does not generalize natural vegetation as an inherent fire hazard independent of site-level species fuel complexes, as is the case with some existing models (Radeloff, Hammer, Stewart, et al., 2005; Stewart et al., 2007; Theobald & Romme, 2007). Our model also ignores edges that represent a change in housing density or a political boundary, rather than a shift in fire risk. The community-specific definitions of FS, and the use of landscape and stand-level detail in this model, make it useful for municipal risk and hazard management, and community capacity building.

The majority of this study involved the detailed delineation of forest edges using remote sensing data, allowing development, forest remnants and trees to intermingle. Despite the use here of expensive and detailed *QuickBird* multispectral imagery and LiDAR data, this model is still useful for those municipal managers who do not have funding for, or access to, such data. Aerial photography, such as USGS Digital Orthophoto Quarter Quads (DOQQs), may have enough detail for risk managers to refine broadly delineated FFC data. Representing forests and urban areas as a fuzzy edge with intermingling of urban and forest fuels is essential for proper modeling of WUI using FS. The intermingling of houses, outlying structures, and forest and ladder fuels at forest edges is truly characteristic of WUI, and represents an inherent elevated fire hazard.

2.5.3 LIMITATIONS

While this model is a strong starting point for WUI modeling using FS, it is limited by some generalizations and assumptions[7]. Buildings are represented using an FFC of mixedwood, 50% crown closure. To properly select a representative fuel code, further research is needed regarding fire behavior when wildfire encounters a structure, either to choose the most representative existing fuel code or to generate a new Canadian FBP System fuel code and associated fire behavior curve. There is also a limited understanding of how manicured urban lawns behave in fires. We chose to represent lawns as non-fuel, making them fuel breaks. While it is understood that lawns do significantly deter fire spread (Randall et al., 2009), they are an extensive fuel type in urban areas and require further research for accurate modeling using Canadian FBP System fuel codes.

2.6 CONCLUSION

We have created a replicable model for delineating community-scale WUI by modifying FFC data to accurately represent forest edges, using this modified layer to model FS, and using natural breaks to identify WUI threshold FS values without the use of proxies for risk. This method does not generalize fire risk within census boundaries and considers important community features such as landscape and local forest fuels when assessing wildfire risk. This model accounts for site-specific drivers of wildfire dynamics such as slope and fuels and uses a publicly available fire behavior model, in the form of *Burn-P3*. Fire risk, as represented by WUI, is modeled as relative within a community for risk management at a large scale. Due to the community scale of modeling and the inclusion of site-specific drivers of fire risk we have created a useful model for municipal-level fire risk assessment and management.

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[7] Additional discussion of generalization and assumptions is included in Appendix C, C.1.3.

2.7 ACKNOWLEDGEMENTS

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CHAPTER 3 PROJECTING WILDFIRE RISK IN THE ACADIAN FOREST REGION UNDER CLIMATE CHANGE

Ellen Whitman was responsible for the writing and research of this manuscript. Kate Sherren and Eric Rapaport were thesis supervisors and co-authors. This manuscript will be submitted to the journal Landscape and Urban Planning, or Regional Environmental Change. Footnotes are included for the ease of the reader of this thesis, and are not part of the manuscript.

3.1 INTRODUCTION

Future anthropogenic climate change should impact individual drivers of wildfire risk both directly and indirectly, but it is unclear how climate change will affect overall future wildfire risk in the Acadian Forest Region (AFR). Fire risk is the chance that a fire might start, as affected by local fuel and weather conditions, and causative agents. By contrast, fire hazard is the potential fire behaviour of a fuel type, regardless of ignition sources and fire weather (Hardy, 2005). To determine the severity of future wildfire risk in the AFR we employed a mixed-methods approach, combining the modelling of future forest conditions, climate and fire behaviour with a modified Analytical Hierarchy Process (AHP) with wildfire expert participants.

The AFR is a transitional forest zone between the northern boreal and eastern, primarily non-coniferous forests (HRM, 2012; Steenberg et al., 2011). This forest zone covers the Maritime Provinces and extends into Maine and Québec (Simpson, 2008). Due to the transitional nature of this forest zone it is characterized by mixedwood forests, with non-coniferous and coniferous tree species growing alongside one another. Typical climax tree species in this region include eastern hemlock (*Tsuga canadensis*), red spruce (*Picea rubens*), sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*) and yellow birch (*Betula allehaniensis*) (Nova Scotia Museum of Natural History [NSMNH], 1994a). Given this diversity, the local species composition is a strong determinant of wildfire risk at a given site, as coniferous trees are much more susceptible to wildfire (Hély et al., 2000; Johnson, 1992).

Global climate change is ongoing, and it is understood that change caused by anthropogenic emissions of greenhouse gasses (GHGs) will continue into the future, even if dramatic changes are made to GHG emission rates today (Intergovernmental Panel on Climate Change [IPCC], 2007). Using an average high emissions and climate change scenario developed from the Intergovernmental Panel on Climate Change (IPCC) A1F1 and A2 scenarios (Nakicenovic et al., 2000) Richards and Daigle (2011) projected future increases in both temperature and precipitation in the Atlantic Canadian AFR, over time (Figure 1.4). These shifts in the local climate will have impacts on both negative and positive drivers of wildfire risk in this region, altering future forest wildfire risk.

Due to the complex relationships between these wildfire risk drivers, and because of the transitional nature of the AFR zone it is unknown whether climate change will lead to an increase or decrease in future fire risk in this region. In this study we first examined the drivers of fire risk in the AFR under future climate change, using a systems model. This model guided our modelling of fire behaviour and forest composition under an altered climate, and thus fire risk as represented by fire susceptibility (FS), using three climate change scenarios. Finally, soliciting weights from wildfire experts allowed us to draw conclusions about the relative importance and net impact of fire risk drivers.

3.2 BACKGROUND

Climate change will have impacts on both positive and negative drivers of fire risk in the AFR. Changes to these drivers may increase or decrease future risk directly, and interactions amongst drivers may further accelerate changes to the AFR and to risk. An increase in temperature during the fire season, as is projected for the AFR (Figure 3.1 a) (Richards and Daigle, 2011), would act as a direct positive driver, increasing the future fire risk, as large fires require dry fuels and wind. These conditions are produced in periods of warm-dry weather (Johnson, 1992). An increase in local temperature would thus likely increase the area burned that is observed in Canadian forests and attributed to climate warming (Gillett et al., 2004; Maness et al., 2012). Furthermore, modelling of the length of the fire season under climate change projected an increase in fire season length across Canada (Flannigan & Wang, 2012; Wotton & Flannigan, 1993). This would act as a positive driver of future fire risk, as a longer fire season would offer more days where conditions allow ignitions to escalate into wildfires.

Increasing precipitation is also projected for the AFR (Figure 3.1 b). This increase would act as a negative driver of fire risk, as wetter fuels and frequent rain reduce wildfire risk. In Canada, the annual area burned is negatively correlated with the number of days with rain (Flannigan & Harrington, 1988; Johnson, 1992). Cary et al. (2006), however, projected an increase in area burned under a warmer climate, despite increased precipitation. While an increase in the volume of precipitation is projected for this region under climate change, the majority of this increase is expected in winter months, with rather limited increases and even decreases in precipitation projected during the summer fire season (Figure 3.1b) (Richards & Daigle, 2011). These complexities make the future influence of this fire risk driver unclear.

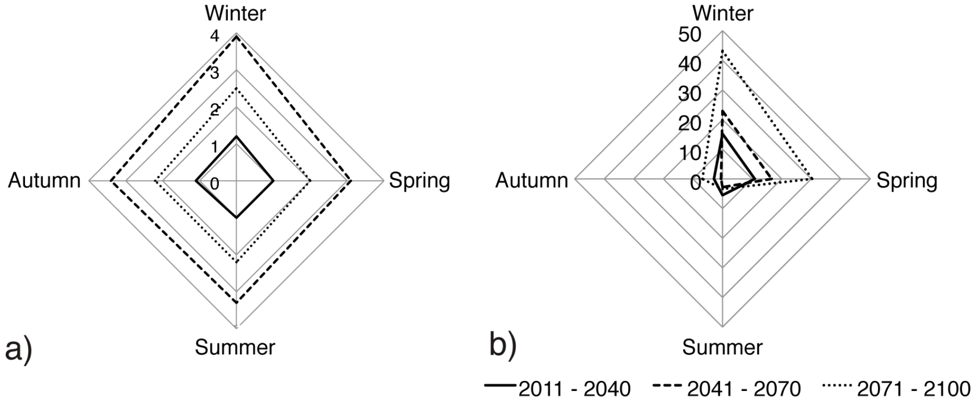

Figure 3.1 Projected a) Change in temperature (ºC) and b) Change in precipitation (mm) from 1980s baseline values for the HRM for 2020 (solid), 2050 (dashed) and 2080 (dotted). Data from Richards and Daigle, 2011.

The changing future climate will also influence tree species in the AFR, altering future fuels for wildfires. Modelling suggests that in the mixedwood AFR community a warmer wetter climate will favour non-coniferous southern species over boreal tree species (Bourque et al., 2010; Iverson et al., 2008; McKenney et al., 2007; Steenberg et al., 2013). Many of these boreal tree species are conifers, and as conifers are more susceptible to ignition during wildfires a transition to a primarily non-coniferous forest would act as a negative driver of future fire risk in this area (Hély et al., 2000; Johnson, 1992).

In the short to mid term, however, this shift in species may act as a positive driver of wildfire risk. It is likely that tree species spread to appropriate habitat will lag behind shifts in suitable habitat ranges, suggesting boreal species may remain in the AFR, despite maladaptation (Iverson & McKenzie, 2013; McKenney et al., 2007; Steenberg et al., 2013). Declining and maladapted trees are more susceptible to wind throw, conifers especially so (Foster, 1988a, 1988b; Johnston et al., 2009; Rostami, 2011). Hurricanes and tropical cyclones in this region are expected to increase in size, severity and pass closer to the Nova Scotia landmass (Jiang & Perrie, 2007). This would likely increase occurrences of wind throw and act as a positive driver of fire risk (Foster, 1988a; Jiang & Perrie, 2007; Liu et al., 2008). Wind thrown trees may become ladder fuels, conducting surface fires up into the canopy, and also contribute to the accumulation of coarse woody debris on the landscape, both of which would increase fire risk (Brown et al., 2003; Kurt & Scott, 2007; Liu et al., 2008). As coniferous species are more susceptible to wind throw and are more likely to burn in a wildfire, increasing storm events may expedite their removal from the landscape, opening up areas for colonization by non-coniferous tree species, eventually reducing future wildfire risk (Foster, 1988a, 1988b; Johnson, 1992).

This increase in coarse woody debris may be somewhat offset by decomposition, the rate of which is expected to speed up under warmer, wetter climates (Chambers et al., 2000; Fleming et al., 2002). Decomposition acts as a negative driver of fire risk, leading to a decline in flammability in the understory (Fleming et al., 2002; Johnston et al., 2009). Decomposition and wildfires release carbon into the atmosphere, however, potentially contributing to climate change, due to additional accumulation of GHGs (Chapin III et al., 2008; Fleming et al., 2002).

Changing seasonal temperatures and precipitation patterns in the AFR may also influence the frequency, severity and timing of pest and disease outbreaks. In general, warming temperatures are associated with increased metabolic rates in insects, increased vegetation consumption, accelerated development, and spread of forest pest populations. Warmer temperatures in the winter can also reduce insect mortality (Dukes et al., 2009). Insect outbreaks can lead to changes in local microclimate and create dry, dead fuels, acting as a positive driver of wildfire risk (Fleming et al., 2002; Maness et al., 2012). Outbreaks of spruce budworm (*Choristoneura fumiferana*) and the native spruce beetle (*Dendroctonus ruffipenis*) in the boreal forest and parts of the AFR are projected to increase under climate change (Bentz et al., 2010; Gray, 2007), driving future fire risk up (Bentz et al., 2010; Fleming et al., 2002; Gray, 2007; Taylor & MacLean, 2009). Invasive insects with tree hosts (such as the brown spruce longhorn beetle (*Tetropium fuscum*) in the AFR) are also expected to benefit from future climate change (Dukes et al., 2009). Over time, the biotic disturbances of Atlantic Canadian forests from insect pests and pathogens are expected to increase moderately (Johnston et al., 2009).

The altered future climate will also impact the susceptibility of trees to forest pathogens. Weakened, maladapted trees are more susceptible to disease and fungus than those that are healthy (Dukes et al., 2009; Johnston et al., 2009). Direct effects of climate change on tree pathogens may include: “(i) increased growth and reproduction; (ii) altered propagule dispersal, transmission rates, and infection phenology; and (iii) changes in overwinter survival” (Dukes et al., 2009, p. 236). Phenology, or life cycle timing, of insects and insect predators may also be modified by climate change. The disruption of synchrony between life cycle events of forest pests and plants or insects and their predators is a likely pathway for climate change to influence future forest composition and wildfire susceptibility (Johnston et al., 2009; Logan, Régnière, & Powell, 2003).

We developed a conceptual understanding of these drivers using systems modelling, which was synthesized in a detailed conceptual model of interacting AFR fire risk drivers under climate change[8]. The systems model represents the complex interrelationships between wildfire risk drivers under climate change. Drivers of fire risk

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[8] For the system model of Acadian Forest Region wildfire risk drivers under climate change described in section 3.2 Background, see Figure 3.6.

were identified as either a negative or positive driver, depending on how they influence other drivers and overall wildfire risk. These drivers were narrowed down into a list of the key elements for simulation and expert assessment.

3.3 METHODS

To assess the impacts of future climate change on fire risk in the AFR we took a mixed-methods approach. We: 1. Developed a conceptual understanding of the dynamics of fire risk as described in the background; 2. Modelled future fire weather and FS using *BioSIM* and *Burn-P3* (Canadian Forest Service, 2012, 2013); and, 3. Employed a modified AHP to elicit from experts average weights of relative importance, and confirm the net impact, of AFR fire risk drivers.

3.3.1 STUDY AREA

Our research is based in the Halifax Regional Municipality (HRM). This coastal area of the province of Nova Scotia is located within the AFR, and supports the majority of Atlantic Canada's population (Statistics Canada, 2012). The two HRM communities of Spryfield and Beaver Bank were selected as cases (Figure 2.1).

Forests composed primarily of coniferous tree species surround Spryfield. This area has a recent history of wildfire, which caused the destruction of homes and evacuations of residents in recent years (CBC News, 2010b; Fanning, 2010; Shea, 2012). Forests in Beaver Bank are composed of mixedwood and non-coniferous associations of tree species, with no recent history of wildfire. These two communities represent a range of tree species compositions in the AFR, allowing insight into site-specific future fire risk dynamics. Their location in the HRM wildland-urban interface has allowed us to inform urban planners in this area about projected fire risk[9], but our conclusions here are not limited to peri-urban areas of the AFR.

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[9] Chapter 2.

3.3.2 MODELLING

The AFR system and wildfire risk drivers described in section 3.2 Background were consolidated in a systems model. This model directed our choice of wildfire risk elements for spatial modelling. Changes to climate will impact fire weather, and, over time, influence the AFR tree species composition. Fire weather and tree species drivers were adopted in our scenario modelling of future fire risk, as these were primary drivers of fire risk and could be incorporated in fire behaviour modelling[10].

We used *BioSIM* to model future fire weather in the AFR, drawing on downscaled climate normals for Shearwater, HRM, Nova Scotia (Figure 3.2). Shearwater is the nearest weather station for which climate normals have been downscaled in this region, and is located approximately 20 km from Beaver Bank and 8 km from Spryfield. The Canadian Forest Service developed *BioSIM* for modelling forest pest outbreaks, behaviour, and populations as they relate to, and are driven by, climate variables. The *BioSIM* model was recently expanded to include a fire weather module, which was used in this study (Régnière et al., 2012). Climate normals represent the average weather and climate in a location (Environment Canada, 2013), over a span of 30 years. Past and projected climate normals were obtained from the Canadian Forest Service. Ouranos, a Canadian consortium of climate scientists and professionals (Ouranos, 2013), generated the projected climate normals. These projected climate normals were developed using the Canadian Regional Circulation Model 4.2.0 (Music & Caya, 2007) at a 45 km grid resolution. These runs were driven by the 3rd generation coupled global climate model, using the IPCC Special Report on Emissions Scenarios (SRES) A2, high GHG and aerosol emissions scenario (Nakicenovic et al., 2000; R. St-Amant, Personal communication, July 19, 2012). Mean climate normals were downscaled to the nearest weather station, using a delta method (R. St-Amant, Personal communication, July 19, 2012). Each climate normal set was used to simulate fire weather on typical days, using normals ranging from 1961 - 1990 to 2071 - 2100.

The Fire Weather Index (FWI) is a numerical rating of potential fire intensity, and is a proxy for fire danger used across Canada (Hély et al., 2000; Nova Scotia Department

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[10] For additional detail about models and methods see Chapter 1, Section 1.5 of this document.

of Natural Resources [NSDNR], 2013). *BioSIM* modelling produced daily precipitation, temperature, FWI values, and other associated fire behaviour outputs from the Canadian Forest Fire Behaviour Prediction (FBP) System, using climate normals (Figure 3.2). Microsoft Excel was then used to refine and summarize the datasets produced in *BioSIM*, removing null values and performing basic statistical analysis. Equations of the FBP System were further used in Excel to calculate the daily Head Fire Intensity (HFI) FWI value for each day modelled (Canadian Forest Service, 2012; Forestry Canada Fire Danger Group, 1992). Consistent with previous research using *Burn-P3*, the HFI was used to limit the simulated daily fire weather data used in the fire behaviour model to only those days where the HFI was ≥ 4000, where fire suppression is difficult or impossible (Figure 3.2) (Parisien et al., 2005).

Daily wind direction cannot be modelled by *BioSIM*, and is a necessary input to model fire behaviour in *Burn-P3*. The meteorological values produced by *BioSIM* also cannot be used to derive wind direction, due to the complex relationship between pressure, friction, and the centripetal and Coriolis forces, which produces wind (Ahrens, 2012). Historical weather data from 1990 to 2012 provided by the Nova Scotia Department of Natural Resources (NSDNR) were used to calculate the mode (most common) wind direction on each day of the fire season. Mode wind directions were matched to *BioSIM* data by corresponding month and date (Figure 3.2).

3.3.3 SCENARIOS

Burn-P3 iteratively models fires and fire behaviour across a landscape using the equations and assumptions of the Canadian FBP System, and provides data about FS and burn probability (Canadian Interagency Forest Fire Centre, 2011a). We used *Burn-P3* to model FS over time in the two study communities (Figure 3.2). The FS of a given cell is equal to the percentage of the modelled burns that ignited it. For the purpose of this study 2000 fires were simulated for each time step of each scenario. *Burn-P3* modelling required: (a) inputs of weather and wind from *BioSIM* and NSDNR historical wind data (for the climate normals years of 1981 - 2010, 2021 - 2050, and 2071 - 2100), (b) a Digital Elevation Model (DEM), and (c) maps of local Forest Fuel Codes (FFC) (Figure 3.2). The FFC layer used in this study was modified using remote sensing data to identify an accurate forest edge, using the method described by Whitman, Rapaport and Sherren (2013)[11]. Each FFC has associated fire behaviour in the *Burn-P3* model.

