Forest Dynamics and the Application of a Natural Disturbance-Based Management Model in Duck Mountain Provincial Forest, Manitoba

By

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A Thesis submitted to the Faculty of Graduate Studies of the
University of Manitoba
in partial fulfillment of the requirements of the degree of

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ABSTRACT

The forest industry has been moving towards the adoption of ecosystem-based forest management techniques to achieve sustainable forest management. Different models have been developed in an attempt to incorporate increasingly diverse management goals. This study utilizes data from a pilot Forest Land Inventory (FLI) i) to identify major successional pathways in Duck Mountain Provincial Forest of western Manitoba, ii) to analyze tree species cover variability within each pathway, iii) to assess the successional variability of forest stands originating from large, catastrophic fires that occurred in the 1880s and 1890s, and iv) to assess the applicability of the three structural cohort natural disturbance based management (NDBM) model, developed for the mixed boreal forest of Quebec. Cluster analyses were performed to classify the upper forest canopy into 3 major successional pathways: Trembling aspen-white spruce, jack pine-black spruce, and black spruce-eastern larch. Ordination analysis was used to determine the relationship among vegetation, environmental, and structural variables within each major pathway. Results suggested that the FLI environmental variables distinguish well among the three major pathways primarily along a moisture/slope gradient, but explain little of the species and structural variability within each pathway. Forest stands originating from large fires in the 1880s and 1890s were analyzed to assess successional variability within each major pathway. Results showed that despite being of similar age, there was a large amount of variability in structural development, but 2-layered canopies with a continuous to discontinuous upper canopy tended to dominate in these stands. The landscape was then classified into structural cohorts. Globally, 59.5%, 34.5%, and 6.0% of the landscape was in cohort 1, 2, and 3, respectively. The three structural cohort management model was applied to the forest under three different scenarios: (1) the current distribution of cohorts, (2) a 110-year fire cycle (current), and (3) a 60-year fire cycle (pre-European settlement). Results suggest that, in terms of forest structure, the current distribution of cohorts in DMPF is closest to the expected distribution under a 60-year fire cycle. The results are compared to findings in other regions of the mixed boreal forest, and the implications for forest management in DMPF are discussed.

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DEDICATION

This thesis is dedicated to Davian, who has always believed in me.

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1. INTRODUCTION

1.1 The mixed boreal forest

The boreal forest is the world's largest terrestrial biome, representing approximately 32% (14 million km^2) of the planet's forest cover (Burton et al., 2003). Boreal forests comprise about 83% of Canada's total forest and 35% of Canada's total land area (CCFM, 2000), and boreal forest-related industry is estimated to make up approximately 60% of Canada's overall forest economy (Burton et al., 2003). Although the boreal ecosystem as a whole is generally similar in appearance and shares similar dominant canopy species, there are important regional differences in terms of which species attain dominance (Larsen, 1980). These differences are primarily determined by changes in physiography and climate (Wiken, 1986; Elliott-Fisk, 1988; Barnes et al., 1998), and in Canada the boreal forest is subdivided into seven ecozones as defined by Wiken (1986) (Fig. 1a). Often the boreal forest is classified according to the dominant form of vegetation cover, or the physiognomic-structural classification (Scott, 1995; Barnes et al., 1998). According to the relative dominance of coniferous and broadleaf species, Canada's boreal forest can be classified into (1) coniferous forest; (2) mixed forest, dominated by both coniferous and broadleaf-deciduous species; and (3) deciduous forest (Fig. 1b). A fourth forest type, the transitional forest, corresponds to the boreal forest–tundra ecotone, which represents a transitional zone between the northern coniferous boreal forest and the shrub-dominated tundra (Scott, 1995).

The mixed boreal forest is a dynamic system, and may be defined as forests that are dominated by both broadleaf species and coniferous species at some point throughout