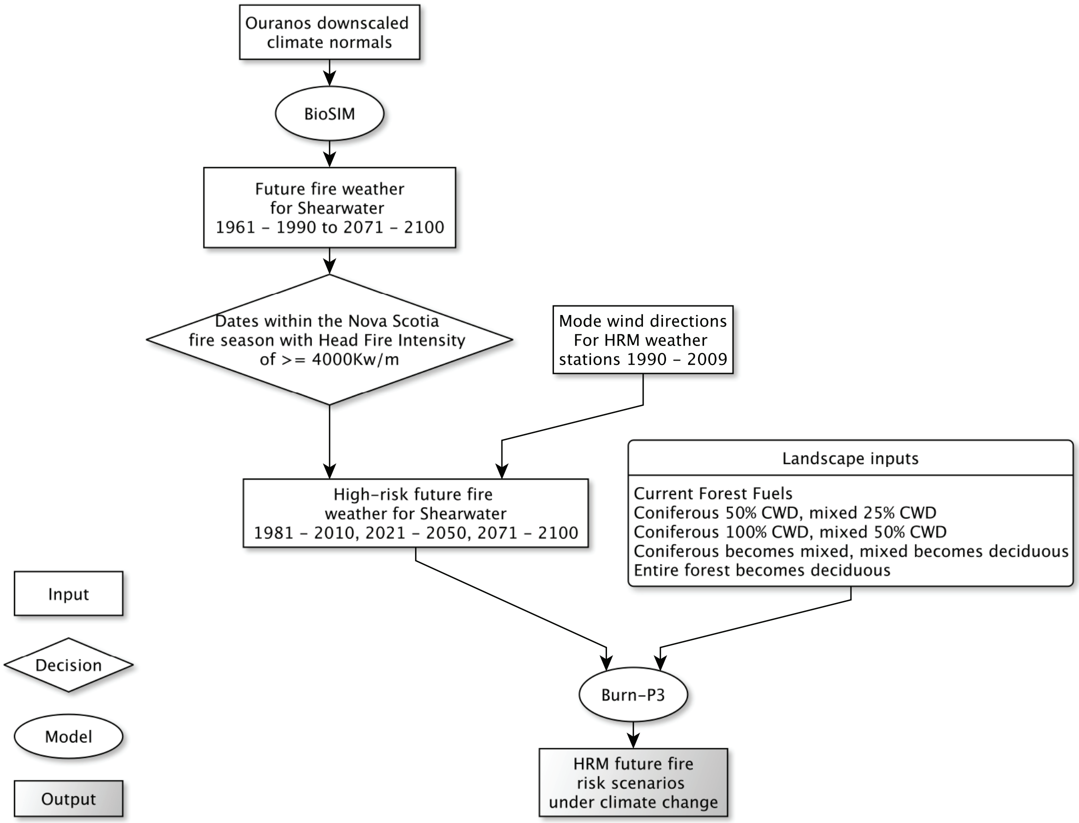

Figure 3.2 Climate change scenario modelling inputs, models and outputs.

Three scenarios of future climate and AFR forest conditions were developed that covered a range of possible future conditions in the AFR: 1. The climate changes, while fuels remain static in their current condition; 2. The climate changes and local fuels respond with moderate changes; and, 3. The climate changes and extreme fuel changes occur simultaneously (Table 3.1). These scenarios represent the range of potential outcomes of climate change in this region, providing an extreme high and extreme low scenario of fire risk in the face of uncertainty, in hopes of defining an envelope of possibility that a realistic scenario might lie within. Five assumptions were common to all three scenarios:

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[11] Chapter 2 of this thesis.

1. Development and changes in land use cease, and the forest edge remains the same indefinitely.
2. Seasonal patterns in wind direction and fire season length remain the same over time.
3. Areas that are currently slash fuels from clear cutting will remain that way until 2021 - 2050, when they will become non-coniferous tree stands. This represents the projected tree species for this region, and ignores the potential role of woodlot owners in determining future fuels.
4. Blowdown and dead stands are represented by the slash fuel code of the dominant tree species of formerly coniferous stands. While a slash fuel code is not ideal for representing wind throw there exists no FFC for wind throw debris at present, and this choice of FFC is acceptable for modelling of extreme wind throw (D. Oikle, personal communication, March 11, 2013).
5. As spruce is the most common coniferous genus in Nova Scotia and conifers dominate the AFR, (NSDNR, 2008; NSMNH, 1994b) mixedwood stands received a spruce slash fuel code when modelling called for a species change over time due to wind throw or decline.

FFCs of the modified FFC layer (Whitman, Rapaport & Sherren, 2013) were reclassified to reflect the fuel compositions of the study site according to each climate normal year of the three scenarios (Table 3.1). Each 30-year time step of these scenarios was modelled in *Burn-P3* with projected weather, and FS was examined at each time point and in comparison across scenarios.

Table 3.1 Scenarios of climate change in the Acadian Forest Region modelled in *Burn-P3*.

Scenario	**Climate**	**Fuel**
Scenario 1. Climate change only	The local climate changes according to the SRES A2 model, downscaled to Shearwater, Nova Scotia.	The local forest fuel codes remain static over time and do not change in composition from that of the present date.
Scenario 2. Climate change and moderate fuel changes	The local climate changes over time according to the downscaled SRES A2 scenario model.	The local forest composition changes over time, with 50% of the area of any coniferous stand blowing down or dying (represented by a slash fuel code) and 25% of the area of mixedwood stands blowing down or dying by 2021 - 2050. By 2071 - 2100 coniferous stands have transitioned to mixedwood stands, and mixedwood stands have become uniformly non-coniferous.
Scenario 3. Climate change and extreme fuel changes	The local climate changes over time according to the downscaled SRES A2 scenario model.	The local forest composition changes over time, with 100% of coniferous stands blowing down or dying (represented by the slash fuel code) and 50% of the area of mixedwood stands blowing down or dying by 2021 - 2050. By 2071 - 2100 the entire forest has become non-coniferous.

3.3.4 ANALYTICAL HIERARCHY PROCESS

To further examine the relative importance of the drivers of risk we conducted an expert elicitation exercise with Canadian wildfire experts as participants. To recruit participants we compiled an initial list of wildfire researchers and fire risk management practitioners, and contacted them by email, requesting they participate in a weighting exercise. Those contacted who chose to participate and completed the online weighting exercise were asked to suggest other participants, using a snowball sampling approach. Eight participants were recruited through our initial email, and an additional four participated, following their recommendation by a peer.

In an online *Opinio* environment (ObjectPlanet, 2012) participants were presented with all possible pairs of seven drivers of wildfire risk and asked to indicate if the two fire risk drivers were equally important, or if one was more important than the other. These fire risk drivers (or elements) were: (a) Precipitation, (b) Coarse woody debris, (c) Temperature, (d) Decomposition rate, (e) Length of fire season, (f) Local tree species composition, and (g) Weakened tree stands on the landscape[12]. If the participant indicated that one driver was more important, they were then presented with a verbal scale with eight options ranging from "slightly more important" to "extremely more important" to rate the relative importance of the more important driver[13] (Nyerges & Jankowski, 2010).

Participant responses were recorded and the verbal scale of importance was converted into numerical values ranging from 1 (equal importance) to 9 (extremely more important) (Nyerges & Jankowski, 2010). Responses were entered into a matrix where the importance of driver A to driver B was equal to the inverse of the importance of driver B to driver A. Weights of relative importance out of one were calculated for each driver, using the method of weight calculation developed by Thomas Saaty for pairwise comparison using eigenvectors, outlined in Nyerges and Jankowski (2010). Weights were averaged across all participants, generating a single weight out of one for each fire risk driver[14].

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[12] For definitions provided to participants and screencaps of the *Opinio* environment see Appendix B, Section B.3.

[13] Format of pairwise comparison environment included in Appendix B, Section B.2.

[14] Detailed discussion of the AHP process is included in Appendix B and Chapter 1.

3.4 RESULTS

Our modelling of future fire weather and climate during the fire season showed similar trends to those found by others (Richards & Daigle, 2011), as both projected temperature and precipitation increased. Both maximum and mean daily temperature during the fire season were projected to increase steadily over time (Figure 3.3), assuming climatic conditions similar to those expected under the A2 high GHG and aerosol emission scenario (Nakicenovic et al., 2000). Maximum daily temperature during the fire season rose from 27 ºC to 33 ºC, and mean daily temperature during the fire season increased from 14 ºC to 18 ºC over the climate normals years of 1961 - 1990 to 2071 - 2100 (Figure 3.3). Daily mean precipitation volume during the fire season was also projected to rise over time, increasing from 2.69 mm at 1961 - 1990 normals to 3.01mm in the climate normals years of 2071 - 2100. These increases did not occur steadily, with some intermediate increase and decreases in volume (Figure 3.4). Mean daily fire weather index values increased from 2.2 to 3.5 from 1961 - 1990 climate normals conditions to 2071 - 2100 climate normals (Figure 3.5).

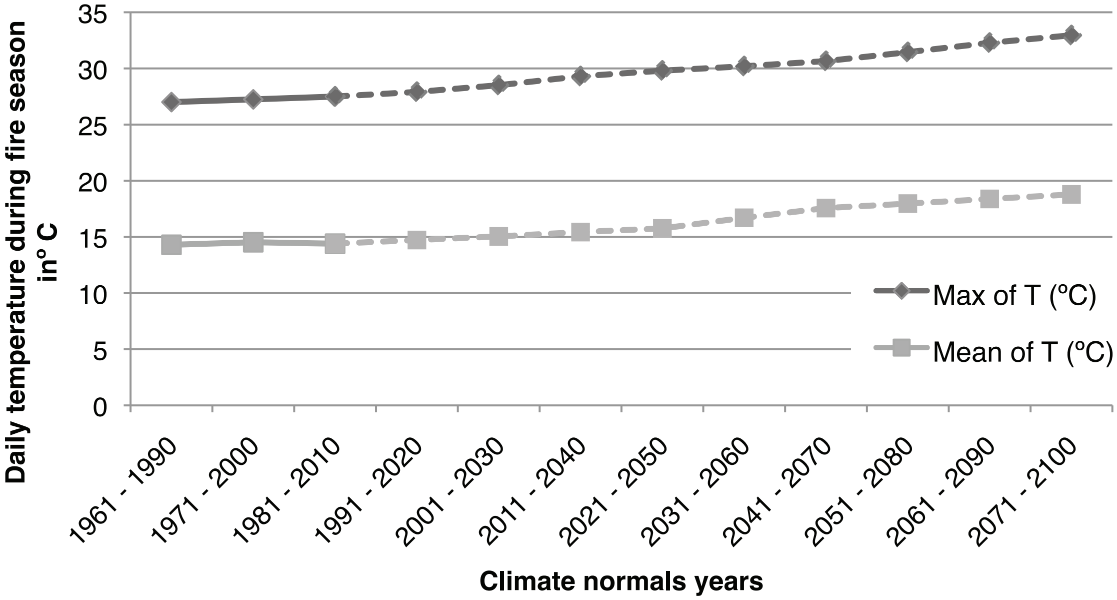

Figure 3.3 Past (solid) and projected (dashed) temperatures for Shearwater Nova Scotia over time during the fire season. Data generated using downscaled SRES A2 scenario climate normals in *BioSIM.*

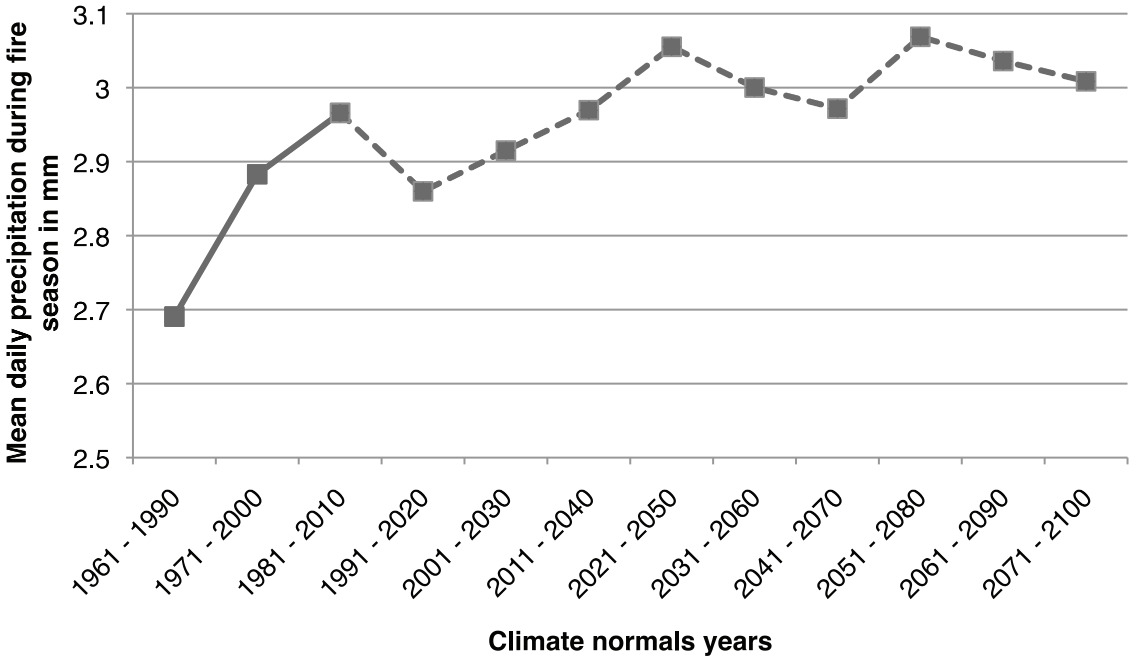

Figure 3.4 Past (solid) and projected (dashed) mean daily precipitation for Shearwater Nova Scotia over time during the fire season. Data generated using downscaled SRES A2 scenario climate normals in *BioSIM*.

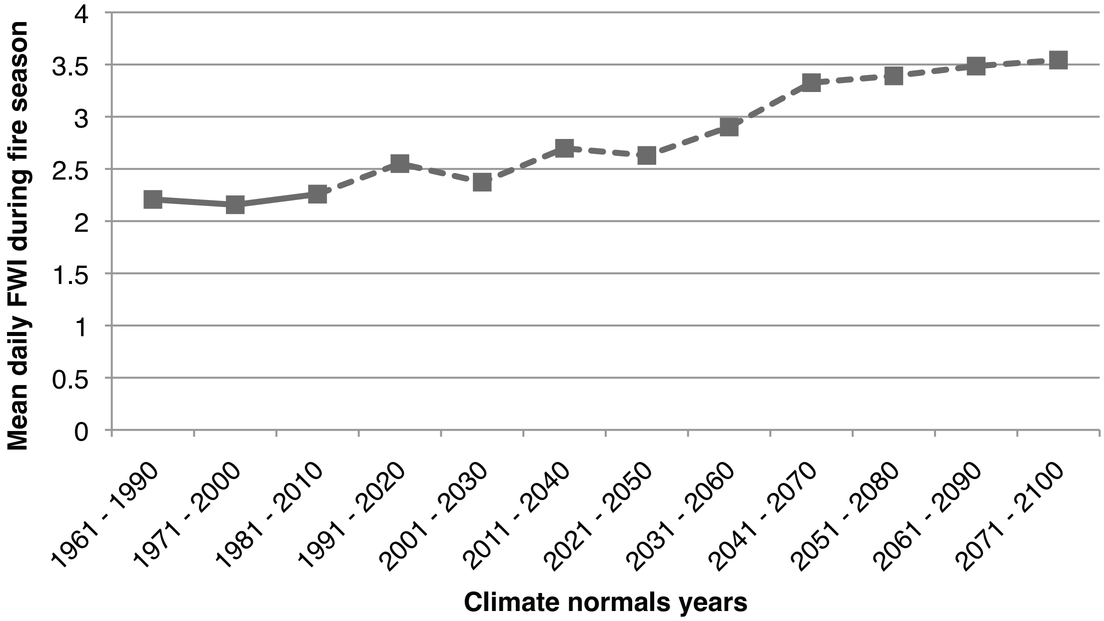

Figure 3.5 Past (solid) and projected (dashed) mean daily FWI values for Shearwater Nova Scotia over time during the fire season. Data generated using downscaled SRES A2 scenario climate normals in *BioSIM*.

A systems model was developed from an extensive literature review (Section 3.1 Introduction), and represents the impacts of climate change on various fire risk drivers in the AFR, and the interrelations amongst these drivers (Figure 3.6). Anthropogenic emissions of GHGs and aerosols are driving climate change in the AFR. Changes in climate, such as the increased temperature during the fire season and lengthening fire season, are projected to act as positive drivers of future fire risk in this region. This is offset somewhat by projected increases in precipitation, which would act as a negative driver of fire risk, however, increasing fire weather severity overall is a positive driver of future fire risk (Figure 3.6), as indicated by the projected increase in FWI over time (Figure 3.5). Changes in climate are also projected to drive a shift towards a non-coniferous tree species community in the AFR, disfavouring boreal tree species. This is a negative driver of future fire risk, however this species shift also involves complex interactions (Figure 3.7).

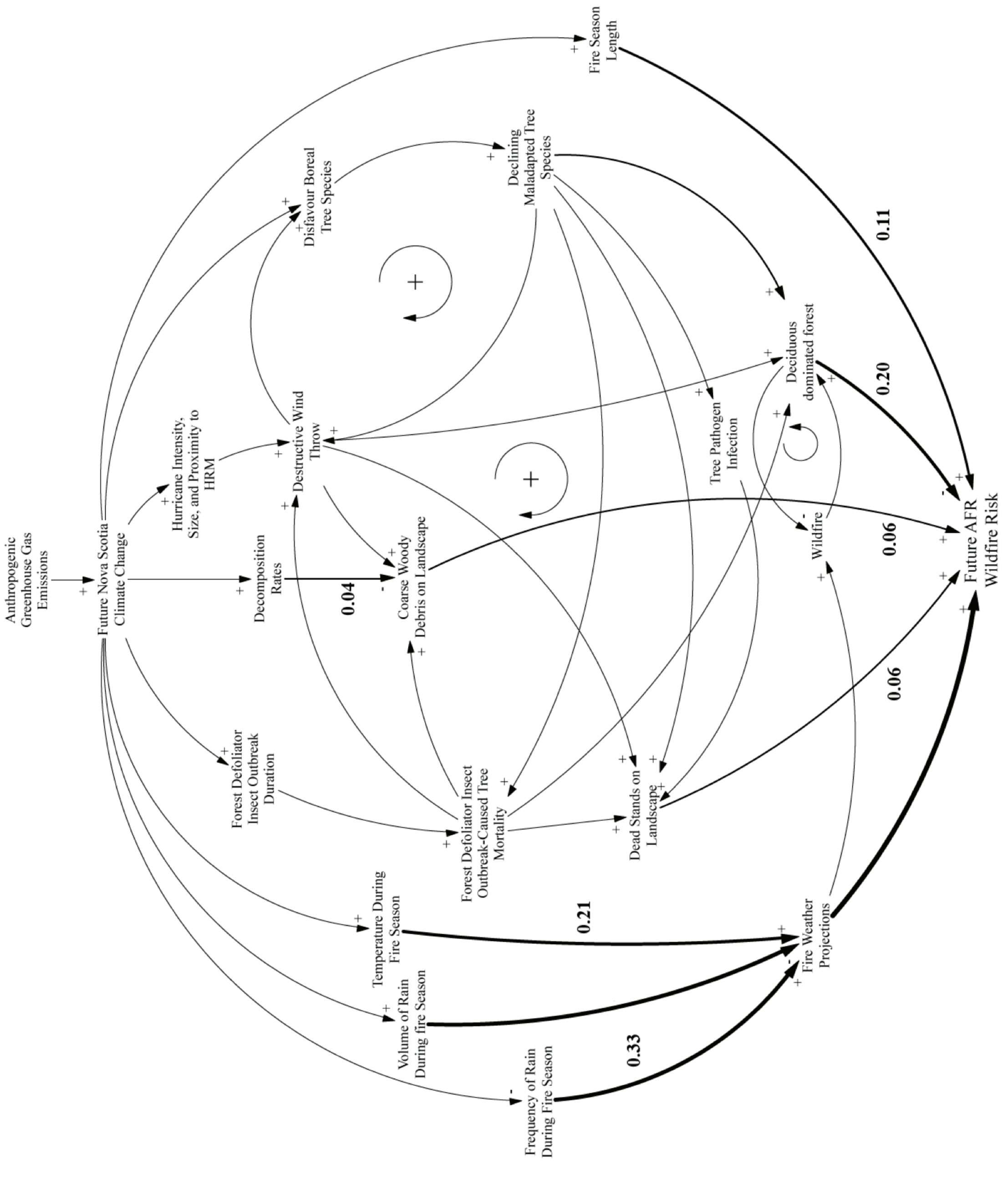

Figure 3.6 Systems model of fire risk drivers and future climate change pathways in the Acadian Forest Region, with relative weights of importance of drivers generated through an Analytical Hierarchy Process.

A reinforcing feedback loop exists within the shift in local tree species under climate change. Wind throw further disfavours coniferous tree species (Foster, 1988a, 1988b), which in turn would lead to damaged and declining stands of conifers on the landscape. Maladapted trees are more susceptible to wind throw and mortality from forest pest outbreaks (Johnston et al., 2009), and many of the maladapted boreal species are coniferous (Bourque et al., 2010; Steenberg et al., 2013). This in turn should increase destructive wind throw, further exacerbated by increasing tropical cyclone severity, size and proximity (Figure 3.7) (Jiang & Perrie, 2007). This positive feedback loop may lead to an escalation of the accumulation of coarse woody debris on the landscape, which is a positive driver of future fire risk (Figure 3.6). Increases in coarse woody debris may be counterbalanced in part by increasing decomposition rates under climate change, reducing flammability of debris and negatively driving wildfire risk (Anderson, 1991; Brown et al., 2003; Chambers et al., 2000).

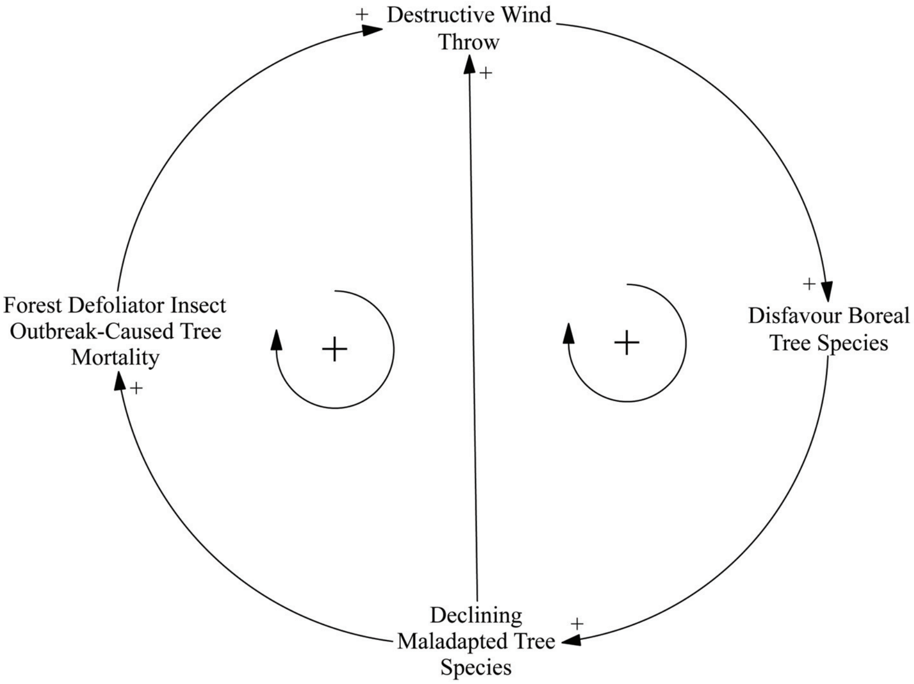

Figure 3.7 Positive feedback loops in the Acadian Forest Region system, under future climate change.

3.4.1 SCENARIO MODELLING RESULTS

Modelling of FS in *Burn-P3* produced mean and maximum FS values for each community, according to the three scenarios modelled[15]. If the forest composition

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[15] Maps of FS for each scenario at the time steps modelled are included in Appendix D.

remains the same over the next century, while projected changes to fire weather occur (Scenario 1), over time fire risk will increase in both communities (Figures 3.8 a; 3.9 a). In Beaver Bank changes to climate with a static forest composition led to an increase in the mean FS of 5%, from 0.88 to 0.93. The FS in Beaver Bank under Scenario 1 increased from a maximum of 4.7 in 1981 - 2010 climate normals conditions to 4.95 in 2071 - 2100 climate conditions, also an increase of 5% (Figure 3.8 a). Increases in FS under Scenario 1 were especially pronounced in Spryfield, reflecting the extensive presence of coniferous tree species that are susceptible to ignition in wildfires (Figure 3.9 a) (Hély et al., 2000; Johnson, 1992). In Spryfield, Scenario 1 produced a projected 10% increase in the mean FS, from 1.34 in 1981 - 2010 conditions, to 1.48 under 2071 - 2100 climate normals, and an increase of 18% from a maximum FS of 10.4 to 12.3 (Figure 3.9 a).

Under the moderate fuel composition changes modelled in Scenario 2, the two communities showed different responses, despite having simulated the same changes to fire weather and fuels over time. Beaver Bank was projected to experience an intermediate increase in FS under 2021 - 2050 climate normals conditions, reaching a mean FS of 1.27, and a maximum of 5.9; representing a 44% and 25% increase from the baseline, respectively (Figure 3.8 b). Additional complexity in local forests was introduced by simulating 25% wind throw within standing mixedwood fuels, and 50% within coniferous stands. In this primarily mixedwood and non-coniferous area, wind throw or slash fuel codes have more severe fire behaviour than the existing FFCs. Once the forest community had transitioned to a mixedwood and non-coniferous forest in the scenario, FS was reduced approximately 40% from a baseline mean FS of 0.88 to 0.52. In this scenario, however, the maximum FS under 2071 - 2100 scenario conditions was higher than the baseline, rising 6% from 4.7 to 5.01 (Figure 3.8 b).

In Spryfield, where the existing tree species composition includes volatile coniferous fuels, the modelled removal of these fuels and their replacement with slash and mixedwood and non-coniferous FFCs, produced projections of a relatively steady decline in wildfire risk over time (Figure 3.9 b). Reflecting the reduction of these flammable coniferous fuels, under Scenario 2 Spryfield transitioned from a mean baseline

FS of 1.34 to a mean FS of 0.75 in a 2071 - 2100 climate, representing a 44% decrease from the baseline FS, with maximums steadily declining as well (Figure 3.9 b).

Under Scenario 3 both communities showed a steady overall reduction in wildfire risk over time, as represented by both mean and maximum FS (Figures 3.8c; 3.9c). This scenario included a rapid transition to a non-coniferous forest, and the extirpation of all purely coniferous stands by 2021 - 2050. In Beaver Bank the community FS declined 65% from a baseline mean of 0.88 to 0.31, in 2071 - 2100 climate normals conditions (Figure 3.8 c). In Spryfield a 74% reduction from a mean FS of 1.34 in 1981 - 2010 climate normals, to a mean FS of 0.35 in 2071 - 2100 climate conditions was projected (Figure 3.9 c).

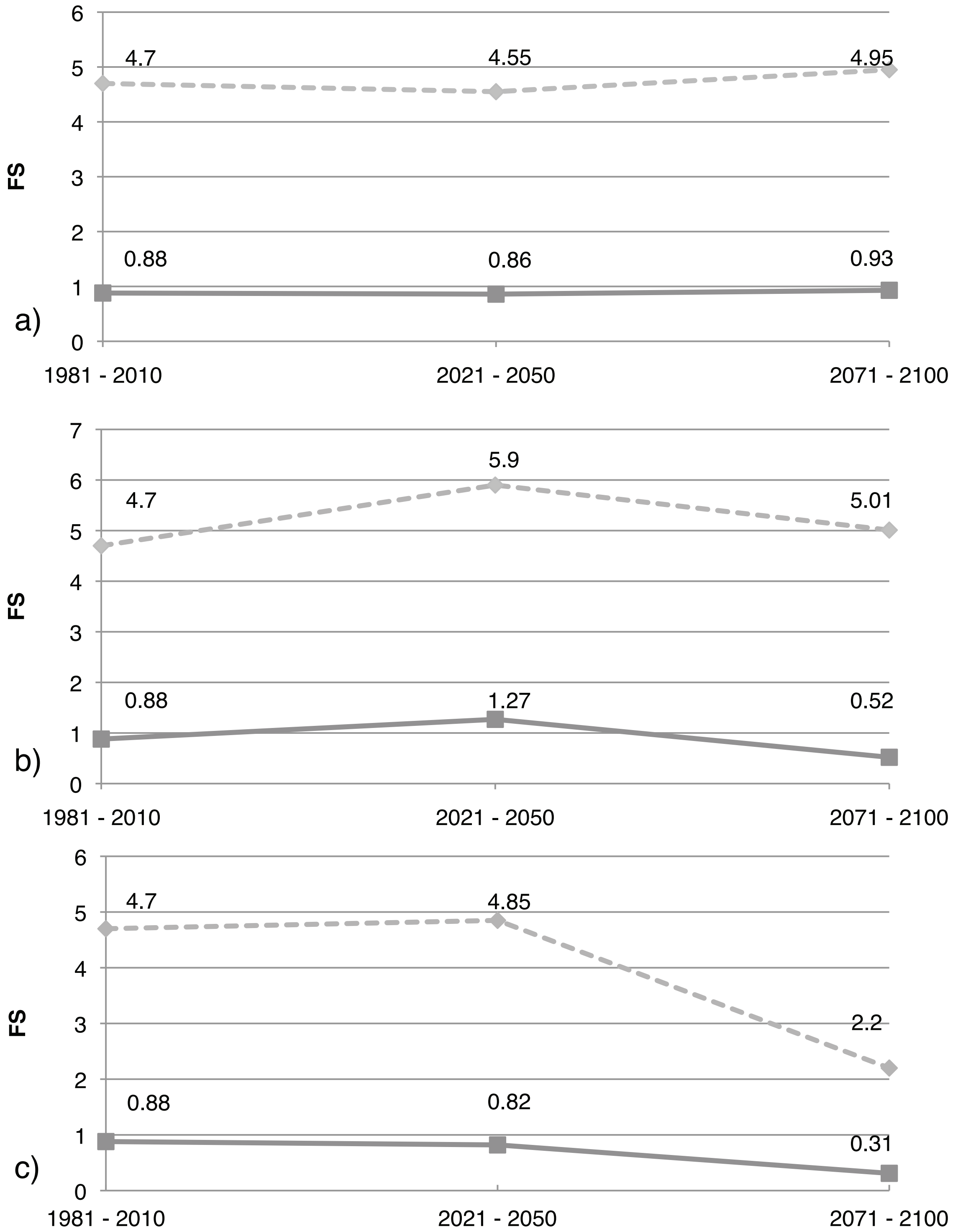

Figure 3.8 Mean (solid grey) and maximum (dashed light grey) fire susceptibility for the Beaver Bank community under climate and fuel changes according to Scenario 1 a), Scenario 2 b), and Scenario 3 c) (Table 3.1).

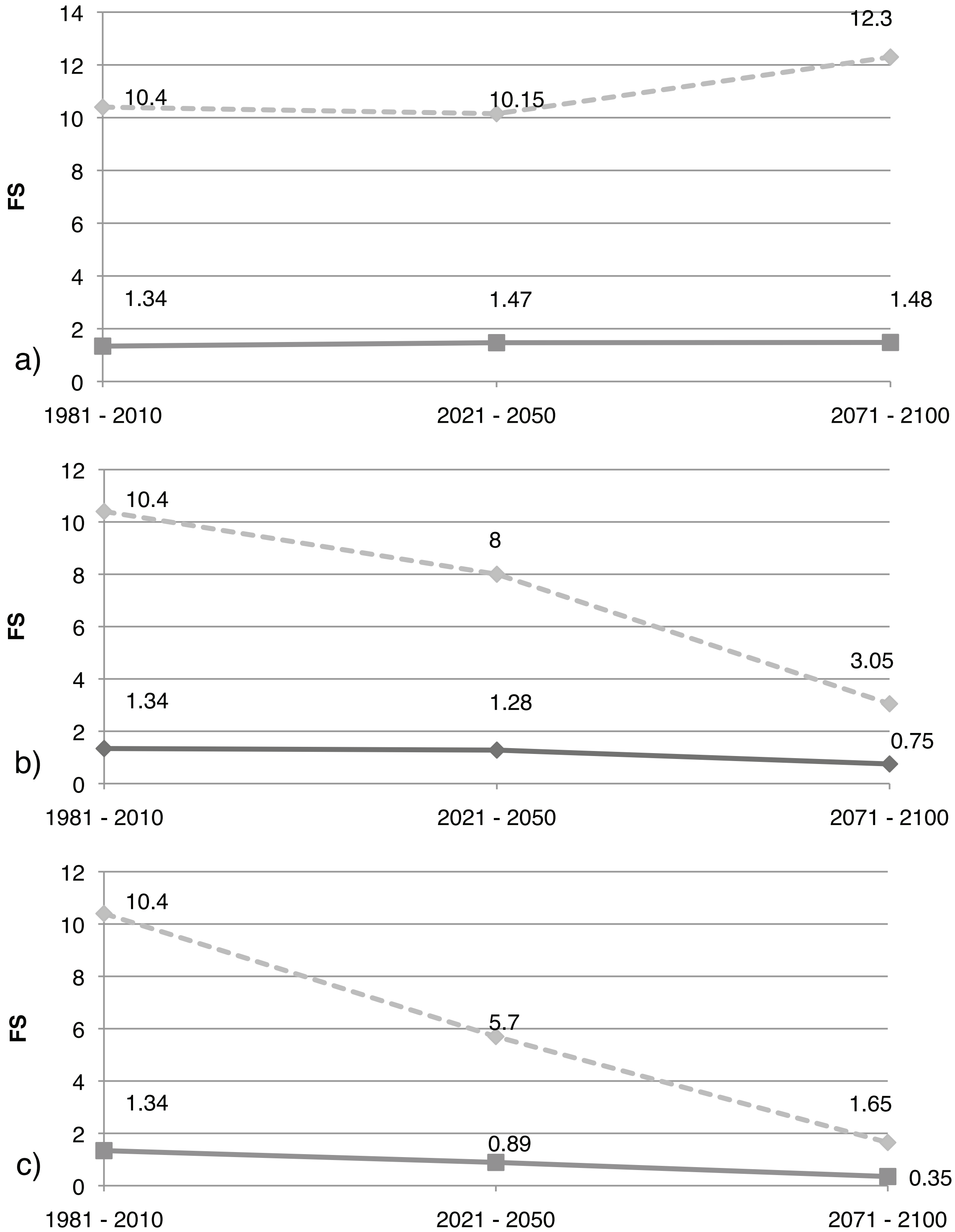

Figure 3.9 Mean (solid grey) and maximum (dashed light grey) fire susceptibility for the Spryfield community under climate and fuel changes according to Scenario 1 a), Scenario 2 b), and Scenario 3 c) (Table 3.1).

3.4.2 ANALYTICAL HIERARCHY PROCESS RESULTS

We used a modified AHP process to elicit relative weights of importance for wildfire risk drivers. These weights are represented by line width in the AFR systems model (Figure 3.6). Participants weighted precipitation as the most important wildfire risk driver, with a mean and median weight of 0.33 out of one. Individual participant weights for this driver ranged from 0.45 to 0.18 (Table 3.2, Figure 3.6). Temperature and local tree species were approximately the second most important drivers. These were seen as approximately equally important with mean weights of 0.21 and 0.20, respectively (Table 3.2, Figure 3.6). Temperature and local tree species drivers also had similar ranges with maximum weights of 0.43 and 0.41, and minimum individual weights of 0.02 and 0.04. The fourth most important wildfire risk driver was fire season length, which received a mean and median weight of 0.11. Individual participant weights of fire season length importance ranged from 0.21 to 0.03. The remaining three wildfire risk drivers of coarse woody debris, weakened tree stands, and decomposition rate were weighted as least important by our participants. Coarse woody debris and weakened tree stands received average weights of 0.06 and had the same maximum and minimum weights, while decomposition rate received a weight of 0.04, and had the smallest individual maximums and minimums of any driver (Table 3.2, Figure 3.6).

Table 3.2 Average weights of relative importance of wildfire risk drivers, generated using a modified Analytical Hierarchy Process with expert participants.

Weight	Precipitation	Temperature	Local tree species	Fire season length	Coarse woody debris	Weakened tree stands	Decomposition rate
Mean	**0.33**	**0.21**	**0.20**	**0.11**	**0.06**	**0.06**	**0.04**
Max	0.45	0.43	0.41	0.21	0.15	0.15	0.11
Min	0.18	0.02	0.04	0.03	0.02	0.02	0.01
Median	0.33	0.20	0.19	0.11	0.04	0.05	0.02

The responses of the experts lacked internal consistency, with non-significant Consistency Ratio (CR) values calculated for every participant response matrix. The minimum participant CR value was 0.17 while the maximum was 1.28, where CR<0.1 shows significance. While this shows a lack of internal consistency (i.e. consistency of one participant's response matrix within itself), the similarity in order of importance across participant responses suggests that this may not have been due to random responses. Each driver was weighted of the same importance by at least a third of all participants. Precipitation, decomposition, fire season length and weakened tree stands were further weighted in the same order of importance by ≥ 41% (5 out of 12 participants) of all participants.

3.5 DISCUSSION

3.5.1 SCENARIO MODELLING

Modelling of climate normals under the high GHG emission A2 SRES scenario (Nakicenovic et al., 2000) projected significant increases in mean daily FWI during the fire season at Shearwater, Nova Scotia, which indicates an increase in the severity of fire weather under future climate change (Figure 3.5) (NSDNR, 2013). Current emissions updates suggest we are emitting GHGs at a greater rate than that assumed in the A2 scenario, which may indicate that this scenario underestimates increasing severity of FWI (International Energy Agency, n.d.; Nakicenovic et al., 2000; Richards & Daigle, 2011). Further, this increase in FWI occurred in spite of increases in daily precipitation volume (Figure 3.4), which could be expected to offset increases in daily temperature and reduce or moderate wildfire risk. This finding is supported by the work of Cary et al. (2006), who projected an increase in area burned under future climate change, with both increasing temperature and precipitation. The overall increase in FWI values over time may also reflect the distribution of precipitation increases throughout the year, which favours winter and spring months (Richards & Daigle, 2011). It is, however, important to consider the relative reliability of temperature and precipitation projections. Confidence in climate modelling is lower for projections of precipitation than for temperature, and errors are more pronounced with downscaled climate models (Solomon et al., 2007), such as those used in this study.

The FS distributions from the two study communities suggest that the severity of increases in risk were site-specific. Areas with significant populations of conifers, such as Spryfield, may experience more dramatic increases in wildfire risk, if tree species compositions do not shift within the next 100 years (Figure 3.9 a). The modelling of wind throw and addition of the slash fuel code did not lead to a notable increase in risk in Spryfield (Figure 3.9 b), however, in Beaver Bank, which has largely non-coniferous forest at present, simulating wind throw led to an intermediate increase in wildfire risk (Figure 3.8 b), further emphasizing the role of site-level landscape dynamics in future fire risk.

If the forest composition of the AFR shifts to a non-coniferous-dominated forest community, as favoured by the future climate (Bourque et al., 2010; McKenney et al., 2007; Steenberg et al., 2013), projected wildfire risk is reduced despite simultaneous increases in daily FWI values (Scenario 3, Figures 3.8 c; 3.9 c). It appears that increasing fire weather severity is offset by the lesser flammability of non-coniferous fuels. This finding is consistent with those of researchers who modelled wildfire ignitions under future climate conditions in an unlimited tree dispersal scenario, where non-coniferous trees rapidly colonized boreal forest regions (Terrier, Girardin, Périé, Legendre, & Bergeron, 2013). In our modelling this reduction in wildfire risk is especially pronounced in areas where the current forest composition is made up primarily of coniferous trees, e.g. the Spryfield study area (Figure 3.9 c). The reduced wildfire risk projected in this scenario is due to the assumption that tree species distributions shift as habitat becomes available, which is unlikely. Other research suggests that time lags will slow and alter the colonization of climate-altered habitats as maladapted boreal species remain on the landscape (Iverson & McKenzie, 2013; Steenberg et al., 2013). The trajectory of wildfire risk over time in the AFR may approximately follow Scenario 2 modelled in this study, with wildfire risk increasing over the next century, followed by an overall decline as maladapted boreal species, and specifically conifers, are replaced by non-coniferous species in the AFR (Figure 3.10). Due to the highly site-specific nature of wildfire risk in this region, those areas where conifers remain prominent will likely face an increase in wildfire risk closer to that projected in Scenario 1.