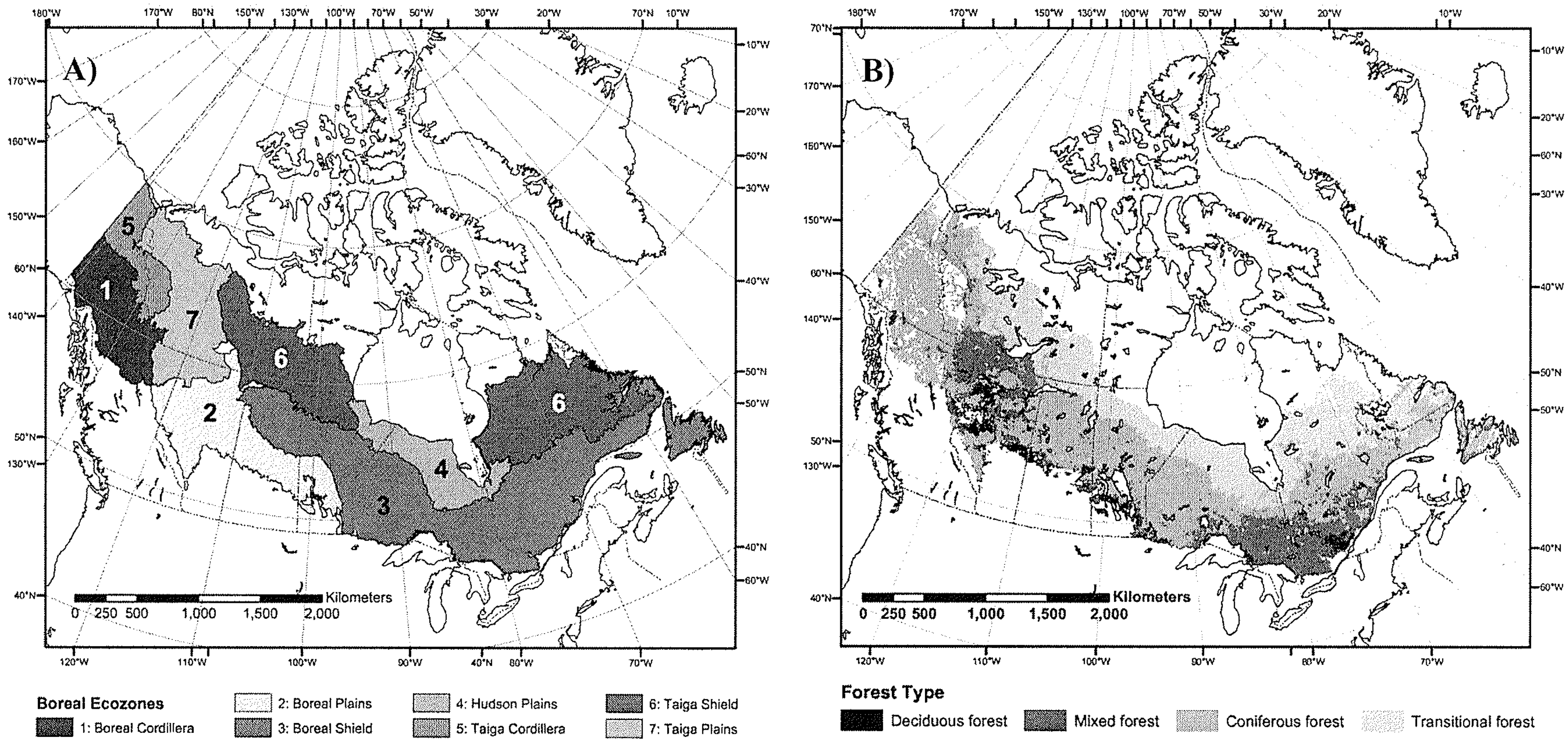

Figure 1. A) The boreal forest ecozones of Canada (data source: Schut, 2005), and B) the boreal forest types of Canada (data source: St-Laurent et al., 1995).

the forests' successional stages, and where no single species comprises greater than 80% of the basal area (MacDonald, 1995). The mixed boreal forest constitutes approximately 18% of the Canadian boreal forest region (St-Laurent et al., 1995). This forest type is concentrated along the southern fringes of the boreal shield and boreal plains ecozones, as well as in the southern portion of the taiga plains (Scott, 1995) (Fig. 1a, b). Compared to the more northern, conifer-dominated forests, the mixed boreal forests tend to be richer and more diverse in terms of species as well as in soil conditions (MacDonald 1995; Côté et al., 2000). Dominant coniferous tree species in the mixed boreal forest include black spruce (*Picea mariana* (Mill.) BSP), white spruce (*Picea glauca* (Moench) Voss), balsam fir (*Abies balsamea* (L.) Mill.), jack pine (*Pinus banksiana* Lamb.), eastern larch (*Larix laricina* (Du Roi) K. Koch), and northern white-cedar (*Thuja occidentalis* L.). The dominant broadleaf species include trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.), paper birch (*Betula papyrifera* Marsh.), and willow (*Salix* spp.) (Rowe, 1972; Larsen, 1980; Gauthier et al., 2000).