3.5.2 EXPERT WEIGHTING

AHP participants weighted fire weather determinants, such as temperature and precipitation, as the most important drivers of wildfire risk. This is consistent with, and appropriate for, boreal conditions, where weather and climate determine fire occurrence and spread (Johnson, 1992, p. 3). Precipitation, a primary negative driver of wildfire risk (Johnson, 1992), received approximately one third of the overall weight of importance. Temperature and fire season length are correlated positive wildfire risk drivers (Flannigan & Wang, 2012), and when considered collectively, also received approximately one third of the weighting. In diverse mixedwood and transitional zones, such as the AFR, however, species composition plays a significant role in determining fire risk at a site level (Hély et al., 2000). When grouped according to the general nature of the fire risk driver, fuel drivers of fire risk (local tree species, coarse woody debris, weakened tree stands, and decomposition) were in fact weighted of approximately equal importance to the fire weather drivers (Table 3.3). Aggregating these drivers better reflects the local importance of fuel as a fire risk driver. Additionally, some participants mentioned in written comments, provided as ancillary to the weights, that a rapid species change would be more important than fire weather drivers of fire risk.

When eliciting weights through an AHP a non-significant CR value indicates that the participant should be contacted again and asked to re-evaluate the pairs of elements (Nyerges & Jankowski, 2010; Saaty, 1990). Due to limitations of time and scope this was not completed for the AHP weighting exercise performed in this study. There was, however, similarity across participants in the weighting of importance of drivers. It is important to note that the average weights of relative importance generated through the AHP process in this study were produced from responses that lacked internal consistency for this reason. Despite these concerns, we feel that the similarity of weighting and consistency with modelled results suggests they are meaningful.

Table 3.3 Distribution of weights categorized by a) precipitation; b) temperature and c) fire season length; and, fuel drivers of d) local tree species, e) coarse woody debris, f) weakened tree stands, and g) decomposition rate.

Precipitation	**Temperature and Fire Season**		**Fuel**			
A	**B**	**C**	**D**	**E**	**F**	**G**
	0.21	0.11	0.20	0.06	0.06	0.04
0.33	**0.32**		**0.36**			

Fire risk and climate change present an extremely complex and dynamic system, and present planners with conditions that have never been encountered previously. The internal consistency of responses may simply reflect the difficulty faced when assessing this system in such binary terms. The definitions and selection of drivers of risk were also a potential source of confusion and error in the AHP weighting exercise. Participants noted and criticized the similarity of, and lack of independence within, the seven fire weather variables used, such as temperature and fire season length. The lack of inclusion of drought as a driver was also a concern for participants. Similarly, participants also expressed that the interrelated nature of local tree species composition, weakened tree stands and coarse woody debris on the landscape may have led to confusion. Several participants also indicated that the role of decomposition as a driver of future fire risk is not often included in fire manager training and fire science, therefore they were not familiar with this driver and were unsure of how to weight its importance. The need for additional clarity in the definition of weakened tree stands, as trees that were alive, rather than dead, was also indicated. This led to concerns that participants were unable to meaningfully assess the importance or role of these lesser drivers of risk, potentially influencing the lack of internal consistency in participant responses.

3.5.3 INTEGRATION

The AHP process led us to aggregate wildfire risk drivers into broader categories of precipitation, temperature and fire season length, and fuels. When the influence of positive and negative drivers of fire risk are considered in these amalgamated categories, it appears that the shift in species composition projected for this region will offset the trend of increasing fire weather severity beyond the next century (Figure 3.10; Table 3.4). This conclusion is valid where time lags do not severely affect tree species range shifts under climate change (Bourque et al., 2010; McKenney et al., 2007; Steenberg et al., 2013). In the instance of time lags, where boreal, and specifically boreal conifers remain in the AFR, wildfire risk will increase (Table 3.4). The eventual decline in risk beyond the 2071 - 2100 climate normals period is due to this shift in the role of fuels from a positive driver of risk at site-level (depending on tree species composition), to a broadly negative fire risk driver, as non-coniferous trees present a significantly smaller wildfire risk than conifers found in the AFR today (Table 3.4) (Hély et al., 2000; Johnson, 1992). Although we did not project fire weather beyond the climate normals years of 2071 - 2100, the decline in fire risk under Scenario 3 due to the shift in species occurred despite a simultaneous trend of increasing fire weather severity, suggesting this trend could continue. Broadly, our projections suggest that the trajectory of wildfire risk in the AFR will increase moderately over the next century, and decline over time as coniferous boreal species are replaced by climate-adapted non-coniferous tree species (Figure 3.10).

Table 3.4 Shift in the role of fuel as a wildfire risk driver over time in the Acadian Forest Region, in accordance with the scenario modelling conducted in this study.

	Role of wildfire risk driver as positive or negative			
Time	**Precipitation**	**Temperature and fire season length**	**Fuel**	**AFR wildfire risk**
Present - 2100	-	+	+	↑
Beyond 2100	-	+	-	↓

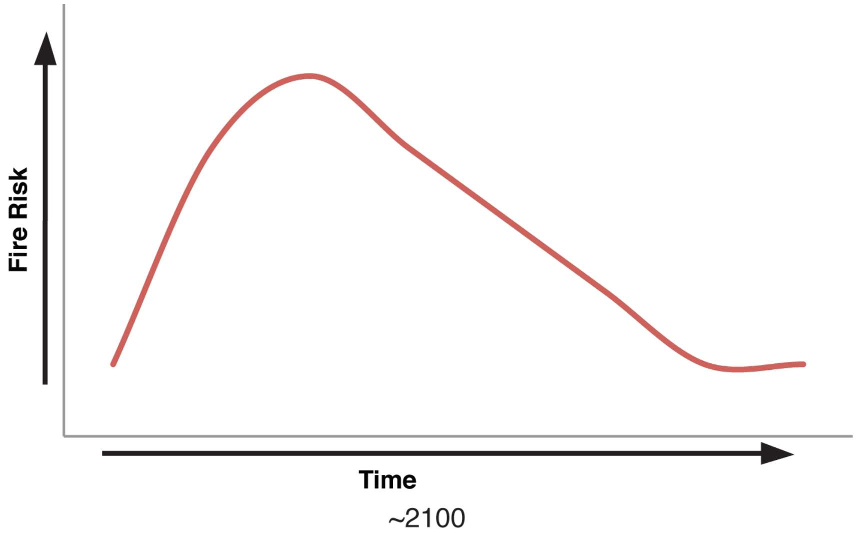

Figure 3.10 Conceptual representation of modelled fire risk in the Acadian Forest Region over time

3.6 CONCLUSION

Following an extensive literature review we developed a systems model of the impact of future anthropogenic climate change on future fire risk in the AFR, clarifying the influence of positive and negative drivers of future fire risk (Figure 3.6). Using *BioSIM* with downscaled climate normals of the SRES A2 scenario we modelled future changes in fire weather in the AFR. This modelling indicated that while both daily volume precipitation and mean and maximum temperatures during the fire season were expected to increase (Figures 3.3 and 3.4), future climate is projected to be a positive driver of future fire risk. Mean FWI values increased over time (Figure 3.5), suggesting that fire intensity in the AFR region will increase steadily as the climate changes (Figure 3.10) (NSDNR, 2013).

Weights of relative importance for seven wildfire risk drivers[16] were elicited from Canadian wildfire experts through an AHP elicitation exercise. Participants indicated that fire weather drivers of precipitation, temperature, and fire season length were the most important determinants of future fire risk, followed by local tree species composition (Table 3.2). When aggregated, precipitation acting as a negative fire risk driver, temperature and fire season length as positive drivers, and fuel variables were of equal importance in determining wildfire risk, emphasizing the role of fuels and landscape dynamics in future fire risk (Table 3.3).

There is potential for this projected increase in fire weather severity to be offset by shifts in tree species composition in the long term, over a century or more of change (+100 yrs) (Bourque et al., 2010; Johnson, 1992; Steenberg et al., 2013; Terrier et al., 2013). Modelling suggests that if the AFR transitions to a non-coniferous tree community, overall future fire risk will be less than that at present, resembling the projections of Scenario 3 (Figures 3.8 c; 3.9 c). Despite this finding, it is likely that colonization of newly available habitats will lag behind shifts in local climate (Iverson & McKenzie, 2013; Iverson et al., 2004; Millar et al., 2007; Steenberg et al., 2013). If total colonization of non-coniferous tree species does not occur, wildfire risk may increase in those areas where mixedwood and coniferous fuels are common, following a trajectory of

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[16] Appendix B.

moderate increases in future fire risk, similar to the modelled Scenarios 1 and 2. Thus, overall wildfire risk in the AFR may increase moderately and at a site-level, until it is eventually offset by the shifting role of tree species as a wildfire risk driver (Figure 3.10; Table 3.4).

3.7 ACKNOWLEDGEMENTS

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CHAPTER 4 CONCLUSION

In this study I assessed current and future wildfire risk in the AFR WUI. I took a case study approach, focusing on the two peri-urban HRM communities of Beaver Bank and Spryfield. Having determined current and future fire risk.

To model WUI I used remote sensing data in combination with fire behaviour modelling. A real forest edge was identified using remotely sensed LiDAR data, and a vegetation index derived from multispectral imagery. FFCs were modified to reflect this edge and used to model FS in *Burn-P3*. A natural break of -1 standard deviation from the mean FS was identified as a threshold value, delineating WUI. Following a literature review, I developed a systems model of interacting drivers of wildfire risk in the AFR under future climate change. This model structured my scenario modelling of future fire risk. Future fire weather in the AFR under climate change was projected in *BioSIM*. FS was then modelled again in *Burn-P3* under scenarios of future forest composition and climate conditions using the projected fire weather. Finally, I developed a conceptual model to balance management trade-offs between peri-urban fire risk and urban forest benefits, incorporating distance and time. Through this process I met my goals of: 1. determining current wildfire risk in the HRM WUI, and 2. examining future fire risk in the AFR WUI under climate change.

4.1 STUDY RESULTS AND OUTCOMES

Delineation of WUI is important for risk management, as it allows managers to apply targeted fuel management treatments, mitigating wildfire risk (Haight et al., 2004). Although models exist for WUI delineation (Beverly et al., 2010; Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007) few of them are ideal for municipal risk management, as they use proxies for wildfire risk, do not take advantage of the detailed data available at the municipal level, do not consider topographic and site-level effects on fire risk, or require broad assumptions about complex fire behaviour. I concluded that delineation of WUI is needed at a site-level scale for municipal management, and developed a transferrable model for WUI delineation incorporating local topography, modified FFC layers derived from remote sensing data, and simulated fire behaviour in the *Burn-P3* model. This model of WUI delineation identifies WUI areas within a case study area using a characteristic of the FS frequency distribution. FS and WUI are not comparable across study sites. Areas delineated using the model matched the definition of WUI.

Projections of future climate change modelled in *BioSIM* suggest that fire weather in the AFR will increase in severity over the next century. This increase in fire weather severity may be offset by the projected shift towards a primarily non-coniferous forest composition (Bourque et al., 2010; Iverson et al., 2008; McKenney et al., 2007; Steenberg et al., 2013), however, due to time lags in the rate of tree range shifts (Iverson & McKenzie, 2013; Steenberg et al., 2013), it is likely that the mixedwood conditions of the AFR will persist in certain sites through the more severe fire weather and fire seasons projected. This suggests that the AFR will experience an increase in wildfire risk in the next century. Wildfire expert participants in the AHP study identified fire weather as the most important driver of wildfire risk, followed by landscape and site drivers, such as fuels. Although tree species alone were not weighted as important as fire weather, fuel-related fire risk drivers were collectively considered equally important as precipitation. After aggregating the fuel-related drivers of risk,these weights reflect the site-level importance of tree species in the mixedwood AFR (Hély et al., 2000; Loo & Ives, 2003).

Urban trees are increasingly rare, and the benefits these trees provide are highly desirable (Canadian Urban Forest Research Group, 2013; HRM, 2012; Nowak & Greenfield, 2012). This study identifies WUI areas with elevated wildfire risk, and projects increasing wildfire risk in the AFR WUI in the future. A typical response to elevated wildfire risk is to implement defensible space around structures at risk, which requires tree removal (HRM, 2011a; Partners in Protection, 2003). Thus, in WUI areas it is difficult to reconcile urban forest promotion with wildfire risk reduction via fuel management treatments.

The use of defensible space fuel management around a home significantly reduces the likelihood that the structure will be destroyed in a fire (Bhandary & Muller, 2009; Gibbons et al., 2012). Modelling of FS using an FFC layer where a defensible space fuel treatment was simulated around structures reduced the number of structures at risk from wildfire in both study communities [17] (Table 4.1). This represented a removal of 171.5 ha of tree canopy in the two communities. Although the urban forest benefits lost along with these trees would be substantial (Canadian Urban Forest Research Group, 2013; HRM, 2012), the removal of these trees reduced the number of structures at risk from wildfire by 31 to 46% in the two communities (Table 4.1).

Table 4.1 Number of structures within fire susceptibility categories in Beaver Bank and Spryfield under current forest conditions, and with a simulated fuel management treatment.

	Fire susceptibility (FS)			**Total number of structures at risk from wildfire**
	>0 - 3	**>3 - 6**	**>6**	
Beaver Bank current forest	35	12	4	51
Beaver Bank simulated fuel management treatment	34	1	0	35 (-31.4%)
Spryfield current forest	136	34	2	172
Spryfield simulated fuel management treatment	82	11	0	93 (-45.9%)

Although urban managers are faced with difficult decisions, it is possible to balance the trade-offs between urban forest benefits and wildfire risk. Despite the conflict between these urban forest benefits and wildfire risk, issues of prioritization can be resolved by including scales of time and distance in the management decisions made around urban trees (Figure 4.1).

At the urban core urban forests are inevitably a more important management priority than wildfire risk (Figure 4.1, a). This recognizes the importance of active urban forest management to prevent urban forest decline, the impossibility of forest wildfire in a densely developed area, and the threatened state of North American urban forests (HRM, 2012; Nowak & Greenfield, 2012). As the distance from the urban core increases,

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[17] Appendix D, Figures D.1 and D2.

priorities shift to WUI wildfire risk reduction at the urban fringe, where wildfire risk is a real threat, and the marginal gain of benefits from additional trees is lower (Hirsch, n.d.; Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007). By delineating WUI, using models such as that developed in this thesis, areas where fire risk is a high priority can be identified to target the urban forest management response for this area. For example, in WUI areas tree species planting can consider wildfire susceptibility of species (e.g. exclude conifers from planting near structures (Johnson, 1992)), and public education campaigns regarding defensible space on private property could be targeted.

Overall management priorities can also to shift as time moves forward, reflecting the potential changes to the fringe forests and to urban forests in the future. In the AFR, as wildfire risk is reduced in response to changes in the forest community over the long term and urban forests become increasingly rare, management priorities can shift from favouring WUI risk management at present, to the protection and promotion of urban forests under future climate change (Figure 4.1, b) (Bourque et al., 2010; Nowak & Greenfield, 2012; Richards & Daigle, 2011).

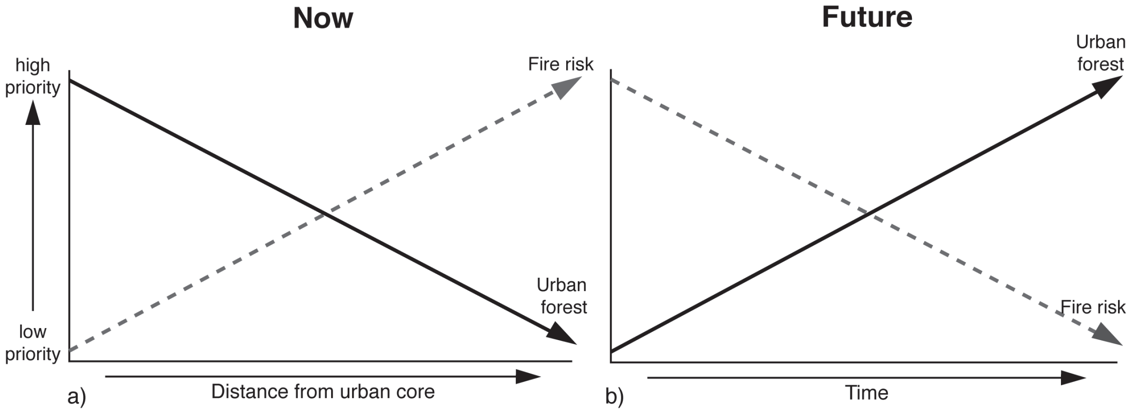

Figure 4.1 Conceptual model of shifting priorities for urban forest management, including scales of a) distance and b) time.

This reframes the risk-benefit trade-off, which only provides win-lose outcomes, to a more refined shifting prioritization allowing managers to protect urban forest benefits and simultaneously reduce risk.

4.2 RECOMMENDATIONS FOR FURTHER RESEARCH

There are several opportunities for future research stemming from this project. Major opportunities lie in expanding the area modelled and in the suite of fire-behaviour models used. In addition, this study did not examine social perspectives of WUI wildfire risk and climate change. Refining the Canadian *FireSmart* defensible space model to reflect the mixedwood forests of the transitional AFR zone could further extend this research. Additional social research of WUI resident preferences would also contribute to the development of a Maritime model for homeowner fuel management. Additional opportunities for further research lie in development of the AHP method, and expansion of the AHP research in this study, as well as in conducting a cost-benefit analysis of peri-urban forests.

The use of a case study approach in this research was a function of the level of detail, but is a limitation. I modelled current and future FS and WUI in only two small communities. Expanding the modelling of FS to encompass the entire municipality would provide an overall rating of relative FS for comparison across the entire municipal management area, allowing broader-scale management. Because of the relative nature of FS and the small number of areas studied, we cannot be sure our histogram-based delineation overcomes problems of comparability. Modelling of a broader area, or additional study sites, may further clarify whether the bi-modal distribution of FS values observed in Spryfield and Beaver Bank (see Chapter 2) is characteristic of WUI areas, or simply a product of the fuel codes modelled in *Burn-P3*. This FS distribution pattern determined the area delineated as WUI, and confirming the commonality of this pattern would support the transferability of this model.

Further research could also be conducted comparing fire behaviour, FS, and WUI delineation as modelled using *Burn-P3*, which is based in the *Prometheus* common object model, to results from *FARSITE*. *FARSITE* is the American fire behaviour and burn probability model, which uses very similar inputs and equations to model fire behaviour (Canadian Interagency Forest Fire Centre, 2011a; USDA Forest Service, 2010a). By comparing results from the two models researchers could highlight differences in the performance of the two, and possibly improve and test the transferability and defensibility of the WUI model I developed.

There are also significant opportunities for social science research in WUI and climate change in this region, and on wildfire expert opinions. This study did not examine public perceptions of current and future wildfire risk in the HRM, nor did it assess the relationship between areas of WUI and vulnerable populations, such as elderly and low-income residents. Studies examining the preferences of WUI residents for fuel management, and the vulnerability of these populations have been conducted elsewhere, but not in Nova Scotia (Gaither et al., 2011; McGee, 2007). Research examining WUI resident preferences can be used to prioritize various management strategies to allow for fuel management without antagonizing residents, or restricting desirable options.

The Canadian defensible space and fire risk management program *FireSmart* was developed in a boreal forest context where wildfire regimes are severe (Government of Alberta & Sustainable Resource Development, 2011; Lauzon et al., 2006). Due to the context in which *FireSmart* originated it may be excessive in its fuel management, or inapplicable in the mixedwood AFR. Discussions with provincial risk management staff, and local forest experts suggested that this model could be further adapted for use in Nova Scotia. Thus, there is a possibility for research to refine the *FireSmart* program recommendations for the AFR context, and for social research on WUI resident preferences, to inform a defensible space and fuel management program tailored to Nova Scotia.

I was not able to make full use of the comments received from expert participants in the AHP weighting exercise, which proved surprisingly rich. The perceptions of future risk and opinions on relative importance of wildfire risk drivers of wildfire experts offer an interesting opportunity for further research regarding their values and personal opinions, rather than expert opinions. For example, this could include a discussion of experts' personal preferences for, and use of, defensible space fuel management.

Another opportunity for further study is the refinement of the AHP method, which produced non-significant consistency ratios for all participants. Future research could determine whether this was a result of the online AHP environment design, the drivers and definitions of wildfire risk drivers chosen, or if this is a function of the inherent complexity of wildfire risk in a changing climate. It is possible that the lack of internal consistency in participant responses may be an artefact of the AHP online tool design, where matrices of relative importance were not presented to the participants as they compared individual drivers. The holding back of this visual information may explain why participants were unable to recall the importance they had assigned to past sets of drivers, and recognize the connections between them. For example, while the participant may have considered A > B and B > C, without being able to see these past responses the logic of weighting A > C may not have been apparent. Conversely, this may have prevented participants from intentionally manipulating their choices to produce a strong matrix, rather than providing responses that reflect their opinions. Further study comparing various AHP weight elicitation methods and environments may prove interesting. The limited timeline and scope of this research prevented me from requesting that participants whose responses produced non-significant CRs re-complete the weighting exercise. This is a standard method in AHP (Nyerges & Jankowski, 2010; Saaty, 1990) and doing so would produce weights of relative importance for wildfire risk drivers that are not limited by concerns of random responses.

Finally, conducting a cost-benefit analysis of the peri-urban forest and wildfire risk reduction from fuel management would provide a quantitative assessment of fire risk and urban forest trade-offs, to better manage urban forests. This could help justify the development of municipal urban forest management strategies, and refine the shifting model of urban forest management prioritization shown in Figure 4.1.

4.3 RECOMMENDATIONS FOR MUNICIPAL WILDLAND-URBAN INTERFACE WILDFIRE RISK MANAGEMENT

The WUI is expanding in North America, and this region has an elevated risk of wildfire (Hammer et al., 2009a; Radeloff, Hammer, & Stewart, 2005; Theobald & Romme, 2007; Vince et al., 2005). Management of this risk is very important (Haight et al., 2004; Vince et al., 2005). My modelling, and that of others, suggests that in the next century wildfire risk in the AFR and in Canada will increase, further elevating the importance of wildfire risk as a management priority in the future (Flannigan & Wang, 2012; Gillett et al., 2004; Wotton, Martell, & Logan, 2003).

The ideal management option to reduce future wildfire risk in the WUI is the restriction or prevention of continuing WUI development. New development at urban fringes comes into contact with wildlands and creates new interface regions. Because WUI has an inherently elevated wildfire risk, preventing the creation of new WUI limits the potential increase in the area at risk from wildfire (Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007; Vince et al., 2005). Furthermore, limiting the new development that creates WUI would conserve natural assets and mitigate other environmental problems associated with the WUI and urban development, such as stormwater management and introduction of invasive species (Canadian Urban Forest Research Group, 2013; Vince et al., 2005).

The tools available to municipalities to prevent and manage the development of new WUI are primarily zoning-based. Zoning has long been established in Canada to manage where and how new development occurs in a municipality (Cullingworth, 1987). After identifying regions at risk from wildfire, a municipality can apply zoning controls before development occurs to direct the construction of communities in such a way as to reduce local fire risk. This includes overlay zoning, incentive zoning and the use of other planning tools. Overlay zoning involves the application of a zone atop existing zones. This overlay zone "supplements the underlying zoning standards with additional requirements" (Vince et al., 2005, p. 84). These requirements can aim to protect environmental features, or apply more stringent development standards, taking into account best practices for fire risk management in building codes and subdivision form (Hughes & Mercer, 2009; Vince et al., 2005). An alternative approach to adding more restrictive zoning is to create incentive zones. Incentive zones offer waivers of zone requirements to developers in return for the developer electing to develop in a way that the municipality encourages (Vince et al., 2005). For example, a municipality could relax restrictions for density or lot sizes in a forested zone, in return for the developer designing a suburb following *FireSmart* design guidelines, or including multiple routes of access to the community in case of fire. In addition to zoning, municipalities (depending on local legal division of powers) can also use concurrency requirements, impact fees, and revise subdivision controls for fire-prone areas to manage the development of the WUI. These strategies target the development form and developer decision-making in fire-risk areas, rather than attempting to manage the development type or use through zoning (Vince et al., 2005).

North America also contains a significant area of existing, long-established WUI (Hirsch & Fuglem, 2006; Radeloff, Hammer, & Stewart, 2005; Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007). Fire risk in these areas could be more actively managed with better public education and community action programs promoting fire risk awareness and mitigation measures that homeowners can employ, such as the *FireSmart* program. If limiting or preventing new development is too costly, difficult, and risky financially, then public education and community engagement are a strong options to protect new WUI homes and residents (Vince et al., 2005). Given the well-known positive impacts of defensible space on home safety (Bhandary & Muller, 2009; Gibbons et al., 2012), and the strong results of advance-preparation education and evacuation programs (Country Fire Service & Government of South Australia, 2010; Mutch et al., 2010), WUI homeowners can be targeted and informed with fire-risk mitigation programs to build community capacity and empower homeowners to manage their own risk (Vince et al., 2005).

The precautionary principle can be summarized as, “when an action may inflict harm, proceed with caution” (HRM, 2012, p. VIII). WUI always has an elevated wildfire risk, and this risk will likely increase under future climate change. In the face of increasing future wildfire risk, the protection of existing WUI and prevention of new WUI development should be a management priority for Canadian municipalities (Federal Register, 2001; Hirsch, n.d.; Radeloff, Hammer, Stewart, et al., 2005; Theobald & Romme, 2007), in accordance with the precautionary principle. To maintain the status quo in WUI development in the future is not an option, and to do so is playing with fire.

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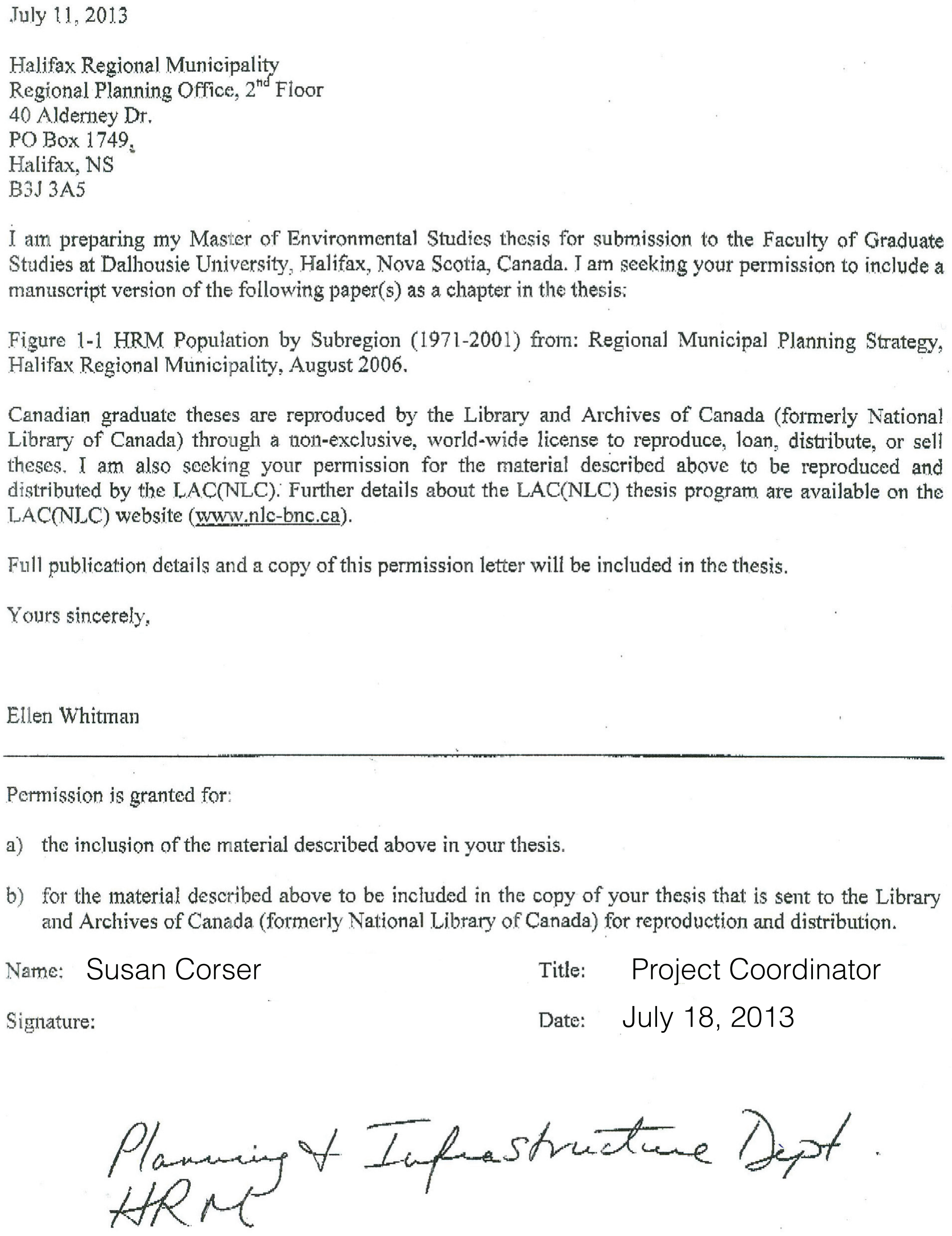
APPENDIX A COPYRIGHT RELEASE



July 11, 2013

Halifax Regional Municipality
Regional Planning Office, 2nd Floor
40 Alderney Dr.
PO Box 1749,
Halifax, NS
B3J 3A5

I am preparing my Master of Environmental Studies thesis for submission to the Faculty of Graduate Studies at Dalhousie University, Halifax, Nova Scotia, Canada. I am seeking your permission to include a manuscript version of the following paper(s) as a chapter in the thesis:

Figure 1-1 HRM Population by Subregion (1971-2001) from: Regional Municipal Planning Strategy, Halifax Regional Municipality, August 2006.

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Title: Project Coordinator

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Date: July 18, 2013

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HRM

APPENDIX B ANALYTICAL HEIRARCHY PROCESS

B.1 RECRUITMENT EMAIL

Dear Dr./Ms./Mr. PARTICIPANT NAME,

I am a Master of Environmental Studies student in the School for Resource and Environmental Studies at Dalhousie University, Halifax, Nova Scotia. My master's thesis research examines current and future wildfire risk in the wildland-urban interface (WUI), with a focus on management of the Halifax Regional Municipality WUI and the Acadian Forest Region.

I am requesting your assistance as a Canadian wildfire expert to help me understand the relative influence of individual drivers of wildfire on future wildfire risk. Thus far I have reviewed literature about the projected outcomes for individual drivers of wildfire risk under climate change. This review noted that these drivers variously increase and decrease local wildfire risk over time. To consolidate these contradictory drivers of risk into an overall trend of increasing or decreasing local risk I need to be able to weight their individual influence. I am using an Analytical Hierarchy Process (AHP) to elicit those weights from experts like you. The specific task is to compare 21 pairs of wildfire risk elements in terms of their relative influence on fire risk. The AHP pairwise comparison process will be administered using an online tool, and will take approximately 10 minutes to complete.

Through my research and participation in the Wildland Fire Canada 2012 conference, I have identified you as potential expert participant. If you are willing to participate, or would like to receive more detail about this study please email me at XXXXXXX@dal.ca or phone (XXX) XXX-XXXX. My supervisors Dr. Kate Sherren (XXXXXXX@dal.ca) and Dr. Eric Rapaport (XXXXXXX@dal.ca)[18] are also happy to provide additional context. If you are unable to participate, please also email me, and I will remove you from the participant list and you will not receive any further reminder emails. Thank you for your time.

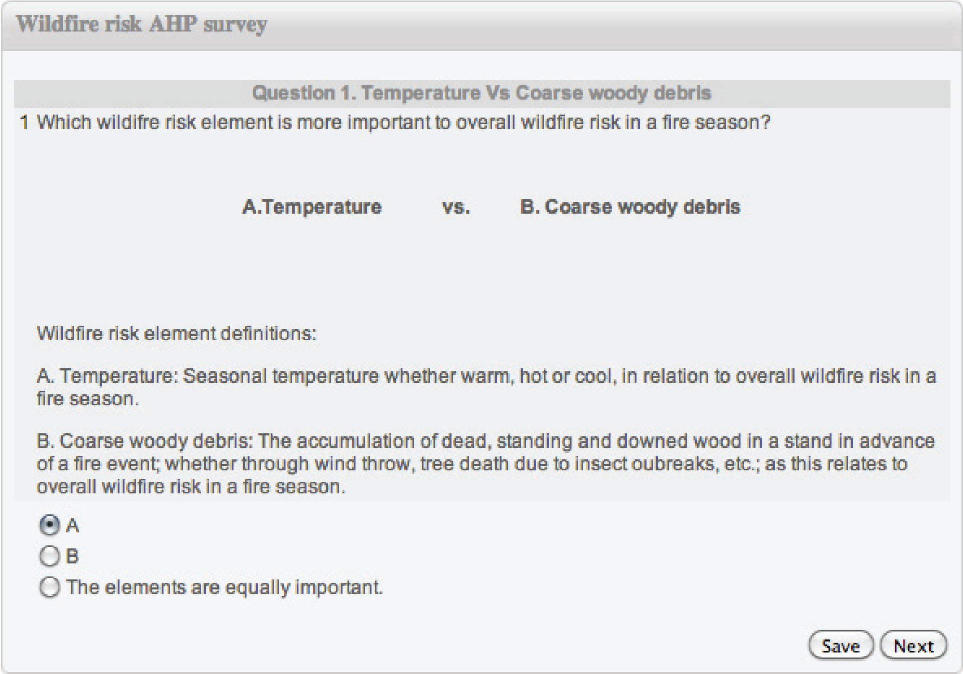
---

[18] Email addresses and phone numbers are represented with Xs here for privacy reasons.

Regards,
Ellen Whitman
Master of Environmental Studies candidate, 2013
School for Resource and Environmental Studies
Dalhousie University

B.2 EXAMPLE ONLINE ANALYTICAL HIERARCHY PROCESS ELICITATION FORMAT

Having agreed to participate in the AHP elicitation exercise participants received an email through *Opinio* (ObjectPlanet, 2012) including a link to the AHP elicitation environment. The first page of the website repeated the information included in the recruitment email, explaining the process, the task and providing contact information. Having clicked "next" the participant encountered the first pair of drivers for comparison, formatted as in Figure B.1. If at this point the participant chose "the elements are equally important" they would proceed to the next pair for comparison, bypassing the weighting of intensity of importance.



Wildfire risk AHP survey

Question 1. Temperature Vs Coarse woody debris

1 Which wildfire risk element is more important to overall wildfire risk in a fire season?

A.Temperature vs. B. Coarse woody debris

Wildfire risk element definitions:

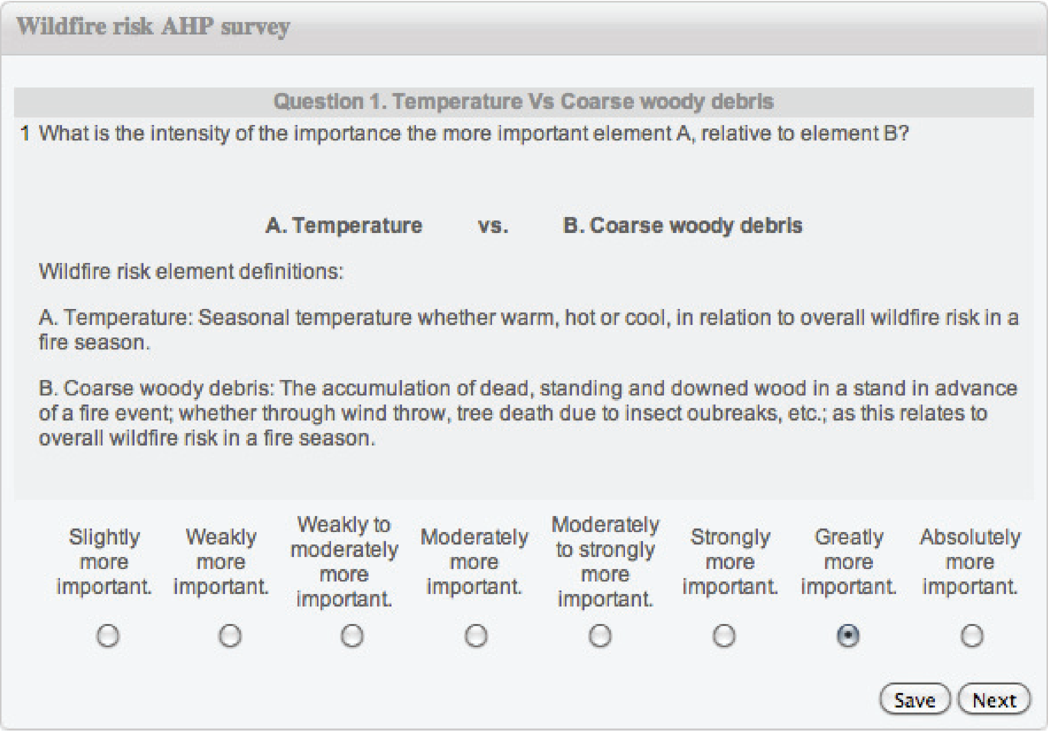
A. Temperature: Seasonal temperature whether warm, hot or cool, in relation to overall wildfire risk in a fire season.

B. Coarse woody debris: The accumulation of dead, standing and downed wood in a stand in advance of a fire event; whether through wind throw, tree death due to insect oubreaks, etc.; as this relates to overall wildfire risk in a fire season.

● A
○ B
○ The elements are equally important.

Figure B.1 Example format used in this study in *Opinio* for pairwise comparisons.

If the participant indicated that one element was more important than the other (‘A’ Temperature, in this example) they then proceeded to a question asking them to weight the intensity of the importance of that element (Figure B.2). The importance of the less important element (‘B’ Coarse woody debris, in this example) is always the inverse of the importance of the other driver element; therefore, the participant compares only one driver’s importance relative to the other. In the example shown here (on a scale of 1 – 9, where 1 represents equal importance), the importance of A relative to B is 8, and B relative to A is 1/8.



Wildfire risk AHP survey

Question 1. Temperature Vs Coarse woody debris

1 What is the intensity of the importance the more important element A, relative to element B?

A. Temperature **vs.** **B. Coarse woody debris**

Wildfire risk element definitions:

A. Temperature: Seasonal temperature whether warm, hot or cool, in relation to overall wildfire risk in a fire season.

B. Coarse woody debris: The accumulation of dead, standing and downed wood in a stand in advance of a fire event; whether through wind throw, tree death due to insect oubreaks, etc.; as this relates to overall wildfire risk in a fire season.

Slightly more important.	Weakly more important.	Weakly to moderately more important.	Moderately more important.	Moderately to strongly more important.	Strongly more important.	Greatly more important.	Absolutely more important.
○	○	○	○	○	○	●	○

Figure B.2 Example format used in this study in *Opinio* to weight the importance of the more important wildfire risk driver element.