The biological and environmental conditions associated with the mixed boreal forest vary across Canada. The eastern region (eastern Ontario and Quebec) of the mixed boreal forest tends to be somewhat richer in species compared to the western regions (Manitoba, Saskatchewan, and Alberta), as it is part of the Boreal-Broadleaf Ecotone, which is a transition zone possessing environmental conditions suitable to both the coniferous trees of the north, and the increasingly dominating broadleaf deciduous trees to the south as growing conditions improve (Rowe, 1972; Scott, 1995). In contrast, the western mixed boreal is part of a transition between the northern boreal forest and the drier grasslands and aspen parklands to the south (Rowe, 1972; Wiken, 1986; Scott, 1995). Rather than a transition to better growing conditions dominated by broadleaf forests as is the case in the east, the mixedwood state in this region is determined by a combination of factors, including moisture stress and shorter fire cycles which help to define the southern limit of the boreal conifer species in this region (Rowe and Scotter, 1973; Scott, 1995).

1.2 Dynamics of the mixed boreal forest

The process of succession and its interruption by disturbances drives the dynamics of a forest landscape. Forest stand dynamics encompasses the changes in forest composition and structure over time during and after disturbances (Chen and Popadiouk, 2002). Environmental differences across the range of the mixed boreal forest have important implications for the development of the forest and the factors affecting forest dynamics (i.e. disturbances, growth). Girardin and Tardif (2005) have shown that in Manitoba, the growth of boreal tree species are not only responding to variation in temperature and precipitation, but also reflect variation in atmospheric pressure heights as well. Climatic trends have a strong influence on the frequency and severity of disturbances such as fire (Flannigan and Van Wagner, 1991; Johnson and Larsen, 1991; Carcaillet et al., 2001). Atmospheric circulation patterns have been linked to differences in drought severity patterns in historical reconstructions, which in turn strongly influence the potential for fire initiation (Girardin et al., 2004a). Climate also determines conditions that may be favourable or unfavourable to other types of disturbances such as insect outbreaks which also play an important role in forest dynamics (Hogg et al., 2002; Sutton and Tardif, 2005; in press). As a consequence of the climatic variability existing across the mixed boreal forest of Canada, there is a significant amount of variability in the regional disturbance regime, and consequently, in forest dynamics.

1.2.1 Disturbances

There are three major factors that determine the forest mosaic over a landscape: geomorphology and regional climate, both of which affect vegetation composition at the regional level (Bridge and Johnson, 2000), and disturbances, which can be either endogenous (from within the system) or exogenous (from outside the system). Disturbances can affect vegetation composition on a stand or landscape level (Attiwill, 1994). Fire and insect outbreaks, such as spruce budworm, are some of the most important natural disturbances in the mixed boreal forest ecosystem, largely determining the age distribution and composition of the forest mosaic over the landscape (Morin, 1994; Gauthier et al., 1996; Bergeron et al., 2002b; Wallenius et al., 2004).

The fire cycle of a given region can be defined as the number of years required to burn an area equal to the size of the study area (Johnson and Gutsell, 1994), and it varies spatially, temporally, and in intensity (Bergeron et al., 2004a). With the global increases in temperature, it is believed that the frequency and severity of fires will increase. Using general circulation models, Flannigan and Van Wagner (1991) have predicted a potential 40% increase in seasonal fire severity and area burned under a $2 \times CO_2$ climate scenario. Flannigan et al. (2001) have, however, reported variable trends in fire weather indexes (FWI) between western and eastern Canada as global temperature increases. Evidence suggests that the length of fire cycles in the mixed boreal forest of eastern Canada has increased since the end of the Little Ice Age (~1850) despite global warming (Bergeron et al., 2001; Lesieur et al., 2002; Bergeron et al., 2004a). The increase in temperature in this region is likely offset by an increase in precipitation resulting from a shift in atmospheric circulation (Girardin et al., 2004a, b).