Having weighted the relative importance of the more important risk driver element they would then proceed through 20 more identically formatted pairwise comparisons until they had compared all seven elements to one another. Once all the pairwise comparisons were complete they were then thanked for their time, given opportunity to express any comments and concerns, reminded of the contact information of the researchers, and given an opportunity to request acknowledgement in manuscripts stemming from this elicitation process. For more detail about the AHP refer to Chapter 1, section 1.5.1Analytical Hierarchy Process; or Chapter 3 of this document.

B.3 WILDFIRE RISK DRIVER DEFINITIONS

Temperature: Seasonal temperature whether warm, hot or cool, in relation to overall wildfire risk in a fire season.

Precipitation: Precipitation or lack thereof, in relation to overall wildfire risk in a fire season.

Fire season length: The length of an entire fire season, whether it is shorter or longer than average, as this relates to overall wildfire risk in a fire season.

Local tree species composition: The tree species composition of local forests, from coniferous and fire-selected species stands to mixedwood and non-coniferous stands, as related to wildfire risk in a fire season.

Coarse woody debris: The accumulation of dead, standing and downed wood in a stand in advance of a fire event, whether through wind throw, tree death due to insect outbreaks, etc., as this relates to overall wildfire risk in a fire season.

Weakened tree stands: Weakened stands of trees, whether damaged by insect or disease outbreaks, or through maladaptation to shifting climate bands, etc., as related to overall wildfire risk in a fire season.

Decomposition rate: The rate at which dead trees and coarse woody debris on the landscape are broken down and converted to organic material and soils, in relation to overall wildfire risk in a fire season.

APPENDIX C CHAPTER 2 DATA, DATA QUALITY, AND MODELLING

C.1 DATA AND GIS PROCESSING

This section describes in detail the data, data sources, and processing steps used in Chapter 2 of this thesis. The potential limitations to this research due to generalization of data and assumptions in the model used to delineate WUI are also discussed, aiding in the interpretation of results from this chapter.

C.1.1 DATA AND PROCESSING

This research involved data of various scales, origins, and formats for eventual use in the *Burn-P3* model. These data, therefore, required processing which in some cases led to generalization. LiDAR data used in this study was collected initially in 2007, and was preprocessed by Dalhousie University GIS centre staff to create the detailed DEM and DSM used in this research. These raster layers had a 1m x 1m resolution, and raster cell values precise to four decimal places. *QuickBird* multispectral imagery was purchased from *DigitalGlobe*, and had red, green, blue and NIR bands. This imagery had a raster cell size of 2.4m x 2.4m, and was collected in the years 2007, and 2005. These remotely-sensed layers were used to modify provincial FFC polygon data, which was created through photo interpretation of 1:10,000 air photos (NSDNR, 2012).

The FFC data were converted into a 1m x 1m raster file in *ArcGIS* by using cell centre assignment, for use with the LiDAR and *QuickBird*. When the FFCs had been expanded into the identified forest edge, the modified FFC raster and the DEM were resampled using nearest neighbour resampling to a 5m x 5m grid resolution for ease of processing in *Burn-P3*, and in keeping with the appropriate resolution for a 1:10,000 scale dataset, which is the scale of the unmodified FFC data (Tobler, 1987). These grids were then converted to ASCII files. It is assumed that, although the level of detail of remote sensing data were reduced through resampling, these grids are still a strong spatial representation of the landscape, and the refinement of the forest edge, while generalized, still provided additional detail excluded in the provincial FFC layer.

C.1.2 SPATIAL ANALYSIS AND MODELLING

The following sections describe the *ArcGIS* processing and modelling performed in Chapter 2 of this research to delineate WUI.

C.1.2.1 IDENTIFICATION OF TREES

To modify the FFC forest edge I generated an accurate raster layer of treed areas and individual trees in the study area (Figure C.1, Step 1). The tree layer was created using a vegetation index to locate areas of dense vegetation and selecting objects of tree height from within the vegetated areas, located using the NDVI. *QuickBird* imagery was processed in the raster calculator environment of *ArcGIS*, subtracting the red band of the multispectral image from the NIR, and dividing by the sum of the NIR and red bands (Equation 1.4). NDVI values range from -1 to 1, with a value of 0 representing no leaf cover, and dense vegetation represented by values of greater than or equal to .5 (Figure C.2) (Carlson & Ripley, 1997; Myneni et al., 1995; Weier & Herring, 2011). The NDVI was used to distinguish vegetation cover from non-vegetated lands. These vegetated areas were further limited to trees by creating a Normalized Digital Surface Model (nDSM) (Equation 1.5) also produced in raster calculator, using LiDAR height data (Figure C.2) (Demir et al., 2008; Priestnall et al., 2000).

The nDSM represents only heights above ground level, such as buildings and trees, and assigns a height value of 0 to ground-level surfaces (Demir et al., 2008; Priestnall et al., 2000). By extracting only nDSM elevations of greater than or equal to five metres (tree height, according to the Nova Scotia Forest Ecosystem Classification) (Neily et al., 2011) that intersect with NDVI values representing dense vegetation I generated an accurate raster of tall vegetation (tree) heights and locations (Figure C.2) (Demir et al., 2008; Waser et al., 2008). I created a smoother vegetation height raster by generalizing a focal statistics maximum, followed by a focal statistics mean. This layer was then reclassified into a Boolean raster layer where trees received a value of 1 and all other land cover received a value of 0, and subsequently into a layer where trees received a value of 0 and all other land cover was assigned a value of 1, completing the identification of trees (Figure C.1, Step 1; C.2).

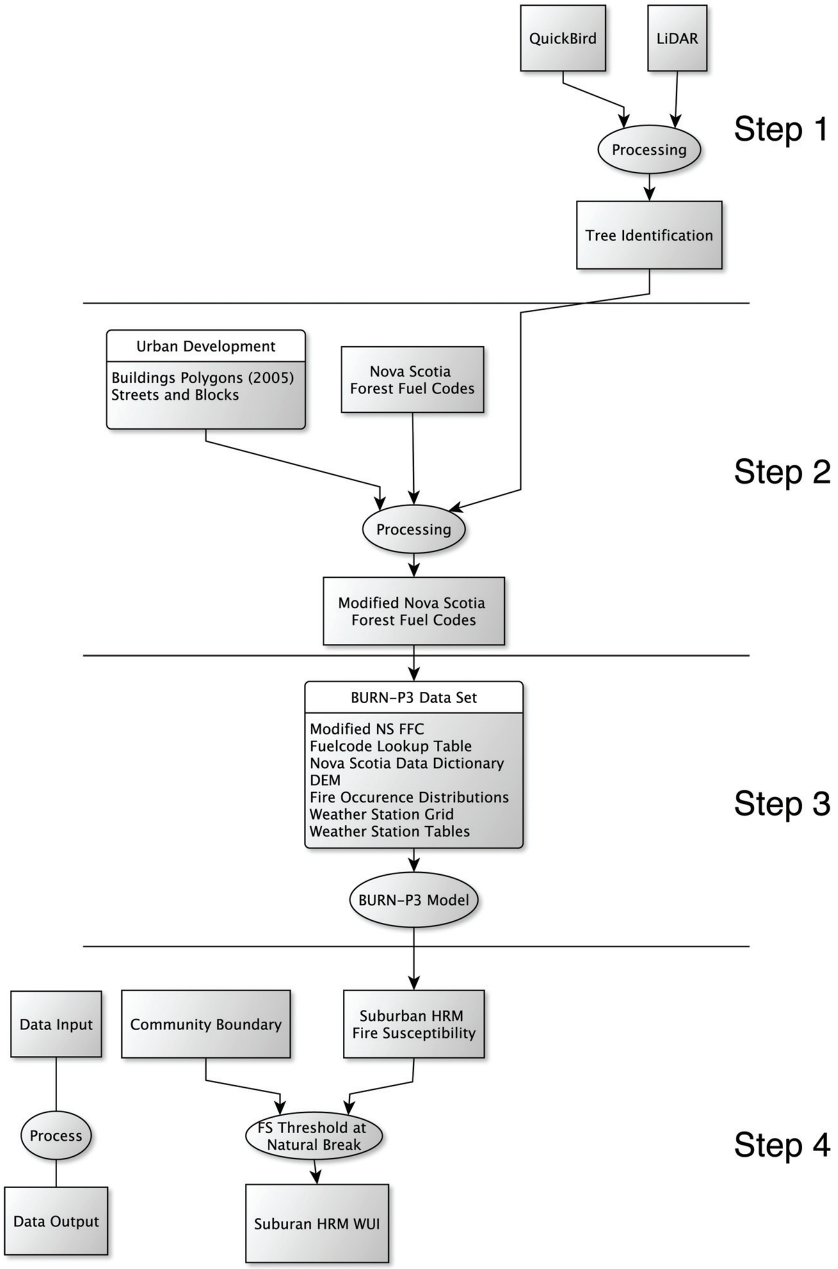


Figure C.1 Steps in *ArcGIS* processing and *Burn-P3* modelling to delineate wildland-urban interface using fire susceptibility. These steps are broken down further in Figures C.2, C.3 and C.4.

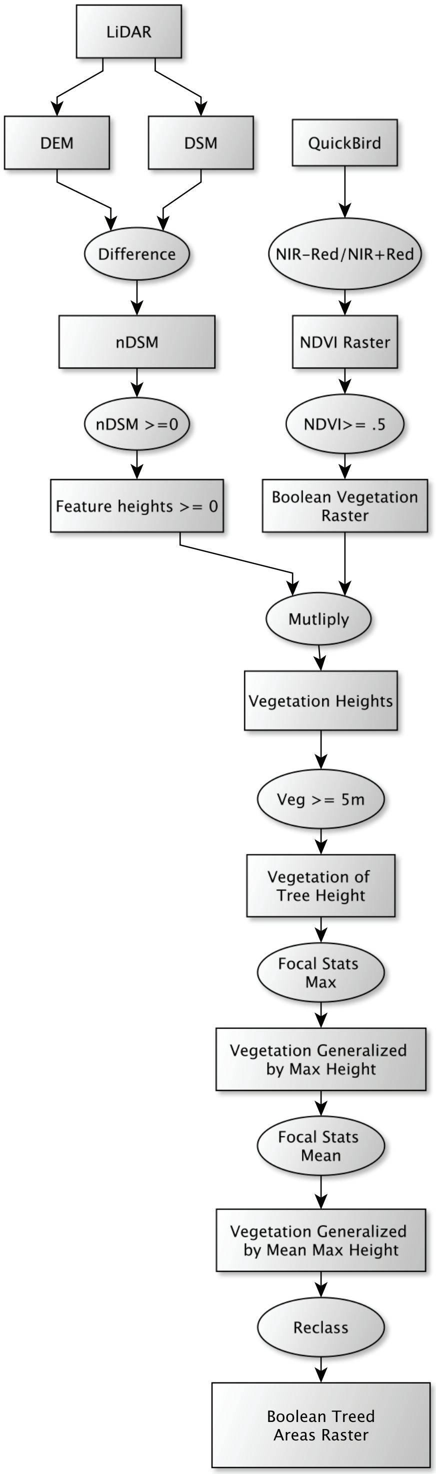


Figure C.2 Detailed steps in *ArcGIS* processing to identify treed areas. This figure explains in detail Step 1 of Figure C.1.

C.1.2.2 MODIFICATION OF THE FOREST FUEL CODE LAYER

To include the detailed forest edge identified using remote sensing data (Figure C.1 Step 1), the FFC layer was rasterized and all tree fuel codes were expanded, growing into the non-fuel areas, while leaving tree fuel borders unaltered. The expanded fuel codes layer was then multiplied with the tree raster, advancing the FFC codes to the actual forest edge (Figure C.1, Step 2; C.3). Non-fuels, and non-tree cells were re-assigned the same value the cell initially contained by adding a raster of non-tree values multiplied with the original fuel codes to create a continuous raster FFC layer. To move the forest edge inwards where it lay inward from the original provincial-scale raster forest edge, rather than outwards, the urban area fuel code was expanded outwards and multiplied with the non-tree Boolean raster. Urban fuel codes in the resulting raster were reclassified to a value of 0, and all other fuel codes were reclassified to a value of 1. This new raster was multiplied with the expanded tree fuels raster to eliminate forest edges that were artificially advanced into urban areas (Figure C.3). Any areas of new development that did not have an urban fuel code due to the age of the data were artificially added at this point by multiplying a raster where these features had a cell value of 0 with the FFC raster. The raster values of 0 were reclassified as a non-fuel FFC, to create a continuous surface of fuel codes in only burnable areas, allowing wildfire to penetrate the urban area. This replaced the urban fuel code, which had no associated fire behaviour. Right-of-ways were rasterized and added with a fuel code of non-fuel to increase the detail of the map, which originally only included highways, and showed fewer fuel breaks than existed in the area (Figure C.3).

The modified fuel code raster layer was vectorized and city blocks, derived from a polyline to vector process on HRM street polylines, were used to select street trees. HRM-owned properties were removed from the selection, based on the assumption that they were park properties, and that these trees are natural forest remnants, which would be accurately represented by the existing forest fuel codes. The remaining selected trees were examined with an aerial *QuickBird* image to determine whether or not they were remnant patches. Remnants were removed from the selection, and the remaining trees were reclassified as deciduous species, to accurately represent street trees. The layer was then rasterized (Figure C.3).

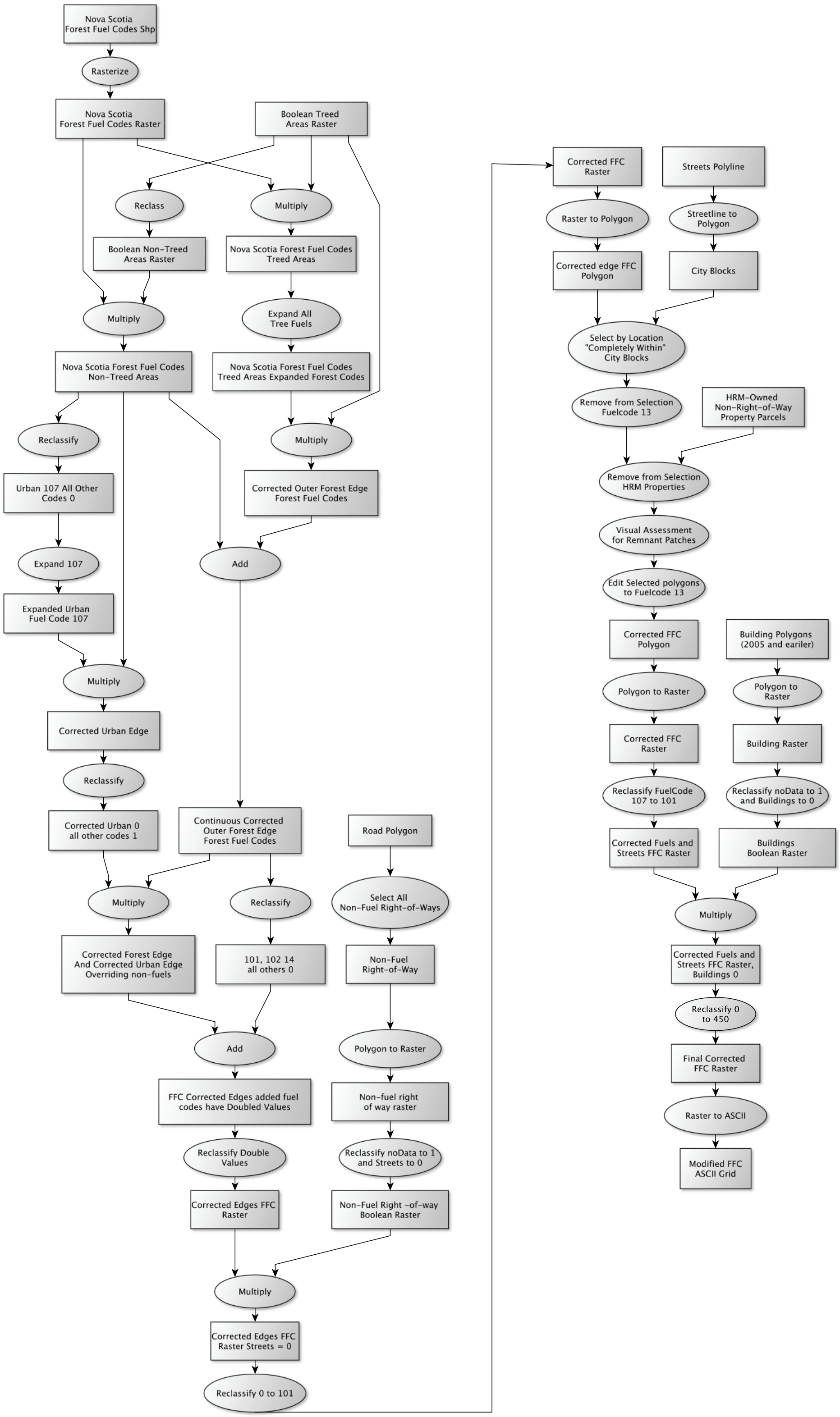


Figure C.3 Detailed steps in *ArcGIS* processing modify the provincial forest fuel code data to include an accurate forest edge. This figure explains in detail Step 2 of Figure C.1.

For both Spryfield and Beaver Bank the final step in updating the FFC layer was the addition of houses. To match the location of development to the *QuickBird* and LiDAR from this time building polygons digitized in or before 2005 were added. These building polygons were rasterized and assigned a value of 0, with the rest of the landscape receiving a value of 1. In raster calculator this layer was multiplied with the modified, expanded FFC layer and cells receiving a value of 0 were reclassified to a fuel code number representing a 50% mixedwood fuel. In this way buildings were included as an ignitable fuel. This produced the final modified FFC layer used in fire behaviour modelling. This modified layer was then converted into an ASCII grid for use in *Burn-P3* (Figure C.3).

C.1.2.3 FIRE SUSCEPTIBILITY MODELLING AND WILDLAND-URBAN INTERFACE DELINEATION

Burn-P3 is a deterministic simulation model used to model fire susceptibility over a landscape. The model uses inputs of ASCII fuel code grids; tables detailing the associated wildfire behaviours; landscape data in the form of a DEM ASCII grid; and historical fire ignition data, also in ASCII grid form (Canadian Interagency Forest Fire Centre, 2011a). All weather and fire ignition data were obtained from the NSDNR Wildfire Management Group. A DEM of the study area was used in *Burn-P3* to model landscape effects on wildfire, such as slope and aspect (For additional detail see section C.1.1 Data and processing). Rasters representative of local weather station areas of coverage, historical fire occurrences in the region, and areas of likely fire occurrences by fire season were obtained from the NSDNR Wildfire Management Group. Raster inputs were resampled to create cell sizes of 5m x 5m and to ensure that each raster grid was of an identical extent, and to meet the requirements for simulation in *Burn-P3*. These processed rasters were then used to create ASCII files from the modified Nova Scotia FFC map created in Step 2, a DEM, spring and summer fire ignitions, weather stations, and lookup tables and a data dictionary of fuel codes and wildfire behaviours, fire occurrence information, the distribution of spread event days and tables of fire weather recorded by local weather station were entered into *Burn-P3* (Figure C.1, Step 3; C.4).

Burn-P3 produced an ASCII surface showing the number of times each raster cell in the study area burned in modelling. This output was used in *ArcGIS* to calculate FS, as the percentage of times a single raster cell burned after 2000 fires were simulated (Equation 1.6) (Figure C.4).

I delineated the WUI by identifying the FS value at -1 standard deviation from the mean FS, and clipping it to the community boundaries in question (Figure C.1, Step 4; C.4). This model allowed me to map and define the WUI using HRM-specific, highly accurate data.