Based on fire history work done in the eastern mixed boreal forest by Bergeron et al. (2001) and Gauthier et al. (2002), estimates of the average fire cycle in the past century ranges from >190 years to >400 years. In the western mixed boreal forest, dendroecological evidence suggests that there was also an increase in the fire cycle at the end of the Little Ice Age (Larsen, 1997; Weir et al., 2000). Historically, the fire cycle in the western mixed boreal forest has been shorter than in the east, ranging from 52 to 118 years over the past century (Larsen, 1997; Bergeron et al., 2004a; Tardif, 2004). Anthropogenic influences such as active fire suppression have made estimates of the current "natural" fire cycle difficult in many study areas throughout the mixed boreal forest, as this activity may potentially influence apparent fire return intervals (Bergeron et al., 2004a; Tardif, 2004). It has been shown that there are potentially one or more shifts (increases or decreases) in fire frequency possible within the period of a stand's development, and as a consequence the forest mosaic, which is governed by this system, is unlikely to achieve a developmental equilibrium with the fire cycle (Weir et al., 2000).

Insect outbreaks also play an important role in the mixed boreal disturbance regime (Johnson et al., 2003; Brassard and Chen, 2006). While fire disturbances have the potential to damage and cause mortality to all vegetation in a given area, insect outbreaks are host-specific. With respect to conifer species, the most important insect disturbance on the landscape is spruce budworm (*Choristoneura fumiferana* Clem.) which tends to occur in large, cyclical outbreaks (Johnson et al., 2003). As stand age surpasses the fire cycle, shorter-lived intolerant species tend to give way to longer-lived tolerant species in the dominant canopy layer (De Grandpré et al., 2000). Spruce budworm plays an important role in recycling late-successional species on the landscape, particularly balsam fir and white spruce (Morin, 1994; Bergeron, 2000; Nealis and Régnière, 2004). Morin (1994) found that the spruce budworm outbreaks and balsam fir forests tend to form an interdependent, self-regulating system in the absence of stand-replacing fires. However, mortality and openings caused by outbreaks can also increase light penetration and favour the recruitment of early-successional species such as trembling aspen into the stand (Nealis and Régnière, 2004). Spruce budworm outbreaks tend to play a more important role in the eastern mixed boreal forest compared to that in the west, because the forest is generally older in the east, and there is a greater proportion of balsam fir, which is spruce budworm's primary host (Su et al., 1996; Bergeron et al., 2004a; Pham et al., 2004).

One of the most important defoliators of broadleaf trees in the mixed boreal forest is the forest tent caterpillar (*Malacosoma disstria* Hbn.), whose primary host is trembling aspen (Peterson and Peterson, 1992). Forest tent caterpillar (FTC) is not necessarily a direct cause of mortality in aspen, but acts as a stressor that, when combined with other stressors such as drought, can reduce growth and predispose the trees to damage from pathogens, eventually leading to mortality (Hogg et al., 2002). Like spruce budworm, FTC outbreaks are also cyclical in nature (Hogg et al., 2002; Sutton and Tardif, 2005; in press). There are many factors that influence the dynamics of FTC outbreaks. The severity and duration of FTC outbreaks tend to be greater in forests that have been fragmented to some degree, which may be due to a reduction of the mortality agents that lead to the collapse of FTC populations (Cooke and Roland, 2000; Roland, 2000). A higher severity of forest tent caterpillar outbreaks has been associated with warmer, drier conditions which favour the insect (Hogg et al., 2002). Because of the drier conditions in the western region of the mixed boreal forest, it follows that FTC disturbances are potentially more severe in this region when compared to the east. Mechanical disturbances such as wind also play an important role within the mixed boreal forest (Brassard and Chen, 2006). Windthrow contributes to the supply of coarse woody debris (CWD), and can alter substrate and microsite conditions which influence regeneration and structural changes in the stand (Brassard and Chen, 2006).

Disturbances create structural variability over the landscape by interrupting the development of the established vegetation, or by making a site available for the establishment of new individuals or communities depending on the type and severity of the disturbance (Pickett and Cadenasso, 2005). Disturbances occur on many different scales, and as a consequence can have very different effects on the development of a forest stand ranging from landscape-level, stand replacing fires, to individual tree mortality due to windthrow, insects, or disease (Barnes et al., 1998). Variability in the degree of disturbance and its interaction with specific site and microsite conditions can lead to multiple successional trajectories for a particular stand, and result in the development of different stand types over a landscape (Lieffers et al., 2003).

1.2.2 Secondary succession in the mixed boreal forest

There are many factors dictating temporal vegetation change at a given site, including type and severity of disturbances, pre-disturbance conditions, abiotic characteristics, vegetation characteristics, and soil (seedbed) characteristics (Bergeron and Dubuc, 1989; Leiffers et al., 1996b; Galipeau et al., 1997; Chen and Popadiouk, 2002). Successional pathways in a particular area can be grouped according to similar abiotic conditions, specifically substrate and drainage characteristics, which partly determine the composition of the post-disturbance cohort (Bergeron and Dubuc, 1989; Gauthier et al., 2000; Chen and Popadiouk, 2002). Compositional changes along a successional pathway are accompanied by a variety of structural changes as well. Following stand replacing disturbances, there tends to be a gradual change from even structured canopies, to a heterogeneous, multilayered canopy with time (Brassard and Chen, 2006).

The overall development of mixed boreal stands can be placed into stages based on structural and compositional changes. Chen and Popadiouk (2002) differentiated between four developmental stages, which are also similar to the stages outlined by Oliver and Larson (1996). These include: (1) **Stand initiation**, following a stand-replacing disturbance. This stage usually involves the domination of fast growing pioneer (intolerant) species such as trembling aspen or jack pine. (2) **Stem exclusion** is observed where individual tree competition becomes important and vertical canopy stratification occurs. (3) **Canopy transition** occurs when mortality due to age or competition begins to break up the pioneer canopy species, and shade tolerant trees in the understory begin to penetrate into the canopy. (4) **Gap dynamics** characterizes the stage where tolerant trees tend to self perpetuate in canopy gaps following individual or small group mortality. This stage is characterized by a complex vertical canopy structure. It is important to note that there are possible variations in the definition of these stages, depending on specific site and forest characteristics (Barnes et al., 1998). Several studies have evaluated the process of secondary succession within the mixed boreal forest across Canada, and there is a large amount of variability across the region (Bergeron and Dubuc, 1989; Bergeron, 2000; Hamel and Kenkel, 2001; Stewart et al., 2001; Green et al., 2002).

In the mixed boreal forest, the dominant early successional species forming the initial cohorts after a stand replacing disturbance often include trembling aspen on fine-textured deposits, and jack pine and paper birch on coarse textured deposits, while black spruce communities dominate on organic peat sites (La Roi, 1992; Hamel and Kenkel, 2001; Lecomte and Bergeron, 2005). Post-disturbance communities of white spruce and balsam fir may also occur on finer textured, moister soils (Lieffers et al., 1996a; Galipeau et al., 1997; Bergeron, 2000; Hamel and Kenkel, 2001; Kenkel et al., 2003). Important late-successional species include black spruce, white spruce, balsam fir, and northern white-cedar, with balsam fir being more abundant in the eastern than the western mixed boreal forests, and northern white-cedar being nearly absent in the west (Bergeron, 2000; De Grandpré et al., 2000; Hamel and Kenkel, 2001).

In the absence of any additional stand replacing disturbance there is a tendency for mixed boreal stands to evolve toward a conifer-dominated stand, initially containing a mosaic of post-disturbance and later successional species, with late-successional conifers becoming more abundant with time, and the accumulation of CWD and snags becoming a major element of stand structure (Bergeron and Dubuc, 1989; Bergeron, 2000; De Grandpré et al., 2000; Hamel and Kenkel, 2001; Brassard and Chen, 2006). Because of the shorter fire cycle in the west, late successional species, particularly balsam fir and northern white-cedar, are less abundant in the western than the eastern mixed boreal forest (Weir and Johnson, 1998; Bergeron et al., 2002b). Subsequently, deciduous species tend to remain more important as time-since-last-fire increases in the western mixed boreal forests (Weir and Johnson, 1998; Hamel and Kenkel, 2001). Studies in this region of the mixed boreal have shown that in the absence of conifer seed sources, pure aspen stands can potentially achieve a self-perpetuating, gap dynamic system (Cumming et al., 2000; Hamel and Kenkel, 2001).

The initial dominance of deciduous species following a stand replacing disturbance is often a function of differential growth rates between early- and late-successional species (Lieffers et al., 1996a; Bergeron, 2000). Using dendroecological analysis, Bergeron and Charron (1994) found that white spruce and balsam fir were recruited into the post-fire cohort within five years following the recruitment of aspen and birch. It was noted that this may partially have been possible because of the pre-fire and surrounding stand composition, which was coniferous, and the small size of the fire