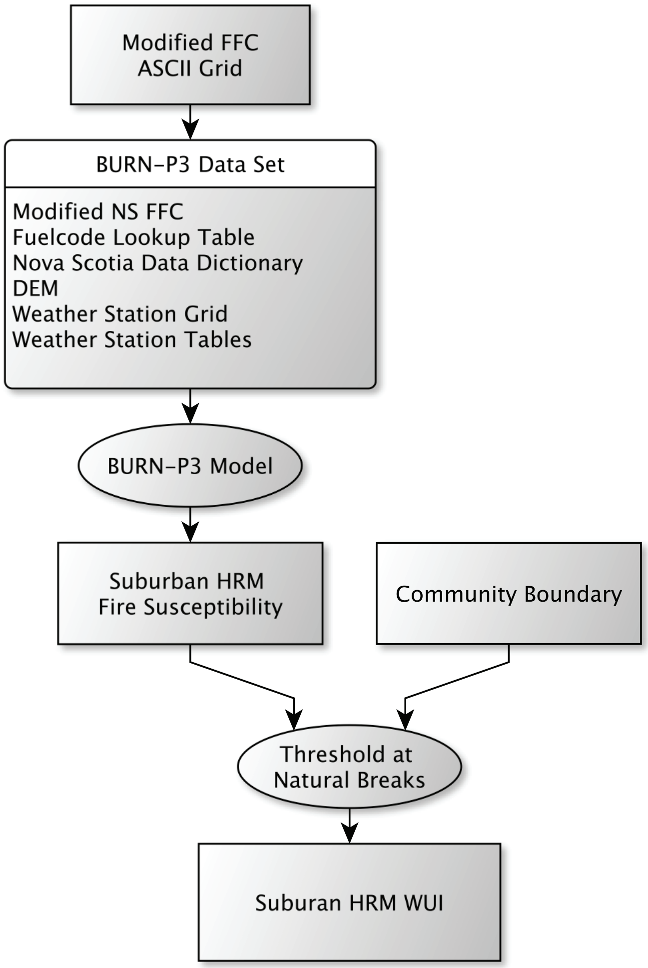


Figure C.4 Detailed steps in *ArcGIS* processing and modelling to calculate fire susceptibility values in the study communities of Spryfield and Beaver Bank. This figure explains in detail Steps 3 and 4 of Figure C.1.

C.1.3 ASSUMPTIONS

To ensure proper operation of the *Burn-P3* model several assumptions were made about local fuel codes and fire behaviour. Weather in Nova Scotia is not such that fires can happen at any time, and to consider all weather would produce probabilities of wildfire too low to interpret. For the purpose of this study only extreme fire weather conditions were considered. All fire weather inputs were taken from real historical, local data, but these values represent a 'fire season' rather than all local weather for an entire year. Associated with the use of extreme fire weather as inputs, only fires greater than or equal to 5 ha were burned in the model. This was to replicate the behaviour of large fires, which spread over large areas and distances. Many smaller fires occur, and these were not considered as they typically do not threaten suburban areas and are easy to suppress if they do. Because of this, the probability maps and values are for ignition probabilities of large and potentially catastrophic fire, rather than any wildfire.

Assumptions about Nova Scotian fuels that are not natively represented in the *Burn-P3* model were based on advice from NSDNR Risk Services, Forest Protection staff. Three fuel codes found in Nova Scotian FFC mapping were approximated from existing fuel codes with similar fire behaviours. The NS1 fuel code (Nova Scotia Special, scrubby brush) was modelled with fire behaviour of the M1-75 fuel code, and with no ignitions in the summer season. The SF (Seasonal Fuels) fuel code was modelled with the fire behaviour of the D1 fuel code, and the CC10 (former clear cut) fuel code was modelled with the fire behaviour of the M1/M2-50 fuel code. The fire behaviours of these codes are representative of the actual fire behaviour in these Nova Scotia-specific fuels, but it is an approximation only.

Appropriate treatments of man-made urban features in *Burn-P3* were elicited from forest and/or fire experts in April 2012. Paved rights-of-way and parking areas were treated as non-fuel, as well as all urban areas, to reflect the negligible risk of fire in manicured lawns. Buildings were treated as 50% mixedwood fuel, demonstrating their capacity to be ignited. Finally, "curing" or drying values for grass and shrub layers were lowered from the default of 60% to 20% to best represent the local grassland fire behaviour. In combination with the fact that trees and shrubs lower than 5m – potential ‘ladder fuels’ – were not identified as urban forest fuels in Step 1 of Section 1.3.2, the above may underestimate risk.

The model inputs used to produce the FS values were derived from fire weather values that represent dry, hot days with minimal precipitation or ‘fire weather’. The FS values produced by this study should be interpreted as the probability of that cell igniting on a day with a high probability of fire occurrence in general, rather than a value representing total long-term risk in that location.

C.1.4 GENERALIZATION AND SCALE MIS-MATCHES

Combining fine-detail data (such as the LiDAR raster) and broader data (such as the FFC layer), conversions from polygon to raster, and the resampling to a 5m x 5m raster all introduced limitations to this research and tended to generalize data. Some generalization is required, whether for the sake of relevance, scale of the research, or processing speed in a model, such as *Burn-P3* (Droppová, 2011; Li, Wilkinson, & Khaddaj, 2001). Despite this requirement, some information and detail is lost through generalization (Droppová, 2011). Although resampling reduced the level a detail in the final ASCII grids, when mixing data of various scales (such as the provincial and municipal datasets used), GIS users generally operate at the limiting scale of the least detailed spatial data, as consolidated data is only as precise as the coarsest input (ESRI, 2012). Scale mismatches are common in urban environments (Bergström, Elmqvist, Angelstam, & Alefsen-Norodom, 2006), and must be managed to the best of researchers’ abilities, though this may lead to loss of detail in the case of generalization.

C.2 FIELD SITE VISITS

Field visits to Spryfield and Beaver Bank were conducted on March 21st, 2012, and May17th, 2012, respectively. Locations for site visits were chosen following fire susceptibility (FS) modelling, and were based on this data. For each community, structures that fell into the highest wildfire risk categories were selected in *ArcGIS*. Selected structures were sorted by increasing value, using the associated polygon ID.

Those structures with the highest polygon ID values were chosen, to randomize the site visit selection. In Spryfield 40 structures were visited, which, due to the common higher FS in these areas, were functionally grouped into seven clusters. In Beaver Bank 42 structures were assessed, which were grouped into five clusters. In each community remnant stands surrounded by development (two in Beaver Bank, three in Spryfield) were also visited to examine the fuel composition and accuracy of both fuel codes and remote sensing data. At a site the proximity of natural trees to structures and general fuel management was noted, as well as the local tree species. This was compared to the data of the modified forest fuel code layer.

Figure C.5 Home and structures built into and alongside forested area in Spryfield, Nova Scotia, March 21st, 2012.

Figure C.6 Coarse woody debris in forested property line in Spryfield, Nova Scotia, March 21st, 2012.

Figure C.7 Area cleared and prepared for development alongside coniferous forest area in Spryfield, Nova Scotia, March 21st, 2012.

Figure C.8 Large home built into forested area in Beaver Bank, Nova Scotia, May 17th, 2012.

Figure C.9 Trailer or manufactured housing development and structures built alongside forested area in Beaver Bank, Nova Scotia, May 17th, 2012.

Figure C.10 Clear cutting slash and regrowth in Beaver Bank, Nova Scotia, May 17th, 2012.

APPENDIX D FIRE SUSCEPTIBILITY DISTRIBUTIONS

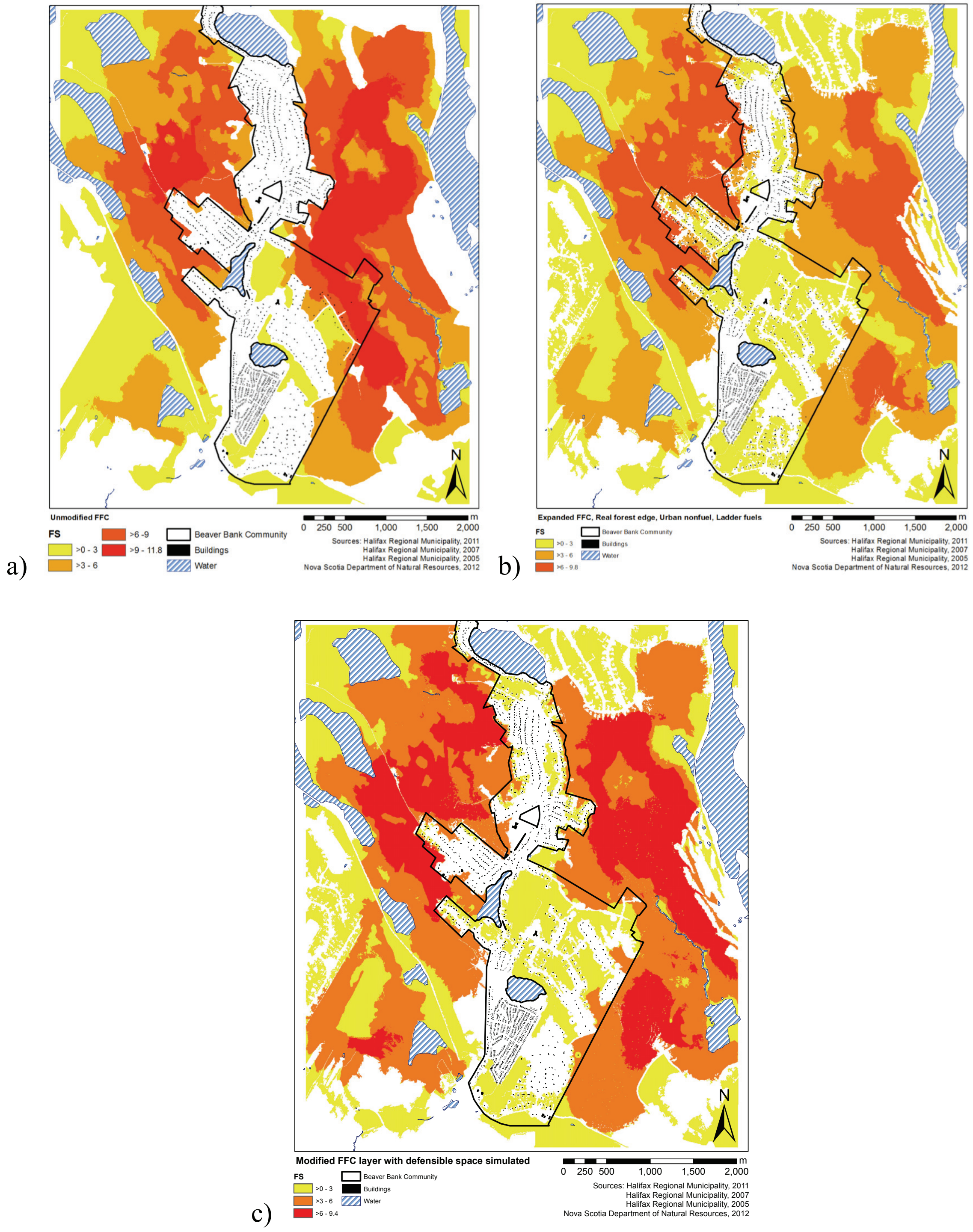


Figure D.1 Fire susceptibility distributions for Beaver Bank generated in *Burn-P3* using a) unmodified, b) modified, and c) modified with a defensible space treatment simulated FFC map layers with fire weather from local weather stations.

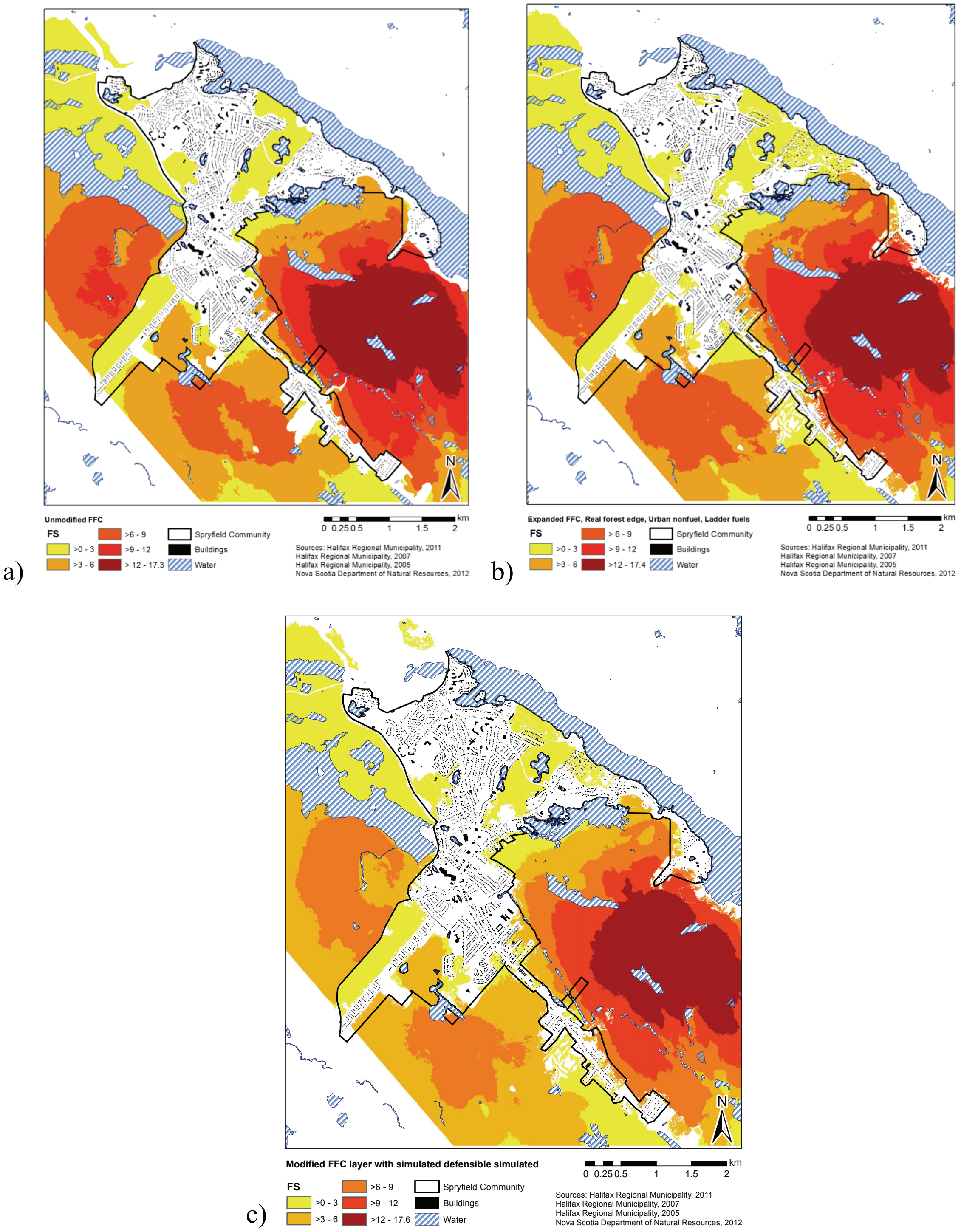
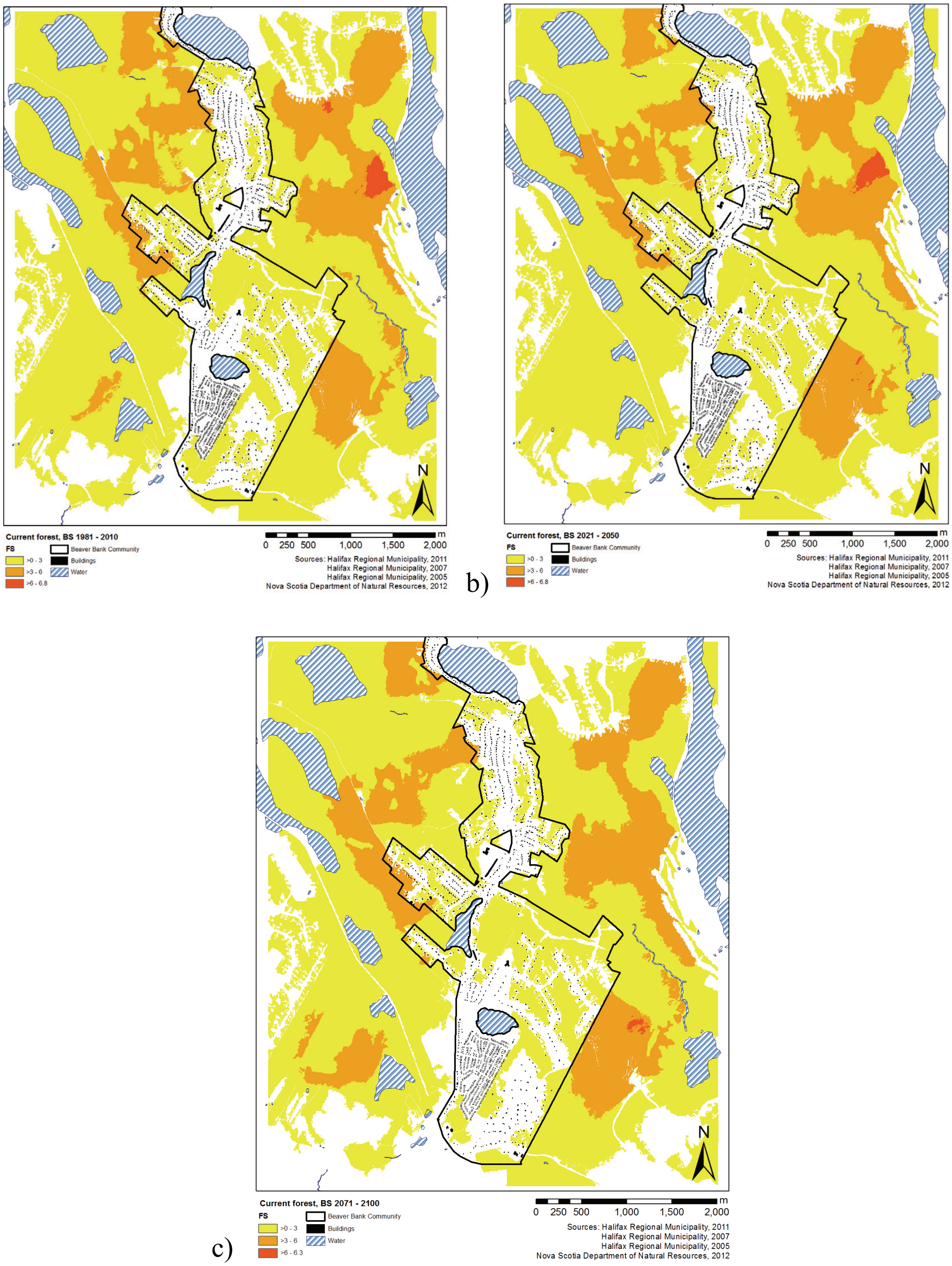
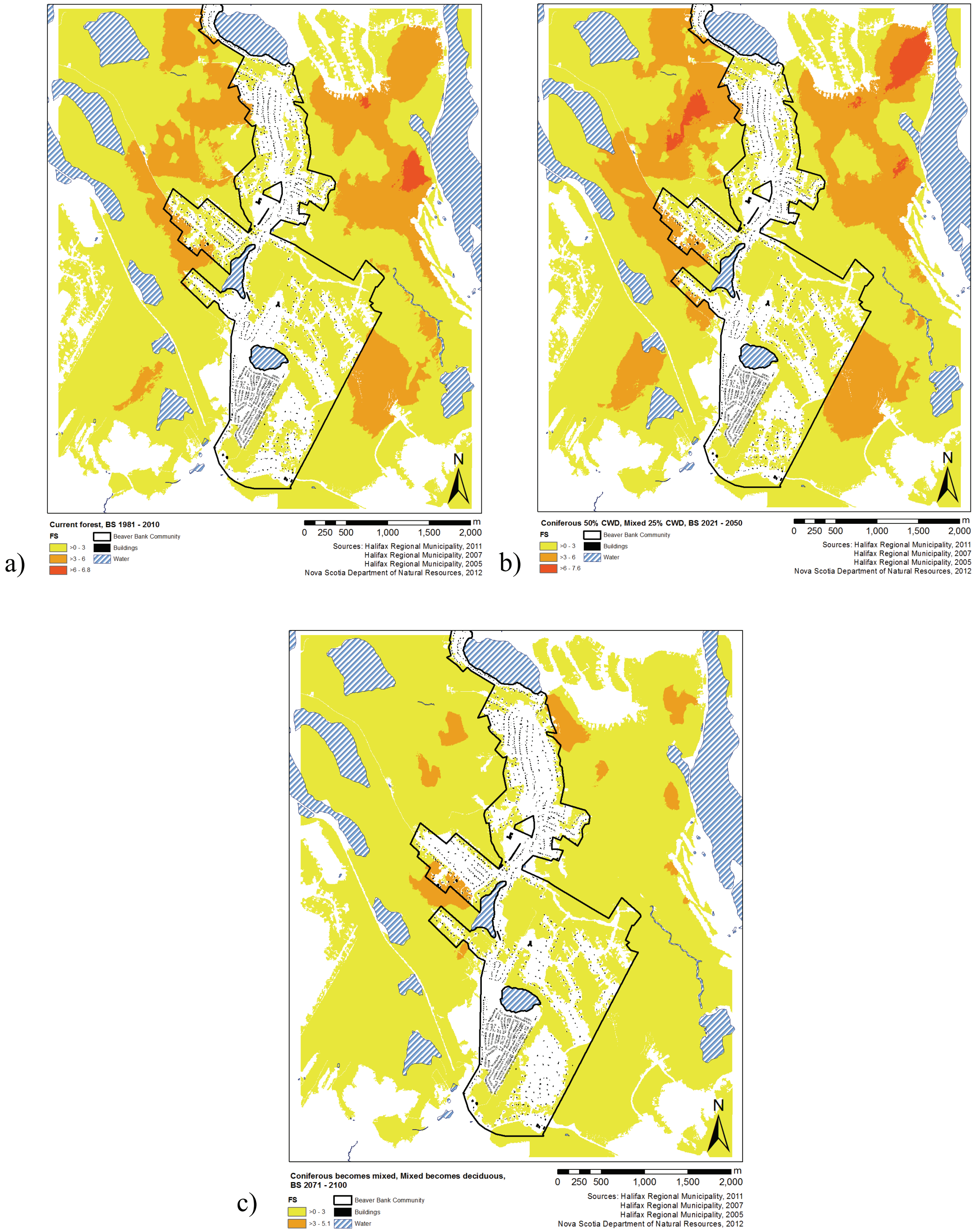


Figure D.2 Fire susceptibility distributions for Spryfield generated in *Burn-P3* using a) unmodified, b) modified, and c) modified with a defensible space treatment simulated FFC map layers with fire weather from local weather stations.



a) b) c)

Figure D.3 Beaver Bank Scenario 1: Climate change only at a) 1981 – 2010 climate normals conditions, b) 2021 – 2050 climate normals conditions; and, c) 2071 – 2100 climate normals conditions.

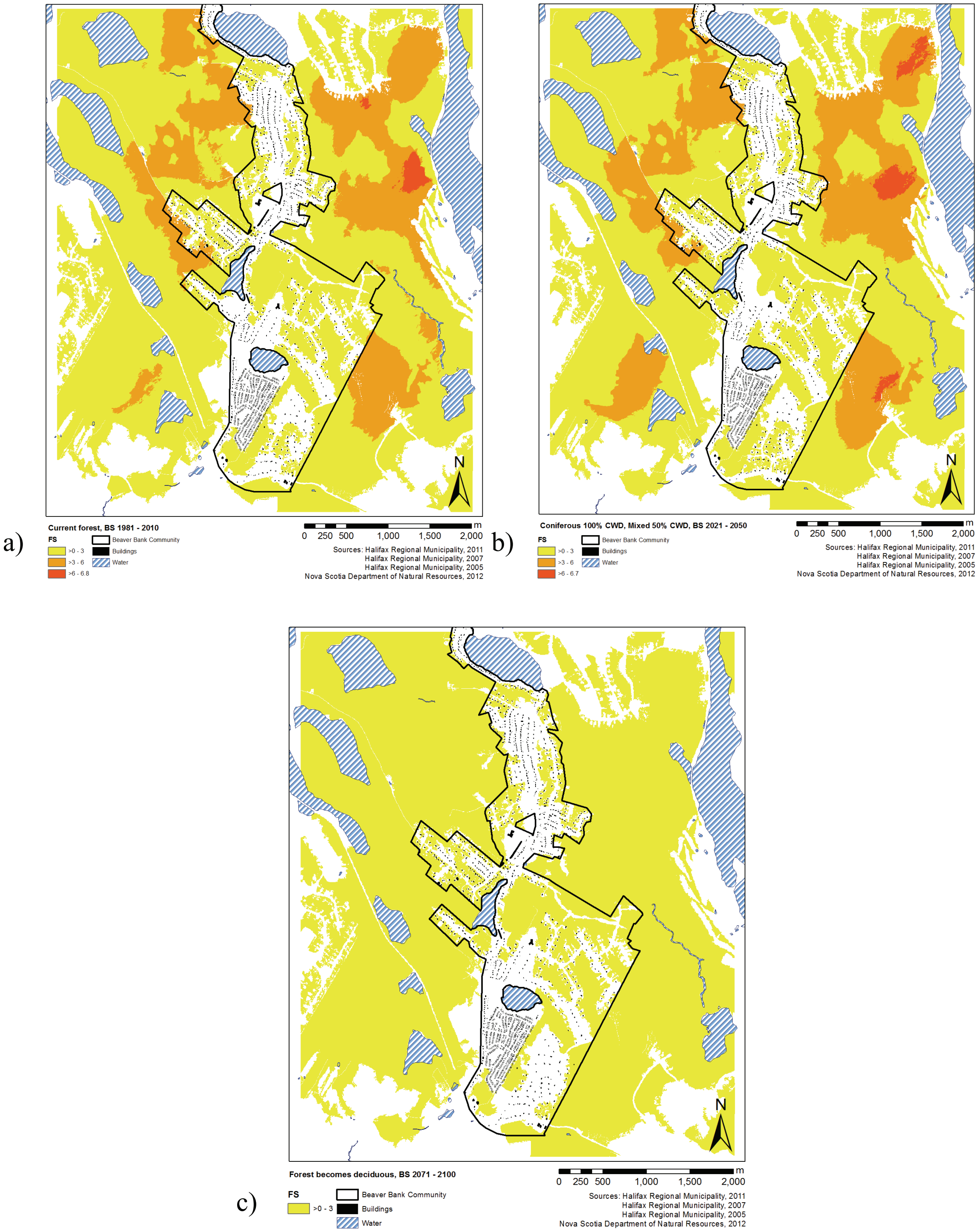


Sources: Halifax Regional Municipality, 2011
Halifax Regional Municipality, 2007
Halifax Regional Municipality, 2005
Nova Scotia Department of Natural Resources, 2012

Sources: Halifax Regional Municipality, 2011
Halifax Regional Municipality, 2007
Halifax Regional Municipality, 2005
Nova Scotia Department of Natural Resources, 2012

Sources: Halifax Regional Municipality, 2011
Halifax Regional Municipality, 2007
Halifax Regional Municipality, 2005
Nova Scotia Department of Natural Resources, 2012

Figure D.4 Beaver Bank Scenario 2: Climate change and moderate fuel changes at a) 1981 – 2010 climate normals conditions, b) 2021 – 2050 climate normals conditions; and, c) 2071 – 2100 climate normals conditions.



Sources: Halifax Regional Municipality, 2011
Halifax Regional Municipality, 2007
Halifax Regional Municipality, 2005
Nova Scotia Department of Natural Resources, 2012

Sources: Halifax Regional Municipality, 2011
Halifax Regional Municipality, 2007
Halifax Regional Municipality, 2005
Nova Scotia Department of Natural Resources, 2012

Sources: Halifax Regional Municipality, 2011
Halifax Regional Municipality, 2007
Halifax Regional Municipality, 2005
Nova Scotia Department of Natural Resources, 2012

Figure D.5 Beaver Bank Scenario 3: Climate change and extreme fuel changes at a) 1981 – 2010 climate normals conditions, b) 2021 – 2050 climate normals conditions; and, c) 2071 – 2100 climate normals conditions.

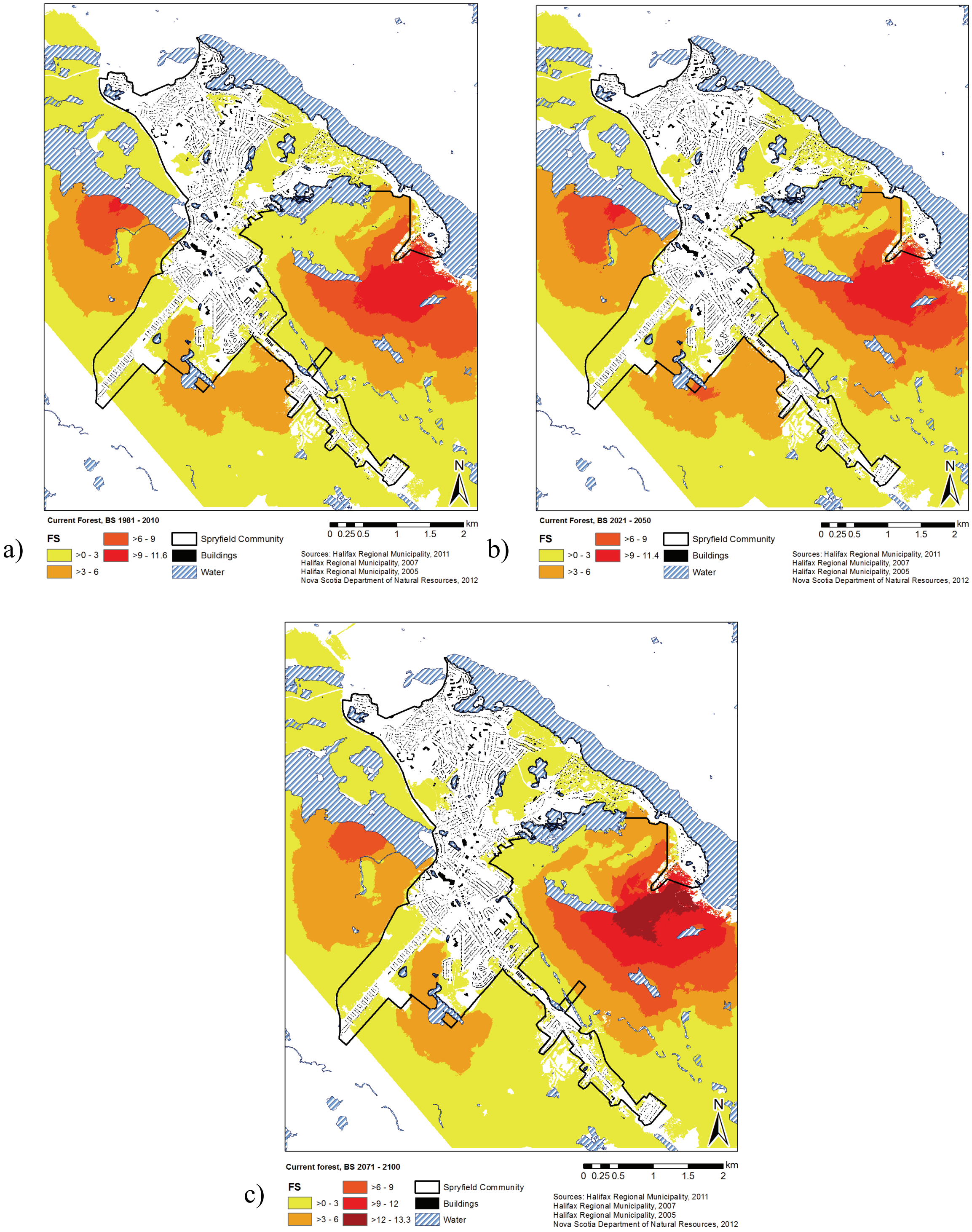


Figure D.6 Spryfield Scenario 1: Climate change only at a) 1981 – 2010 climate normals conditions, b) 2021 – 2050 climate normals conditions; and, c) 2071 – 2100 climate normals conditions.

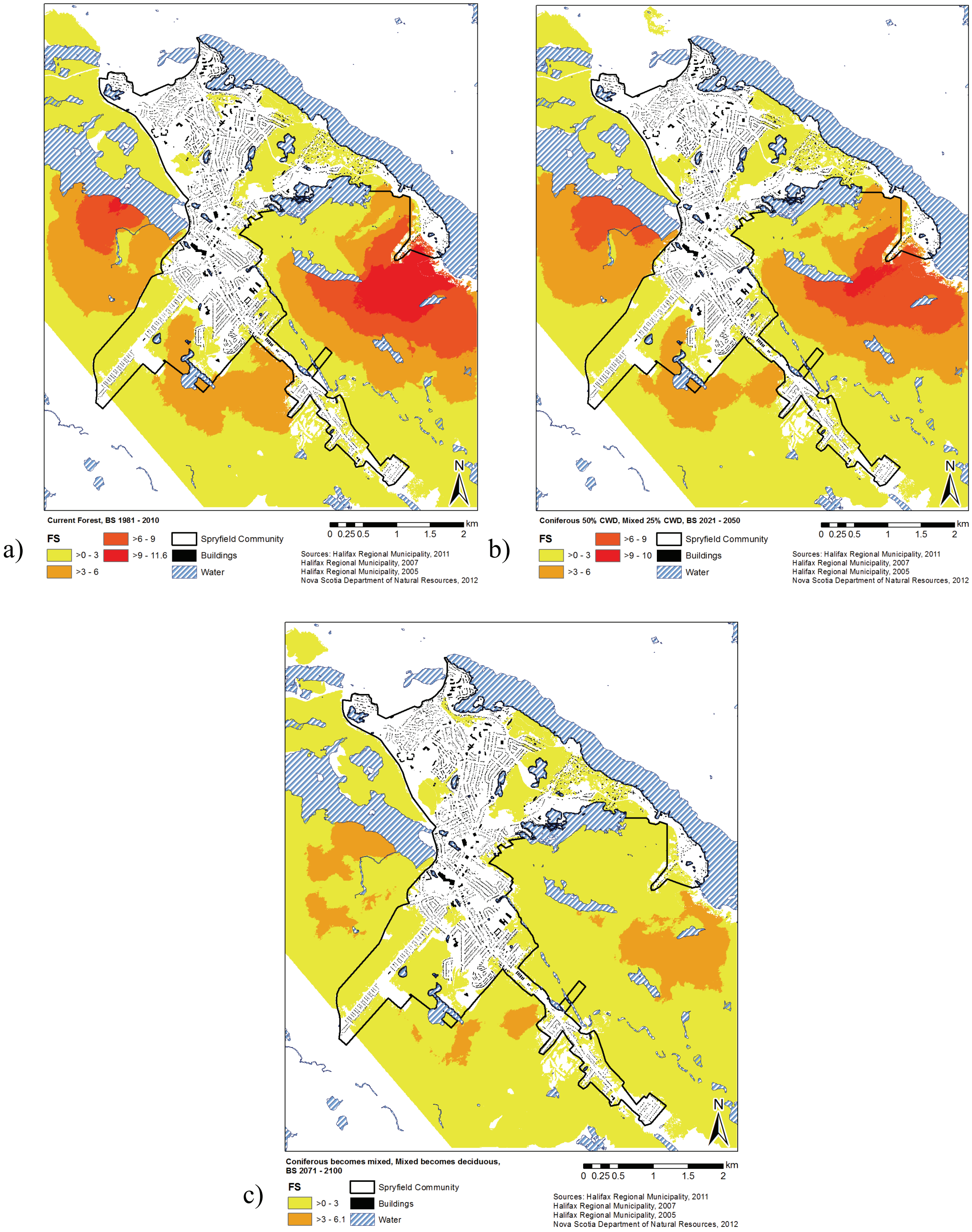


Figure D.7 Spryfield Scenario 2: Climate change and moderate fuel changes at a) 1981 – 2010 climate normals conditions, b) 2021 – 2050 climate normals conditions; and, c) 2071 – 2100 climate normals conditions.

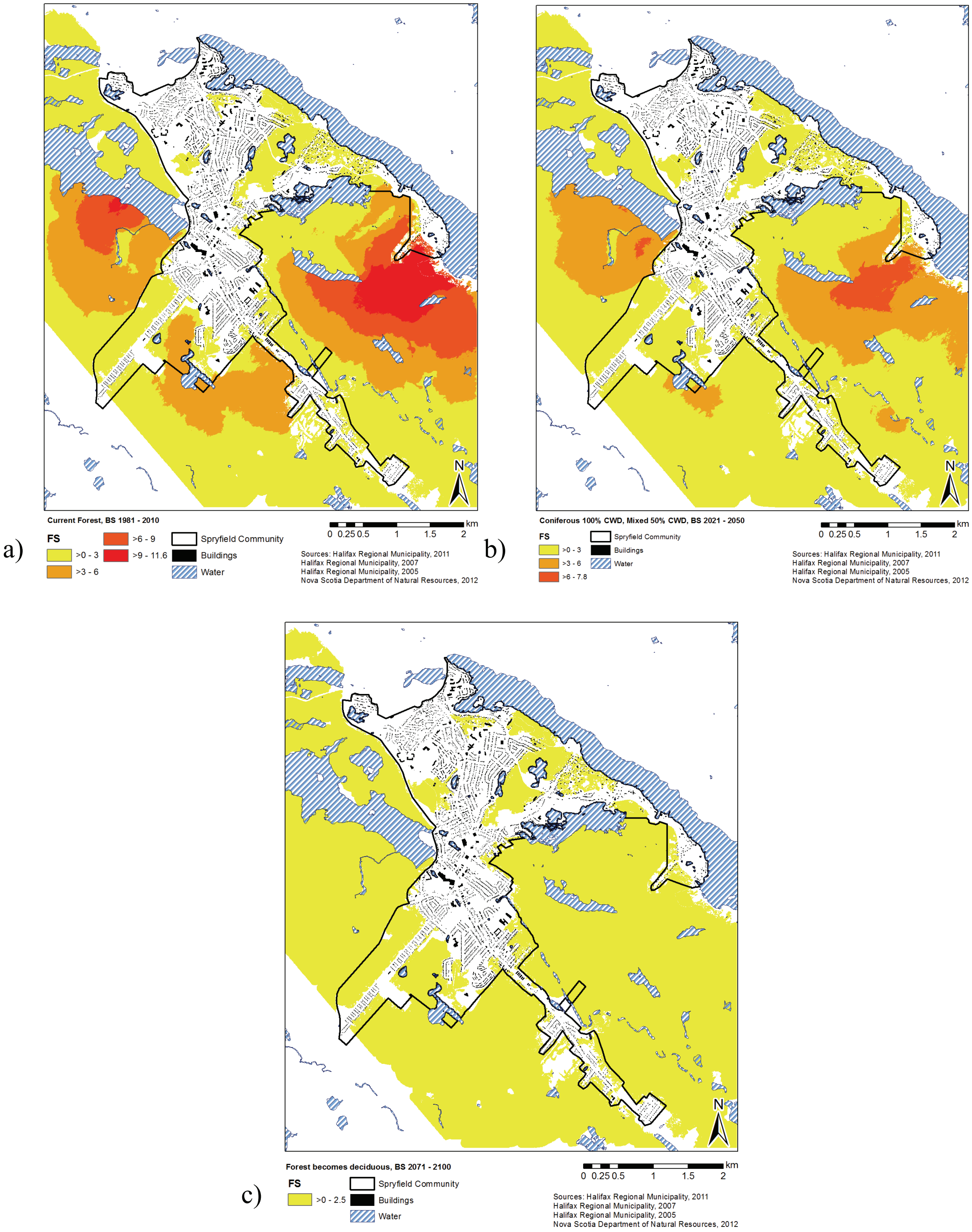


Figure D.8 Spryfield Scenario 3: Climate change and extreme fuel changes at a) 1981 – 2010 climate normals conditions, b) 2021 – 2050 climate normals conditions; and, c) 2071 – 2100 climate normals conditions.